

Chapter 2

Historical Continuity

Abstract The historical continuity of old growth urban forests in the New York, New York, and Philadelphia, Pennsylvania, metropolitan areas was analyzed to reveal the strengths and weaknesses of historical ecology methods for forest dynamics research. The use of paleopalynological techniques to determine forest changes in the time span of the past century to recent millennia requires thoughtful selection of the sampling site but can reveal unrecorded species losses and introductions as well as regional forest changes related to urbanization. Pre-forest clearance witness tree records provide the basis for comparison to ascertain how the modern old growth urban forest differs from the forest primeval. Historical floras from the period immediately following the forest resetting event are catalogs of arboreal species present at the time of old growth urban forest formation that can be used to determine subsequent species diversity changes. Forest remeasurement studies are limited because methodological uniformitarianism and actualism are difficult to achieve due to insufficient detail in recording prior field procedures. However, comparisons of past data and resampling results provide invaluable information on changes in the canopy, subcanopy, and sapling layers that is essential to understanding forest dynamics and historical continuity.

Keywords Urban forest dynamics • Historical forest ecology • Paleopalynology • Witness trees • Historical flora • Forest resampling • Oak-chestnut forest region • New York City • Philadelphia

Introduction

Research methods in historical ecology have been described for Europe (Agnoletti and Anderson 2000) and the United States (Egan and Howell 2001). Of special note is the intensive focus on historical ecology methodology for forests in

Britain (Rotherham et al. 2008). The primary sources for historical ecology research (written documents, paintings, maps, plans, lithographs, historical pictures, aerial photographs, plaques, monuments, and historical events) often have limited value because little is known concerning the perspective and purpose in creating the record (Peter 2008). Providing a land use history is not the goal of historical forest ecology research. Instead, the goal for research on old growth urban forest is to determine forest dynamics. The research objective when exploring historical records is to acquire data on the species and diameters of trees in the canopy, subcanopy, and sapling layers at different points in time. Also, information on tree, sapling and seedling losses to disturbances, diseases and pests, as well as management activities that removed or added trees, saplings, and seedlings is important.

As noted in the first chapter, a comparative analysis of old growth urban forest dynamics across forest types in a metropolis is simplified in regard to climate factors when all of the study sites are in a single forest region. An advantage can be gained when one metropolis is located in proximity to a second metropolis so that comparisons within the forest region are possible. For example, New York, New York, and Philadelphia, Pennsylvania, are separated by 138 km and are in the oak-chestnut region, but New York is in the glaciated section and Philadelphia is in the piedmont section of the region (Braun 1950). From 1600 to 1900, New York and Philadelphia were the two dominant cities in the eastern United States. Regional historical records indicate woodcutting by the end of the American Revolution caused the loss of all forests from the lands that would become the metropolitan areas of New York and Philadelphia (Hoglund 1962). However, two forests are reported to have survived the forest resetting event: William L. Hutcheson Memorial Forest, Rutgers University, New Jersey (Buell et al. 1954) and the Hemlock Forest, New York Botanical Gardens, New York (Rudnicki and McDonnell 1989).

In the oak-chestnut region, the practice of arboriculture with native species of the eastern United States began soon after 1600 and was dominant to the 1850s (Loeb 2010). The presence of a particular native species in a specific old growth urban forest can be related to the history of arboriculture at the locale (DeCandido and Lamont 2004; Fitzgerald and Loeb 2008). Old growth urban forests across the oak-chestnut region have experienced tree species losses caused by diseases introduced from arboricultural planting such as the chestnut blight (*Cryphonectria parasitica* (Murrill) Barr) in 1893 (Pennsylvania Chestnut Tree Blight Commission 1912) and Dutch elm disease (*Ophiostoma ulmi* (Buisman) Nannf.) in 1936 (Hepting 1977).

Historical forest ecology methods are examined in this chapter using examples from the New York City and Philadelphia metropolitan areas. Forest species composition information at the regional level is revealed by the first three methods presented: paleopalynology, early floras, and witness trees. Next, resampling to obtain forest dynamics information on individual old growth urban forests is assessed. As part of the assessment process, a research review and original research

from the oak-chestnut region is given for each of the old growth urban forest types. The closing section of this chapter is an integrative analysis of old growth urban forest dynamics in the Philadelphia and New York City metropolitan areas.

Regional Forest Species Composition

Paleopalynology

The science of paleopalynology examines fossil pollen preserved in peats, lake sediments, and soils. Fossil pollen records have been used in archeology (Dimbleby 1985) and vegetation history spanning the time of human occupation (Behre 1986; Birks et al. 1988; Vera 2000) but not woodland management (Rackham 2003) with one exception (Loeb 1998). A few pollen studies have been performed to analyze the history of regional forest changes during the development of a metropolis (Baker et al. 1978; Loeb 1989a, 1992b, 1998; Peglar et al. 1989; Brande et al. 1990; Seppä 1997).

Undertaking a paleopalynological study without extensive formal training is not a course of action to be pursued. However, learning how to conduct paleopalynological research is not required for the interpretation of pollen diagrams, which is the important component for historical forest ecology research. As noted above, few paleopalynological studies have been conducted in urban settings; therefore, if old growth urban forests are to be studied in a metropolis, then a search for pollen record sites is needed. Once pollen collection and preservation sites have been identified, research into whether the sites have been disturbed (i.e., removal or mixing of the peat, lake sediments, or soil) is essential. Before contacting a paleopalynologist to request an analysis of an undisturbed record, the need for obtaining the most recent portion of the pollen record must be assessed. Traditional core retrieval devices are not effective in sampling the unconsolidated recent sediments representing the modern day. The alternative is a freezing corer designed to control penetration into the sediment so as to retain the entire sample up to and including the sediment–water interface, which is the most recently deposited portion of the record (Loeb 1989a).

Interpretation of pollen diagrams requires some knowledge about pollen deposition. The amount of pollen produced by trees varies by orders of magnitude depending upon the mode of pollination with highest to lowest production being pollination by wind, insects, and self-pollination. Wind-pollinated trees are represented better in pollen diagrams than insect or self-pollinated trees, but tree location closer to the pollen deposition site improves the chances for pollen presence in the core. Increasing areal size of lakes and bogs is related to larger source area for pollen. Organic-rich soils collect pollen from the immediate vicinity. Pollen is transported to a deposition site by the wind and runoff including rainfall intercepted by trees. Although the chemical forming the pollen wall, sporopollenin, usually permits pollen to be well preserved, some genera have thin walls and break apart relatively easily. Differential preservation of pollen can lead to biased pollen samples, which

causes paleopalynologists to discount samples with extensive amounts of corroded pollen. One assumption of pollen record interpretation is all of the pollen grains in a sample are deposited in the same time period but vertical transport of pollen occurs in soils, peats, and lake sediments (Traverse 2007).

The results of a pollen analysis are the counts of the pollen types representing the arboreal and non-arboreal taxa. Samples are displayed in three types of pollen diagrams: percentage, concentration, and influx. The oldest sample is found at the bottom of the diagram and the youngest at the top. Percentage diagrams typically exclude aquatic pollen and fern spores from the total pollen count that makes up the 100% for each sample in the diagram. Concentration diagrams display the number of pollen grains or fern spores per cm^3 in order to show the amount of pollen and spores present in each sample. Pollen concentration can be lower for the uppermost sample of the water–sediment interface than samples from the consolidated sediments further down the core. When sufficient radiometric or historical dates are available for a core, pollen influx diagrams are created by adjusting the pollen concentration for the length of time identified for segments of the pollen record. Lead 210 is a useful radiometric method for recent diagrams since the isotope has a half-life of 22.7 years and near-zero radioactivity occurs in seven half-lives. Total lead is valuable for dating records in urban settings because the addition of tetraethyl lead to gasoline occurred from 1932 to 1973 (Nriagu 1978). The loss of a species from the forest, such as chestnut (*Castanea dentata*; plant nomenclature follows Gleason and Cronquist 1991) to the chestnut blight, is observed in a record by a drop in the pollen of the species. The drop from one sample to the next can be assigned a date based on what is known about the species loss in the area of the lake or bog. Finally, the uppermost sample is assigned the year before the record was sampled.

The goal of interpreting pollen diagrams for historical ecology research is to relate transitions in the pollen percentage, concentration or influx to environmental events, or human-caused changes. When several diagrams from a geographic area are available, comparisons are made for a regional synthesis. A note of caution in utilizing results from palynological research: palynologists interpret a pollen diagram based on a single pollen sample from each segment of the record without consideration of the variability that would be revealed by multiple pollen samples of a segment. In Holocene palynology, which includes the historical period, small pollen changes from one segment of a record to the next are often given great interpretative weight. Also, complex statistical analyses are based on the sequence of changes in the pollen counts of a record using the assumption that samples represent equal time periods within a span bounded by two identified dates. For recent (approximately a century) pollen records from the New Jersey–New York region, an assessment of variability in three adjacent 1 cm samples revealed a broad range for pollen percentages and concentrations (Loeb 1989a). Variability among pollen samples lends credence to the cautionary note of Traverse (2007) to not take “the very elegant equations and graphs presented in this area of palynology too literally.”

With Traverse’s words to the wise in mind, an interpretation of a recent pollen diagram from Lake Surprise, Watchung Reservation, Union County, New Jersey (40° 41’ 0” N, 74° 23’ 0” W; Loeb 1989a), is given to introduce how paleopalynological

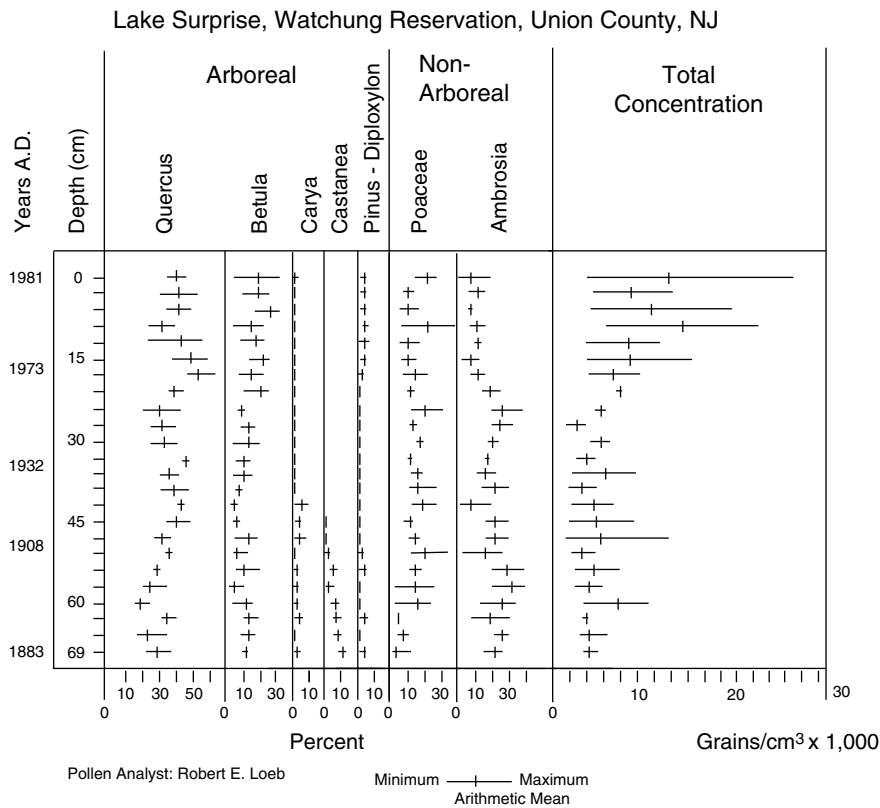


Fig. 2.1 Total pollen concentration and major taxa (greater than 3% mean for any group of three samples) percent pollen diagram for Lake Surprise, Watchung Reservation, Union County, New Jersey (Permission of the American Association of Stratigraphic Palynologists)

analysis applies to old growth urban forest research. Percentage data for the major (greater than 3% mean in any group of three samples) arboreal and non-arboreal genera as well as total concentration data are integrated into one diagram for Lake Surprise (Fig. 2.1). A 72 cm section of sandy clay was obtained from Lake Surprise in early January 1982 using a freezing corer. The top of the lake sediment section is designated as 1981 in the diagram. Sequences of three 1 cm samples were combined to obtain the mean and range with the uppermost segment being used to designate the interval on the diagram (i.e., 69–72 cm was designated as 69). Lead 210 dating provided the date 1883 at the bottom of the record. The chestnut pollen drop between segments 48–51 and 51–54 cm is related to the chestnut blight and dated as 1908 (Pennsylvania Chestnut Tree Blight Commission 1912), which is a revision from the prior designation as 1912 (Loeb 1989a). Addition of tetraethyl lead to gasoline in 1932 corresponds to the increase in total lead between segments 33–36 and

36–39 cm. Removal of tetraethyl lead from gasoline in 1973 is associated with the decrease in total lead between segments 15–18 and 18–21 cm.

In the New York–Philadelphia region, abandoned agricultural fields are first colonized by ragweed (*Ambrosia* spp.), then grass (*Poaceae* spp.), and later trees (Bard 1952). Farm losses to urban expansion is seen as an *Ambrosia* pollen rise in the 66–69 segment followed by a rise in *Poaceae* pollen in the 60–63 cm segment. However, lower percentages for both *Ambrosia* and *Poaceae* pollen begin in the 21–24 cm segment when the expectation is the ragweed decline will precede the grass decrease if rural forests were forming in the region. Instead of rural forests, structures and roads and construction sites were built but did not remain open long enough for ragweed or pollen-producing grass (not lawns) to become established. The *Castanea* pollen decline occurs between 48–51 and 51–54 cm. A *Carya* pollen increase can be observed in the three segments from 42–45 to 48–51 cm, but starting with 39–42 cm *Carya* pollen returns to the lower level before the peak. The canopy opened with the loss of chestnut to the blight and pollen from hickory trees, which do not disperse pollen far (McCarthy and Quinn 1990), was able to enter Lake Surprise without being screened out by chestnut trees. The hickory bark beetle (*Scolytus quadrispinosus* (Say) Solomon) outbreak on Long Island, New York, in 1915 (Anonymous 1916), spread across the region to decimate hickory which is represented by the decrease in *Carya* pollen from 42–45 to 0–3 cm. From segment 21–24 cm to the most recent segment, the higher means for total concentration (primarily arboreal pollen) and pollen percentages for *Betula*, *Pinus* – diploxylon, and *Quercus* represent arboricultural plantings in the growing suburbs of New York City (Solomon and Kroener 1971).

Historical Floras

To understand how the species composition of a regional forest changed after the forest resetting event requires a baseline for comparison, which is a tree species list dated soon after the time of the event. The first urban floras for Philadelphia (Barton 1818) and New York (Torrey 1819) were published 35 years after the end of the American Revolution. Both floras encompassed more than just the cities but the separation between the coverage areas is larger than 50 km. Later floras for the two cities are not appropriate to utilize for comparison because the areal overlap is greater than 50 km (Philadelphia – Keller and Brown 1905; New York – Taylor 1915). The comparison between the two floras is presented in Table 2.1 with nomenclature and designation as alien or native species following Gleason and Cronquist (1991). Because native species from the oak-chestnut forest region were cultivated and used for landscaping (Loeb 2010), Table 2.1 includes the date of first cultivation reported by Rehder (1940). The various terms used to describe the occurrence of each species were reduced to two words “common” and “uncommon” to enable analysis.

The tree species total for the two metropolitan areas was 98. New York had slightly more species (84) than Philadelphia (76). Only 54% of the species were

Table 2.1 Comparison of earliest arboreal floras after forest resetting event for New York, NY (Torrey 1819) and Philadelphia, Pa (Barton 1818)

Species	Cultivation date	New York 1819	Philadelphia 1818
<i>Acer negundo</i>	1688	C	C
<i>Acer pensylvanicum</i>	1755	C	C
<i>Acer rubrum</i>	1650	C	C
<i>Acer saccharum</i>	1753	C	
<i>Alnus rugosa</i>	1769	C	C
<i>Asimina triloba</i>	1736		C
<i>Betula lenta</i>	1759	C	C
<i>Betula lutea</i>	1800	U	
<i>Betula papyrifera</i>	1750	U	C
<i>Betula populifolia</i>	1750	C	C
<i>Carpinus caroliniana</i>	1812	C	C
<i>Carya cordiformis</i>	1650	C	C
<i>Carya glabra</i>	1750	C	C
<i>Carya laciniosa</i>	1800		C
<i>Carya ovata</i>	1629	C	C
<i>Carya tomentosa</i>	1766	C	C
<i>Castanea dentata</i>	1800	C	C
<i>Catalpa bignonioides</i>	1726	C	C
<i>Celtis occidentalis</i>	1636	C	C
<i>Cercis canadensis</i>	1641	C	
<i>Chamaecyparis thyoides</i>	1736	U	
<i>Cornus florida</i>	1730	C	C
<i>Crataegus crus-galli</i>	1691	C	C
<i>Crataegus monogyna</i> (A)	LC	C	C
<i>Crataegus pedicellata</i>	1902	C	U
<i>Crataegus phaenopyrum</i>	1738	C	
<i>Crataegus spathulata</i>	1806		U
<i>Diospyros virginiana</i>	1629		C
<i>Fagus grandifolia</i>	1800	C	C
<i>Fraxinus americana</i>	1724	C	C
<i>Fraxinus caroliniana</i>	1724	C	
<i>Fraxinus nigra</i>	1800	C	C
<i>Fraxinus pennsylvanica</i>	1783	C	C
<i>Gleditsia triacanthos</i>	1700	U	C
<i>Hamamelis virginiana</i>	1736	C	C
<i>Ilex opaca</i>	1744	U	U
<i>Juglans cinerea</i>	1633	C	C
<i>Juglans nigra</i>	1686	C	C
<i>Juniperus communis</i>	1560	U	
<i>Juniperus virginiana</i>	B 1664	C	
<i>Liquidambar styraciflua</i>	1681	C	U
<i>Liriodendron tulipifera</i>	1688	C	C
<i>Magnolia virginiana</i>	1688	C	C
<i>Malus coronaria</i>	1724	C	C
<i>Morus rubra</i>	1629	C	

(continued)

Table 2.1 (continued)

Species	Cultivation date	New York 1819	Philadelphia 1818
<i>Myrica cerifera</i>	1699	C	
<i>Nyssa aquatica</i>	B 1735	U	U
<i>Nyssa sylvatica</i>	B 1750	C	C
<i>Ostrya virginiana</i>	1690	C	
<i>Picea mariana</i>	1700	U	U
<i>Pinus rigida</i>	1713	C	C
<i>Pinus strobus</i>	B 1553	U	
<i>Pinus virginiana</i>	B 1739	C	C
<i>Platanus occidentalis</i>	1640	C	C
<i>Populus balsamifera</i>	B 1700	U	
<i>Populus deltoides</i>	B 1750	U	
<i>Populus grandidentata</i>	1772	U	U
<i>Populus heterophylla</i>	1765	U	
<i>Populus tremuloides</i>	1812	U	
<i>Prunus pensylvanica</i>	1773	C	C
<i>Prunus serotina</i>	1629	C	C
<i>Prunus virginiana</i>	1724	C	C
<i>Quercus alba</i>	1724	C	C
<i>Quercus bicolor</i>	1800		C
<i>Quercus coccinea</i>	1691		U
<i>Quercus falcata</i>	B 1763	C	C
<i>Quercus ilicifolia</i>	1800		U
<i>Quercus imbricaria</i>	1786		U
<i>Quercus marilandica</i>	B 1739		C
<i>Quercus muehlenbergii</i>	1822		C
<i>Quercus nigra</i>	1723		U
<i>Quercus palustris</i>	B 1770	C	C
<i>Quercus phellos</i>	1723	U	C
<i>Quercus prinus</i>	B 1800	C	C
<i>Quercus rubra</i>	1691	C	C
<i>Quercus stellata</i>	1819	C	C
<i>Quercus velutina</i>	1800	C	C
<i>Quercus x heterophylla</i>	1882		U
<i>Rhus copallina</i>	1688	C	
<i>Rhus glabra</i>	1863	C	C
<i>Rhus typhina</i>	1898	C	C
<i>Rhus vernix</i>	1713	C	C
<i>Robinia pseudoacacia</i>	1635	C	C
<i>Salix alba</i> (A)	LC	C	C
<i>Salix babylonica</i> (A)	1730	C	
<i>Salix discolor</i>	1809		C
<i>Salix elaeagnos</i> (A)	LC	C	
<i>Salix eriocephala</i>	1898	C	C
<i>Salix lucida</i>	1830	C	
<i>Salix nigra</i>	1809	C	
<i>Salix petiolaris</i>	1802		C

(continued)

Table 2.1 (continued)

Species	Cultivation date	New York 1819	Philadelphia 1818
<i>Salix viminalis</i> (A)	LC		C
<i>Sassafras albidum</i>	1633	C	C
<i>Thuja occidentalis</i>	1536	C	
<i>Tilia americana</i>	1752	C	C
<i>Tsuga canadensis</i>	1736	U	C
<i>Ulmus americana</i>	1752	C	C
<i>Ulmus rubra</i>	B 1830	U	U

Nomenclature and identification of a species as alien (A) follows Gleason and Cronquist (1991). Date of first cultivation is from Rehder (1940) with the code LC meaning long cultivated (at least prior to 1500) and the letter B preceding a date signifies cultivation began before the year indicated. Occurrence of the species is common (C) or uncommon (U)

reported in both floras which indicates the arboreal floras of the two metropolitan areas were quite different. Considering the five alien species, two were found in both floras and two were present only in the Philadelphia flora. Nearly half of the species (48) had the occurrence term common in both floras, while only five species were uncommon in the two floras. Among the species in only one flora, 21 were common and 15 were uncommon. Of the nine species with different assessments of occurrence, five had common for Philadelphia and uncommon for New York. Just nine species had a first cultivation date after 1818, indicating many of the native species that were reported to have grown spontaneously in the two metropolitan areas could have originated in arboricultural plantings at particular sites.

Witness Tree Records

Witness trees recorded in federal land survey documents from Canada and the United States have been used to determine forest composition for the time period just before the European settlement (Miller 1965). The trees are noted as boundary line or corner markers in the federal land survey records which do not include the New York and Philadelphia metropolitan areas. Pre-European settlement (in the locale of the survey) witness tree records in the oak-chestnut forest region (Collins 1956; Greller 1972; Russell 1979; Loeb 1987) may have limited value because of possible surveyor bias in the selection of trees and misidentification of tree species. To ascertain if witness tree records in the New York and Philadelphia metropolitan areas can be used to differentiate between forest regions, the county and township summaries with more than 100 witness trees (Loeb 1987) are compared for the oak-chestnut region and adjacent pine-oak region (Braun 1950). Witness tree records are available for two areas in the oak-chestnut region, southeast New York (4 groups of records) and north East New Jersey (14 groups of records), while only south East New Jersey (5 groups of records) exists to represent the oak-pine region. The two-tailed *T*-test of independent group means was used to determine if significant differences

Table 2.2 Major taxa (>2% of at least one survey group) means, standard deviations (St Dev), number of surveys (*N*), and significant results at 0.05 level for two-tailed T-test between means of witness tree surveys from 1665 to 1790 in counties and townships of southeast New York (SENY), north East New Jersey (NENJ), and south East New Jersey (SENJ; data from Loeb 1987)

	SENY	<i>N</i> =4	NENJ	<i>N</i> =14	SENJ	<i>N</i> =5
	Mean	St Dev	Mean	St Dev	Mean	St Dev
<i>Acer</i> spp.	1.3	1.3	3.6	2.3	5.0	4.1
<i>Betula</i> spp.	0.5	0.5	2.2	2.1	0.4	0.5
<i>Carya</i> spp.	0.8	1.3	5.8	5.3	1.6	2.2
<i>Castanea dentata</i>	10.8	7.3	6.6	3.5	2.4*	2.1
<i>Chamaecyparis</i> spp. and <i>Juniperus</i> spp.	0.3	0.4	0.1	0.3	6.2**	5.7
<i>Fraxinus</i> spp.	2.3	2.8	2.1	1.5	0.6	0.5
<i>Juglans</i> spp.	5.5	4.7	3.9	4.4	1.0	1.1
<i>Pinus</i> spp.	0.0	0.0	0.9	0.5	26.2**	16.6
<i>Quercus</i> spp.	67.0	9.0	63.6	12.6	47.0***	18.1
<i>Quercus rubra</i>	13.5****	5.9	7.7	4.2	6.4	2.9
<i>Quercus velutina</i>	17.3	5.4	14.6	3.2	9.0*	3.4
<i>Quercus alba</i>	33.3	5.4	39.6	9.6	29.0	14.5

*Mean for SENJ is significantly less (at 0.05 level) than means for SENY and NENJ

**Mean for SENJ is significantly greater (at 0.05 level) than means for SENY and NENJ

***Mean for SENJ is significantly less (at 0.05 level) than mean for NENJ

****Mean for SENY is significantly greater (at 0.05 level) than means for NENJ and SENJ

exist between the groups of records for the major (>2% of any group) tree taxa (surveyor use of common names limits association with species binomials). The *T*-tests calculations were performed using PASW Statistics (formerly SPSS Statistics) version 17 and the significance level selected was 0.05.

Examining the witness tree records (Table 2.2) for significant differences in the percentages of the codominants chestnut and pine (*Pinus* spp.) can serve to differentiate the two regions (oak-chestnut versus oak-pine). The means for chestnut in north East New Jersey and southeast New York are significantly greater than the mean for south East New Jersey. For pine, the south East New Jersey mean is significantly larger than the means for north East New Jersey and southeast New York. Cedar (*Chamaecyparis* spp. and *Juniperus* spp.) is the taxon in the witness tree records associated with the Atlantic white cedar (*Chamaecyparis thyoides*) swamp forests of the oak-pine region (Braun 1950). The mean for cedar in the south East New Jersey group is significantly greater than the means for north East New Jersey and southeast New York. For black oak (*Quercus velutina*), the means in both oak-chestnut region areas are significantly larger than the mean for the oak-pine region groups, which appears to be related to the preference of black oak for the moist-rich soils of the oak-chestnut region over the sandy dry soils typical of the oak-pine region (Hannah 1968). Northern red oak (*Quercus rubra*) had a significantly larger mean in the southeast New York survey group than both survey groups for East New Jersey. Perhaps Native Americans caused greater tree mortality in southeast New York than north and south East New Jersey because northern red

oak responds well to canopy openings (Graney 1987). In summary, the colonial period witness tree records showed significant differences among the major taxa that indicate the oak-chestnut and the oak-pine regions existed prior to European settlement (Table 2.2).

Forest Dynamics

Analysis of forest dynamics requires comparisons to determine the past, present, and future of the forests in a metropolitan area. The goal is to identify several sites with differing ecological histories for each old growth urban forest type. Although all sites should be considered and forest measurements begun where none existed before, old growth urban forests with previous forest research present the opportunity to compare the current forest with past conditions (Peterken and Backmeroff 1988). Two principles apply when prior studies are available for comparison to a remeasurement of the forest: methodological actualism and uniformitarianism. Methodological actualism refers to performing the forest remeasurement with the methods utilized in the prior study. Duplication of methods and relocation of sample sites is needed for comparisons of long-term changes because two types of forest variability exist. Henry Gleason's (1939) description of the two types is "In time-variability, the environment changes from one time to another on the same spot. In space-variability, the environment differs from spot to spot at the same moment." For example, in 1935, three different results were produced by using three different sampling methods in the St John's Pond Forest, Long Island, New York, and when resampling was conducted in 1982, the forest changes through time were also different (Loeb 1990). Methodological uniformitarianism refers to using the same methods at all sites being compared, which is difficult at best for historical ecology research considering the variety of forest measurement procedures used in the past. Even if methodological uniformitarianism cannot be achieved with field methods, comparisons of results must be made on the same type of statistics, such as stems per hectare or percentage of stems.

The ideal methodology is annual forest inventory but considering the lifespan of trees and the expense of conducting an inventory, continuous forest inventory is sufficient as long as the periodic measurements are performed as scheduled. Repeated measures statistical methods (Crowder and Hand 1990) can be applied to the data as well as analysis of changes in the forest distribution (Cho and Boerner 1991; Ward et al. 1996). The forest inventory records the location, size, and condition of every member of the canopy, subcanopy, and sapling classes within the entire area designated as the forest (McBride and Nowak 1989). Seedlings are counted in sections of the forests but not located until the individual stems join the sapling class. When a high-resolution global positioning system (GPS) and a large capacity geographic information system (GIS) are available, precise measurements of tree dimensions and position can be recorded, which permits concerns about forest variability introducing uncertainty into measurements of changes through time to be safely set

aside. More relevant to historical forest ecology research is how recently and infrequently GPS and GIS technology has been used. Past forest records do not have precise location information or the capacity to correct field investigator errors. For example, the New York City Department of Parks and Recreation has a park map series created in the 1930s which gives the location, species identification, and diameter measure for all of the trees. A remapping of Seton Falls Park, Bronx, New York, using the 1936 map revealed inaccurate locations, misidentifications to the family level, and diameter measures indicating incredible rates of tree growth and shrinking trees (Loeb 1982). Forest dynamics research employing historical forest maps must account for errors, and comparative analysis is limited to reliable information (Stalter and Kincaid 2008).

Historical records of urban forest composition rarely involve maps locating trees. Instead the common record used for comparison in resampling research has an unclear description of method and imprecise location of sampling points. Often there is nothing more than a total for each species and a location associated with the information which could be an ill-defined tract for remnant and landscaped forests or entire cities for street forests. The following historical forest ecology research studies and literature reviews for the three old growth urban forest types show forest dynamics based on resampling; however, true methodological actualism and uniformitarianism was not possible because of the methods' descriptions, or lack thereof, in the prior research reports.

Street Forest

Original Research: East Orange, Haddonfield, and Moorestown, New Jersey

Solotaroff (1912) described the state of the art for urban shade-trees planting and maintenance in the New York and Philadelphia metropolitan areas at the beginning of the twentieth century. Also, Solotaroff provided a summary of the East Orange, New Jersey, street forest species (Table 2.3) and reported the dbh (diameter at breast height, 130 cm) distribution as 1,698 trees under 15.2 cm dbh, 7,036 trees from 15.2 to 45.7 cm dbh, and 2,219 trees greater than dbh 45.7 cm. In 2004, a survey of the entire street forest was performed but no dbh data were recorded (Anonymous 2006). From 1911 to 2004, the total number of stems dropped by 40%. The number of species present in 1911 but not reported in 2004 was 24, and 10 species were noted in 2004 but not in 1911. The ratio of alien to native taxa in 1911 of 17:33 and 10:19 in 2004 demonstrates a decline for both alien and native species diversity. Red maple (*Acer rubrum*), silver-maple (*Acer saccharinum*), and sugar maple (*Acer saccharum*) went from comprising over two-thirds of the forest in 1911 to less than one-tenth in 2004. The stems per hectare (sph) for Norway maple (*Acer platanoides*) declined slightly from 1911 to 2004 but rose from 11% to 17% of the forest. Lindens (*Tilia* spp.), plane trees (*Platanus* spp.), and pin oak (*Quercus palustris*) joined Norway maple to comprise the four most frequent taxa

Table 2.3 Street tree taxa stems per hectare for East Orange, New Jersey in 1911 (Solotaroff 1912) and 2004 (Anonymous 2006) as well as Haddonfield, New Jersey and Moorestown, New Jersey in 2010

	First date cultivated	East Orange		Haddonfield			Moorestown		
		Year of Inventory		Sapling	Subcanopy	Canopy	Sapling	Subcanopy	Canopy
		1911	2004						
1688	<i>Acer negundo</i>	0.013	0	0.014	0.030	0.014	0	0.006	0.001
LC	<i>Acer platanoides</i> (A)	1.185	1.125	0.088	0.508	0.230	0.002	0.066	0.019
LC	<i>Acer pseudoplatanus</i> (A)	0.049	0.014	0	0.010	0.007	0	0	0
1650	<i>Acer rubrum</i>	2.790	0.485	0.127	0.647	0.29	0.033	0.250	0.050
1725	<i>Acer saccharinum</i>	2.183	0.044	0.003	0.015	0.093	0	0.013	0.028
1753	<i>Acer saccharum</i>	2.187	0.068	0.026	0.448	0.655	0.003	0.057	0.058
1809	<i>Aesculus glabra</i>	0.007	0	0.004	0	0	0.003	0.009	0
1576	<i>Aesculus hippocastanum</i> (A)	0.173	0	0.001	0.003	0.007	0.002	0.005	0.001
1784	<i>Ailanthus altissima</i> (A)	0.006	0.001	0.001	0.004	0.005	0	0	0
1736	<i>Betula nigra</i>	0	0.008	0.005	0.004	0.001	0	0.001	0
1812	<i>Carpinus caroliniana</i>	0.004	0	0	0.004	0	0.001	0.023	0.001
	<i>Carya</i> spp.	–	0.023	0	0	0	0	0.003	0.002
1650	<i>Carya cordiformis</i>	0	–	0	0.026	0.008	0	0.001	0
1750	<i>Carya glabra</i>	0.006	–	0	0	0.001	0	0	0
1629	<i>Carya ovata</i>	0.002	–	0	0	0.001	0	0	0
1766	<i>Carya tomentosa</i>	0.002	–	0	0	0	0	0	0
1854	<i>Castanea mollissima</i> (A)	0	0.033	0	0	0	0	0	0
	<i>Catalpa</i> spp.	0.028	0	0.001	0.003	0.011	0	0	0
1730	<i>Cornus florida</i>	0.003	0	0.034	0.148	0	0	0.001	0
1800	<i>Fagus grandifolia</i>	0.006	0.001	0.003	0.005	0.008	0	0.009	0.01
1724	<i>Fraxinus americana</i>	0.078	0.159	0.03	0.249	0.229	0.003	0.022	0.004
1784	<i>Ginkgo biloba</i> (A)	0	0.240	0.003	0.004	0.008	0	0.015	0.001
1700	<i>Gleditsia triacanthos</i>	0.001	0.175	0.029	0.066	0.047	0.007	0.032	0.001
1756	<i>Halesia tetraptera</i>	0.001	0	0	0	0	0	0.002	0

(continued)

Table 2.3 (continued)

	First date cultivated	East Orange				Haddonfield				Moorestown			
		Year of Inventory		2004		Sapling		Subcanopy		Sapling		Subcanopy	
		1911											Canopy
<i>Juglans cinerea</i>	1633	0.001		0		0.001	0.011	0.022	0	0	0	0	0
<i>Juglans nigra</i>	1686	0.002		0		0	0.001	0.012	0	0	0.002	0.002	0.002
<i>Liquidambar styraciflua</i>	1681	0.005		0.020		0.015	0.048	0.386	0.001	0.001	0.020	0.011	0.011
<i>Liriodendron tulipifera</i>	1688	0.009		0.007		0.022	0.149	0.140	0	0	0.005	0.004	0.004
<i>Morus alba</i> (A)	LC	0.001		0		0	0	0	0	0	0	0	0
<i>Nyssa sylvatica</i>	B 1750	0.001		0		0.003	0.01	0.005	0	0	0.001	0.001	0.001
<i>Picea abies</i> (A)	LC	0.002		0		0.001	0.045	0.062	0	0	0.002	0.001	0.001
<i>Pinus strobus</i>	B 1553	0.005		0		0.004	0.089	0.071	0.001	0.001	0.031	0.001	0.001
<i>Platanus</i> spp.	–	–		0.802		0	0	0	0	0	0	0	0
<i>Platanus occidentalis</i>	1640	0.022		–		0.027	0.01	0.455	0.004	0.004	0.020	0.041	0.041
<i>Platanus orientalis</i> (A)	1842	0.006		–		0	0	0	0	0	0	0	0
<i>Platanus x acerifolia</i> (A)	B 1700	0		–		0.011	0.007	0.004	0.005	0.005	0.027	0.080	0.080
<i>Populus alba</i> (A)	LC	0.007		0		0	0	0	0	0	0	0	0
<i>Populus x canadensis</i> (A)	1818	0.713		0		0	0	0	0	0	0	0	0
<i>Populus grandidentata</i>	1772	0.001		0		0	0.003	0.001	0	0	0	0	0
<i>Populus nigra</i> (A)	LC	0.002		0		0	0	0	0	0	0	0	0
<i>Prunus cerasus</i> (A)	LC	0.002		0		0	0	0	0	0	0	0	0
<i>Prunus serotina</i>	1629	0.004		0		0.018	0.152	0.048	0.002	0.002	0.008	0.003	0.003
<i>Prunus serrulata</i> (A)	1900	0		0.063		0.010	0.138	0.086	0	0	0.002	0	0
<i>Pyrus calleryana</i> (A)	1908	0		0.179		0.015	0.168	0.068	0.001	0.001	0.013	0.005	0.005
<i>Pyrus malus</i> (A)	LC	0.001		0		0	0	0	0	0	0	0	0
<i>Quercus alba</i>	1724	0.019		0.015		0.001	0.003	0.015	0.001	0.001	0.003	0.002	0.002
<i>Quercus bicolor</i>	1800	0.011		0		0.037	0.033	0	0	0	0	0	0
<i>Quercus macrocarpa</i>	1811	0		0.017		0	0.014	0.005	0	0	0.001	0.001	0.001
<i>Quercus palustris</i>	B 1770	0.022		1.339		0.003	0.145	1.297	0.003	0.003	0.047	0.116	0.116
<i>Quercus rubra</i>	1691	0.011		0.312		0.012	0.262	0.733	0.001	0.001	0.008	0.027	0.027

<i>Quercus velutina</i>	1800	0	0.082	0.003	0.021	0.027	0	0.003	0.004
<i>Salix babylonica</i> (A)	1730	0.004	0	0	0	0	0	0	0
<i>Sassafras albidum</i>	1633	0.002	0	0.003	0.007	0.003	0.001	0.002	0.001
<i>Sophora japonica</i> (A)	1747	0	0.126	0.001	0.011	0.012	0	0.001	0
<i>Sorbus americana</i>	1811	0	0.007	0	0	0	0	0	0
<i>Thuja occidentalis</i>	1536	0.001	0	0.007	0.068	0.011	0.002	0.008	0
<i>Tilia</i> spp.									
<i>Tilia americana</i>	1752	0.130	–	1.059	0	0	0.001	0	0
<i>Tilia cordata</i> (A)	LC	0.045	–	0.003	0.034	0.071	0.007	0.016	0.002
<i>Tilia tomentosa</i> (A)	1767	0	–	0.059	0.181	0.118	0.002	0.047	0.037
<i>Ulmus</i> spp.									
<i>Ulmus americana</i>	1752	–	0.017	0.003	0.064	0.008	0	0	0
<i>Ulmus parvifolia</i> (A)	1794	0	–	0.007	0.086	0.059	0	0.005	0.001
<i>Ulmus procera</i> (A)	B 1770	0	–	0.001	0.015	0.001	0	0.001	0.001
<i>Ulmus pumila</i> (A)	1860	0	–	0	0	0.001	0	0	0
<i>Ulmus rubra</i>	B 1830	0.002	–	0	0.015	0.023	0	0	0
<i>Zelkova serrata</i> (A)	1862	0	0.057	0	0.004	0	0	0.001	0
Other species		0	0	0.047	0.188	0.16	0.006	0	0.001
Total Stems Per Hectare		10,727	6,475	1,074	5,268	6,096	0.341	1,250	0.692
Total Trees		10,953	6,605	784	3,849	4,450	784	2,873	1,591

Native species nomenclature follows Gleason and Cronquist (1991) and alien species nomenclature follows Rehder (1940). The letter A in parenthesis after the species name indicates an alien species. Date of first cultivation is from Rehder (1940) with the code LC meaning long cultivated (at least prior to 1500) and the letter B preceding a date signifies cultivation began before the year indicated. The Haddonfield, New Jersey, and Moorestown, New Jersey, data are in the dbh size classes: sapling (0 cm < dbh < 7.5 cm), subcanopy (7.5 cm ≤ dbh ≤ 45.7 cm), and canopy (dbh > 45.7 cm)

in 2004. All of the species reported for East Orange, New Jersey, in either 1911 or 2004 entered cultivation before 1908. Similar changes in species composition were observed in the New York City street forest; however, plantings and removals from 1900 to 1990 kept the size of the forest nearly stable throughout the 90-year period (Loeb 1992a).

Tree survey records were obtained for the street old growth urban forests in Haddonfield, New Jersey, and Moorestown, New Jersey (60% completed in 2010). The survey data were converted into stems per hectare and divided into three dbh size classes: sapling ($0\text{ cm} < \text{dbh} < 7.5\text{ cm}$), subcanopy ($7.5\text{ cm} \leq \text{dbh} \leq 45.7\text{ cm}$), and canopy ($\text{dbh} > 45.7\text{ cm}$). To enable comparison with East Orange, the data for the two towns were incorporated into Table 2.3 but no major taxon ($>5\%$ of the forest) from the Haddonfield and Moorestown surveys was excluded. The total for taxa in Haddonfield was 144 as compared to 131 for Moorestown and the two towns had 91 species in common. Despite the great species diversity in Haddonfield and Moorestown, eight species from 1911 and two species from 2004 were present in East Orange but not Haddonfield and Moorestown. The species with $\text{sph} > 0.25$ for Haddonfield were white ash (*Fraxinus americana*), callery pear (*Pyrus calleryana*), sycamore (*Platanus occidentalis*), Norway maple, red maple, sugar maple, pin oak, and northern red oak. For Moorestown, only red maple had an $\text{sph} > 0.25$.

Two characteristics are useful for ascertaining if a street forest is an old growth urban forest: for all species the largest size class ($\text{dbh} > 45.7\text{ cm}$) has the highest sph and the species with the greatest sph match the species in historical forests (East Orange in 1911) or historical floras (Table 2.1). The largest size class is nearly half of the Haddonfield forest but less than a third of the Moorestown forest. For East Orange, the species in 1911 with $\text{sph} > 0.25$ were Norway maple, red maple, silver-maple, sugar maple, Carolina poplar (*Populus x canadensis*), and American elm (*Ulmus americana*), while the species in 2004 with $\text{sph} > 0.25$ were Norway maple, red maple, plane trees, pin oak, northern red oak, and lindens. The Haddonfield species with $\text{sph} > 0.25$ match all of the taxa having an $\text{sph} > 0.25$ from East Orange in 1911 except silver-maple, American elm, and Carolina poplar, and in 2004 except lindens. Haddonfield fits the characteristics of a street old growth urban forest. In contrast Moorestown does not because of the low sph in the largest size class and only red maple having an $\text{sph} > 0.25$, which is a common element to both the 1911 and 2004 species with $\text{sph} > 0.25$ in East Orange. The problem with making a judgment concerning Moorestown is the lack of historical perspective on changes in the forest. For example, the populations of pin oak and northern red oak have been decimated by oak wilt (*Xyella fastidiosa*) in Moorestown (but not Haddonfield), and during the past 5 years, 20 different species have been planted and none are species with $\text{sph} > 0.25$ for East Orange (Gibson et al. 2010). Diversifying a street forest is beneficial to lessen the chance of major losses to disease (e.g., Dutch elm disease), but planting trees to maintain the population of viable old growth native species is the essential planning goal to achieve historical continuity.

Landscaped Forest

Original Research: Beechwood, Country Club, and Robert's Hollow Forests, Fairmount Park, Philadelphia, PA

Fairmount Park is located north of center city Philadelphia, Pennsylvania (40° 0' N; 75° 12' W), and the Schuylkill River divides the Park into two sections: East and West Fairmount Park. Three landscaped old growth urban forests are present in the northern portion of West Fairmount Park: Beechwood, Country Club, and Robert's Hollow. An 1868 map (Fairmount Park Commission 1868) depicts the forests as does a 1900 map (Fairmount Park Commission 1900), which also includes the former park trolley system (Fig. 2.2). Research for the historical ecology of Fairmount Park was conducted in the archives of the Fairmount Park Commission, City of Philadelphia, and Dickinson College and the libraries of the Academy of Natural Sciences of Philadelphia, Historical Society of Pennsylvania, New York Botanical Gardens, and Pennsylvania Horticultural Society. Because the historical record is exceptionally rich for old growth urban forests, the information will be divided in two sections: historical ecology and forest dynamics.

Historical Ecology

Before 1770, estates were built and landscaped in the land that would become Fairmount Park, and during the American Revolution the land was virtually cleared of trees. The estates were replanted with native species trees (not just saplings and seedlings) prior to 1800 (Loeb 2010) and were incorporated into the Park before 1868 (Anonymous 1868; Fig. 2.3). Although no records of the species planted in the estates are available, the trees commonly planted in the region prior to 1800 (Adams 2004) were as follows: balsam fir (*Abies balsamea*), red maple, sugar maple, red buckeye (*Aesculus pavia*), sweet birch (*Betula lenta*), southern catalpa (*Catalpa bignonioides*), flowering dogwood (*Cornus florida*), eastern red cedar (*Juniperus virginiana*), tulip-tree (*Liriodendron tulipifera*), black spruce (*Picea mariana*), white pine (*Pinus strobus*), white oak (*Quercus alba*), scarlet oak (*Quercus coccinea*), black oak, black locust (*Robinia pseudoacacia*), sassafras (*Sassafras albidum*), and eastern hemlock (*Tsuga canadensis*). The estates should be expected to have had greater native species diversity than others in the oak-chestnut region because Bartram's Garden, the historical origin of native species stock for arboricultural plantings in the United States (Fry 1996), was nearby.

Landscaping in East Fairmount Park from 1860 to 1868 was focused on planting groups of native species into the existing forest of native species (Sidney and Adams 1859). In planning for further landscaping, a survey of the trees revealed 31,421 trees with dbh of 45.7 cm or greater. The total of smaller dbh trees was estimated to be more than 90,000 but no enumeration by species was done (Cresson 1868). Plantings in the 1876 Centennial Fairgrounds focused on alien angiosperm



Fig. 2.2 Topographical map of Fairmount Park, Philadelphia, excepting Wissahickon Valley, 1900 (Fairmount Park Commission 1900; permission of the Fairmount Park Historic Resource Archive; Collection of Adam Levine, www.phillyh2o.org)

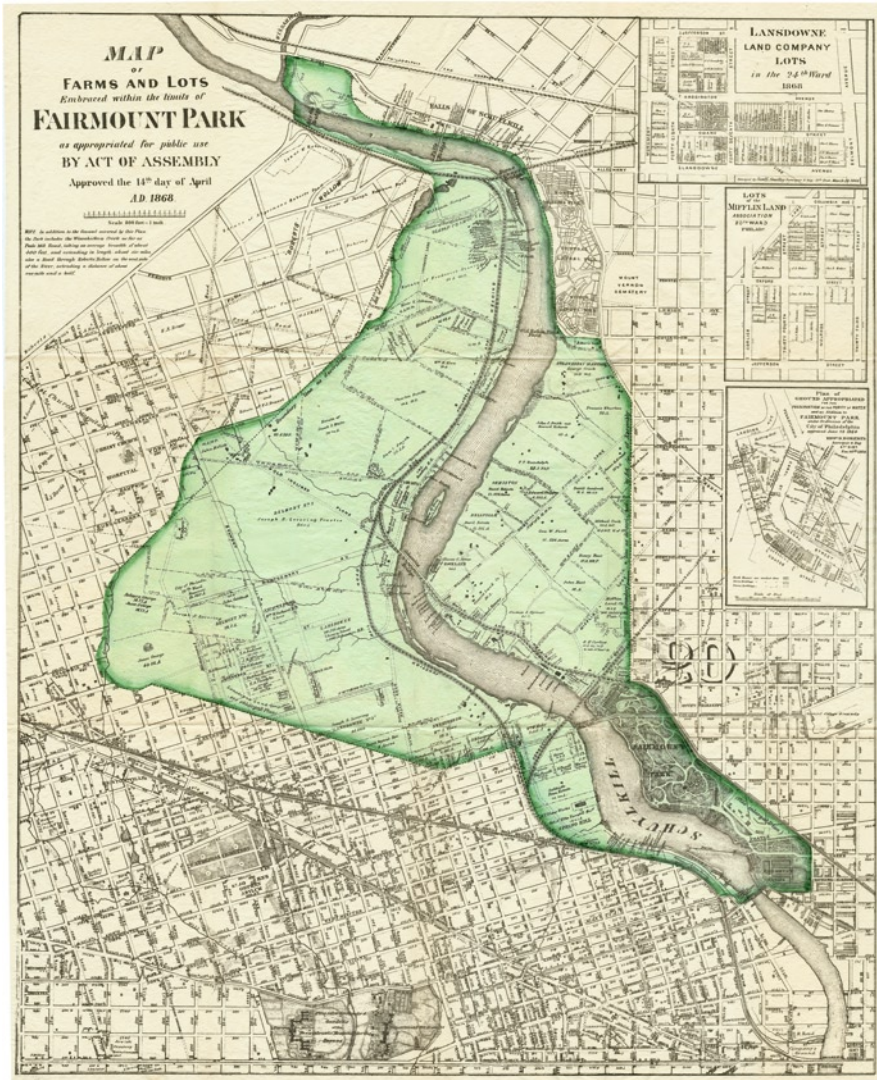


Fig. 2.3 Map of farms and lots embraced within the limits of Fairmount Park (Anonymous 1868, with permission of Collection of Adam Levine, www.phillyh2o.org)

and gymnosperm species which enabled Fairmount Park to have more tree and shrub species than Central Park, New York City, in 1880. By 1970, Fairmount Park lost more than half of the species present in 1880 while Central Park had more species in 1970 than 1873. From 1970 to 2007, 60 tree and shrub species were lost from the Centennial Fairgrounds (Loeb 2010).

The extensive arboricultural plantings made in preparation for the 1876 Centennial Fair (Rothrock 1880) and later (Corson 1937) did not affect the three landscaped old

growth urban forests because of their location away from the Centennial Fairgrounds (Fig. 2.2). Fires started by sparks from railroad engines began with the completion of the Reading Railroad tracks between the west shore of the Schuylkill River and West Fairmount Park on December 9, 1839 (Holton 1989), and ceased with the complete replacement of steam locomotives with diesel locomotives by 1955 (Marx 1976). Sweetbrier (*Rosa eglanteria*) and catbrier (*Smilax rotundifolia*) were so common in the Park that the grass cutting contract for 1895 incorporated specifications for cutting briars (Fairmount Park Commission 1895). Pedestrian access to the Beechwood and Country Club forests was increased by the Fairmount Park Trolley (1897–1946; Cox 1970) having stops at both sites (Fig. 2.2 – note the Country Club forest is at the Brünnenwald trolley stop) but the location of Robert’s Hollow in the most northern point of the Park was not a trolley stop. The Fairmount Park Stable is located next to Country Club and horseback riders continue to be permitted to use the trails in the forest.

In 1908, Oglesby Paul, landscape gardener of Fairmount Park, reported on the condition of the trees in the Park including the Beechwood, Country Club, and Robert’s Hollow forests (Paul 1908). Insect infestations caused deaths for Norway maple, silver-maple, southern catalpa, beech (*Fagus* spp.), walnut (*Juglans* spp.), black gum (*Nyssa sylvatica*), oriental plane-tree (*Platanus orientalis*), white oak, pin-oak, rock chestnut-oak (*Quercus prinus*), black willow (*Salix nigra*), linden, and American elm. In Beechwood, the chestnut blight caused a few trees to be “removed in the year 1907, since when many more have died.” The Beechwood forest was reported to be in poor condition because fires burned the humus layer and killed seedlings and saplings; the public trampled the seedlings and saplings; and sprout growth was short lived. Fires started by railroad engines in the winter of 1907 burned half of Beechwood and the entire area of Robert’s Hollow. Prior to the fire, the Robert’s Hollow forest had a thick layer of humus. Paul wrote “At the Country Club alone do we find a perfectly healthy forest, with an adequate proportion of forest cover, veterans, and saplings to insure its future existence and character.” Furthermore, Paul noted “One of the chief reasons for the excellent condition of the Country Club woods is the thorough forestry work we did there three years ago. Numbers of windfalls and dead and dying trees were cut out at that time, thus affording a chance for the surrounding trees to seed into open spaces, and also admitting sunlight and air to the young saplings already struggling for a foothold.” Although not reported as carried out, Paul recommended removing competitors of oak (*Quercus* spp.) and walnut in thickets of seedlings.

The purpose of Paul’s report was an appeal for an increase in the funding for arboricultural work in Fairmount Park. In 1907, \$3,500 was allotted for tree work in Fairmount Park as compared to \$50,000 for Central Park, New York, which was a quarter of the size of the area under Fairmount Park Commission control (Paul 1908). Over the decades of the twentieth century, the Fairmount Park Commission became responsible for the street forest and the expanding park system of Philadelphia, and as the years passed, relatively less attention was paid to Fairmount Park. The Fairmount Park Commission funded an ecological study of the Park in 1969 which recommended protection of the Beechwood, Country Club, and Robert’s

Hollow forests (McCormick 1971). White-tailed deer (*Odocoileus virginianus*; mammal nomenclature follows Kays and Wilson 2009) returned to the area of Fairmont Park in the 1980s and the Commission initiated a culling program in 1999 (Brown 2005).

As noted above, the 1868 survey included all of the trees with dbh greater than 45.7 cm and was for the entire Fairmont Park (Cresson 1868), but at the time Robert’s Hollow was not included in the Park. Based upon the forest inventory, chestnut was the species with the largest stems per hectare (sph) in 1868 at 21.5 for trees with dbh > 45.7 cm in Fairmont Park. Sugar maple and silver-maple, eastern red cedar, black mulberry (*Morus nigra*), and white oak had an sph greater than 13. Of the 17 remaining taxa with an sph greater than two, only bitternut-hickory (*Carya cordiformis*), southern catalpa, and tulip-tree had an sph greater than 6.5 (Fig. 2.4).

Paul’s report (1908) had 1907 survey results for 18 forests including Beechwood, Country Club, and Robert’s Hollow. The survey methods description was incomplete and the location of sampling sites was not given (Paul 1908), which precluded duplication of methods. The remeasurement method selected was a total survey of the largest minimum area located at least 10 m from the edges of the forest. Since the entire area of Robert’s Hollow (1.9 ha) was surveyed in 1907, a 60 m by 300 m plot (1.8 ha) was placed within the old growth urban forest of Robert’s Hollow in 2007. In the larger forests at Beechwood and Country Club (in 1907, 10 ha and

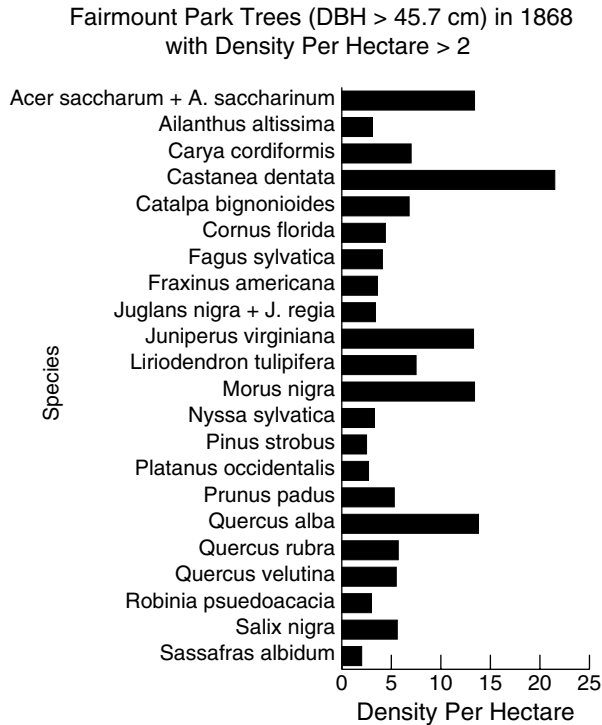


Fig. 2.4 Fairmont Park trees (dbh>45.7 cm) in 1868 (Cresson 1868) with stems per hectare greater than two

12 ha, respectively), a 60 m by 300 m plot was placed within the remaining area (losses due to buildings and roadway construction) for consistency in survey method with Robert's Hollow. The 1907 data were published in diameter size classes (Paul 1908). In order to permit comparison with the 1868 data and to examine forest dynamics, three size classes were selected to report: canopy trees ($\text{dbh} > 45.7 \text{ cm}$), subcanopy trees ($45.7 \text{ cm} \geq \text{dbh} \geq 10.2 \text{ cm}$), and saplings ($10.2 \text{ cm} > \text{dbh} > 2.5 \text{ cm}$). The data from each time period were converted to stems per hectare.

In 1907, chestnut, northern red oak, and black oak in the canopy, and choke-cherry (*Prunus virginiana*) in the subcanopy (Fig. 2.5a–d, respectively) had their highest sph in Beechwood, a lower sph in Country Club, and the lowest sph in Robert's Hollow. Each of the four species had very few or no saplings in 1907. By 2007, chestnut and choke-cherry were absent while northern red oak and black oak had more than half of their populations in the largest size class. In 2007, black oak sph followed the pattern among the forests from 1907. In contrast, the pattern for 1907 and 2007 differed for northern red oak with the total sph for each forest being almost equal, and saplings were present in all three forests in 2007.

Again in 1907, American beech, tulip-tree, and white oak in the canopy, and flowering dogwood in the subcanopy (Fig. 2.6a–d, respectively) had their highest sph in Country Club. Among the four species, only white oak had a lower sph in Beechwood than Robert's Hollow and was distinctive as being the only species with no saplings in all three forests. Flowering dogwood virtually disappeared from all three forests by 2007. American beech decreased in each size class in all three forests except for saplings in Country Club and Robert's Hollow. Tulip-tree's total sph was much closer to equal among the forests in 2007 than 1907, and the population of canopy trees expanded in all three forests by 2007. The number of subcanopy trees increased for white oak in 2007 relative to 1907, and saplings were present only in Country Club and Robert's Hollow in 2007.

The species with their largest sph in Robert's Hollow were rock chestnut-oak, bitternut-hickory, white ash, and Norway maple (Fig. 2.7a–d, respectively). Rock chestnut-oak was uniquely found in Robert's Hollow in 1907 with an sph of 40, and by 2007 was present in all three forests but had an sph of less than one. Bitternut-hickory did not have saplings in 1907 but two-thirds of the population was saplings in 2007. In Robert's Hollow, white ash had only canopy and subcanopy trees in both 1907 and 2007; however, the total sph was nearly 60 times greater in 2007 than 1907. Although Norway maple was found in all three forests just in 2007, Robert's Hollow was the only forest with individuals in all three size classes.

Four species, red maple, black gum, wild black cherry, and sassafras (Fig. 2.8a–d, respectively), changed from being absent in at least one forest in 1907 to being found in all three in 2007. In addition, all four species had a larger sph in 2007 than 1907 for all three size classes in each forest, except for the greater sph for sassafras subcanopy trees in Country Club and wild black cherry had a higher sapling sph for Robert's Hollow. Hornbeam (*Carpinus caroliniana*) was found only in Country Club and had a threefold increase in subcanopy and canopy tree sph to approximately 30 but no saplings were found in 1907 or 2007. For the species with a total sph greater than 10 that were first recorded in 2007, Norway maple, Hercules club (*Aralia*

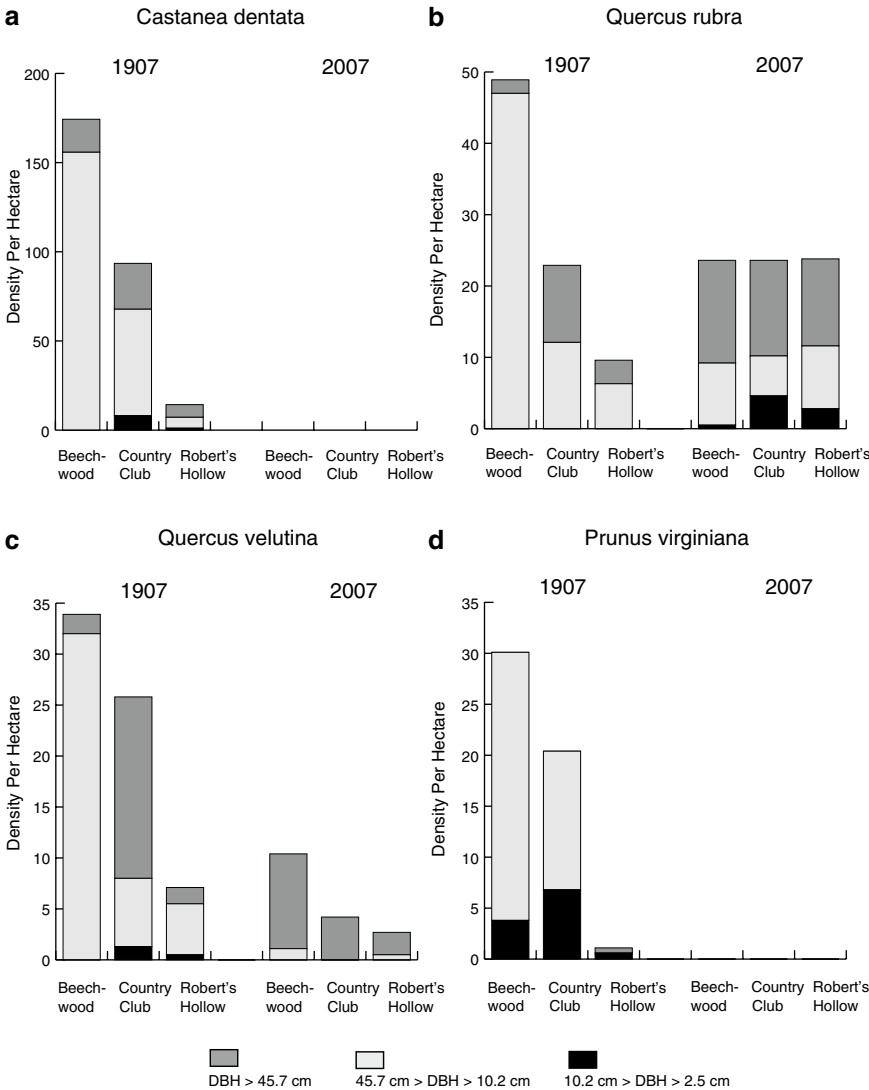


Fig. 2.5 Stems per hectare for 1907 (Paul 1908) and 2007 in size classes $dbh > 45.7$ cm, $45.7 \geq dbh \geq 10.2$ cm, and $10.2 \text{ cm} > dbh > 2.5$ cm for (a) *Castanea dentata*, (b) *Quercus rubra*, (c) *Quercus velutina*, and (d) *Prunus virginiana*

spinosa), sweet birch, and staghorn-sumac (*Rhus typhina*) were in all three forests; shagbark-hickory (*Carya ovata*) and sweet gum (*Liquidambar styraciflua*) were only in Beechwood and Country Club; and umbrella-tree (*Magnolia tripetala*) was just in Beechwood.

Among the species with an sph greater than two in 1868 (Fig. 2.4) and not reported above, sugar maple was not present in 1907 but in 2007 was found only in

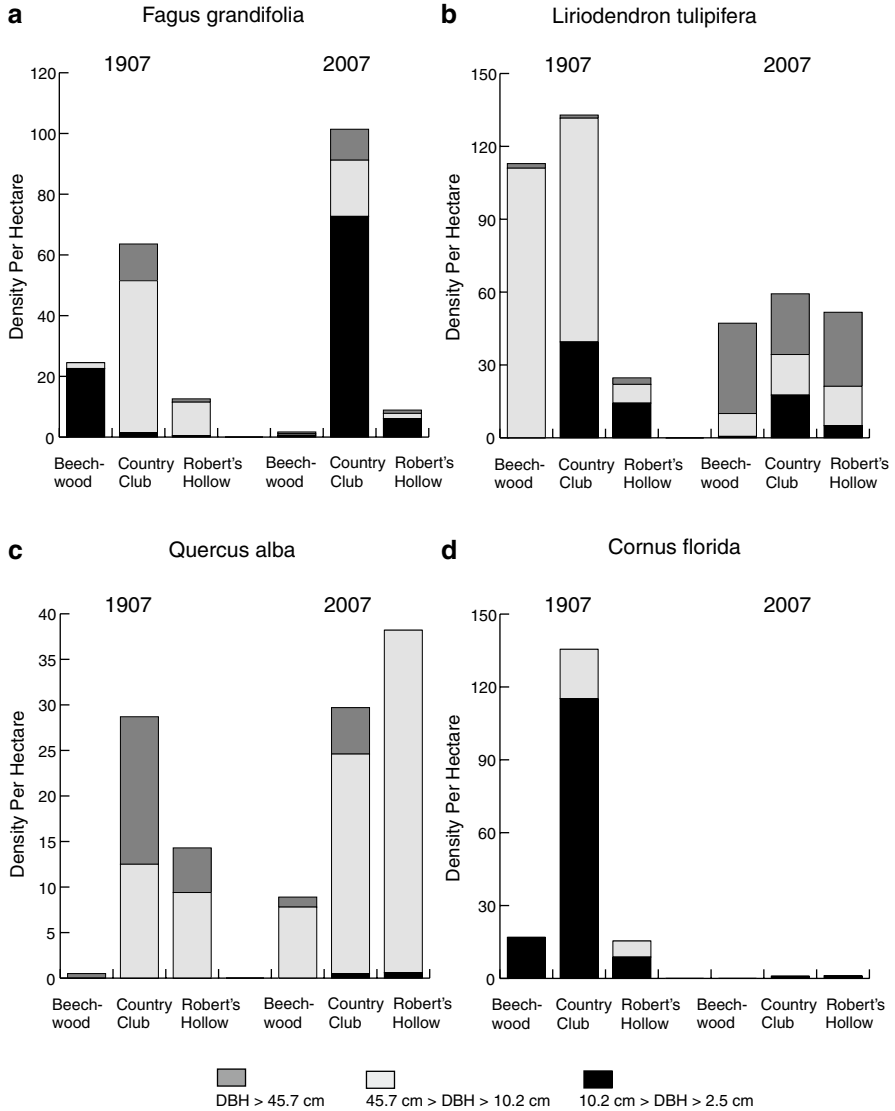


Fig. 2.6 Stems per hectare for 1907 (Paul 1908) and 2007 in size classes dbh > 45.7 cm, 45.7 ≥ dbh ≥ 10.2 cm, and 10.2 cm > dbh > 2.5 cm for (a) *Fagus grandifolia*, (b) *Liriodendron tulipifera*, (c) *Quercus alba*, and (d) *Cornus florida*

Beechwood and Country Club with an sph less than three. Tree of heaven (*Ailanthus altissima*) was reported in Country Club and Robert's Hollow in 1907 but remained only in Country Club in 2007 with an sph under three. Southern catalpa had an sph less than one and was present only in Robert's Hollow during 2007. Black walnut

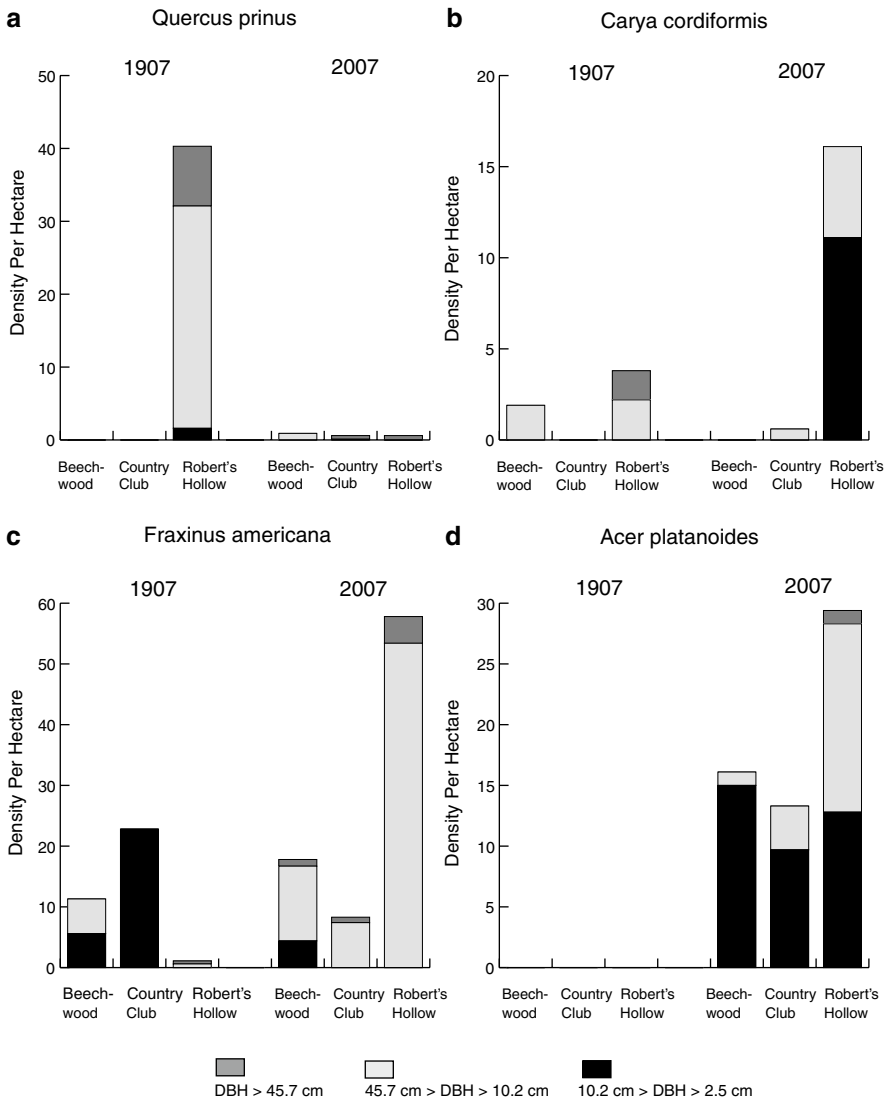


Fig. 2.7 Stems per hectare for 1907 (Paul 1908) and 2007 in size classes dbh>45.7 cm, 45.7≥dbh≥10.2 cm, and 10.2 cm>dbh>2.5 cm for (a) *Quercus prinus*, (b) *Carya cordiformis*, (c) *Fraxinus americana*, and (d) *Acer platanoides*

(*Juglans nigra*) was found in both 1907 and 2007 in Country Club and Robert's Hollow; however, the total sph did not exceed four. Sycamore had an sph less than two and was located just in Robert's Hollow during 1907. Three specimens of white pine were reported only in the sampling notes for Country Club in 1907 and were not present in the area of the forest during the 2007 field work. Black locust was first

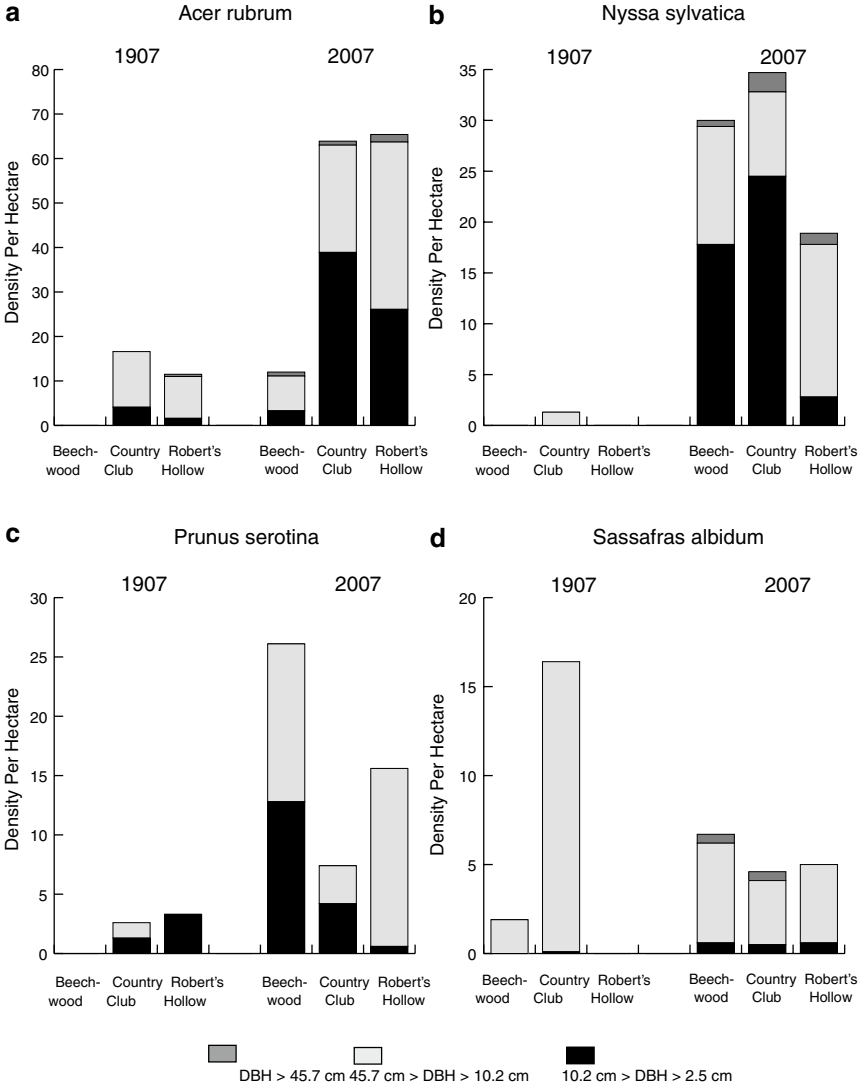


Fig. 2.8 Stems per hectare for 1907 (Paul 1908) and 2007 in size classes dbh>45.7 cm, 45.7≥dbh>10.2 cm, and 10.2 cm>dbh>2.5 cm for (a) *Acer rubrum*, (b) *Nyssa sylvatica*, (c) *Prunus serotina*, and (d) *Sassafras albidum*

recorded in Beechwood and Robert's Hollow in 2007 with an sph less than one. Silver-maple, eastern red cedar, English walnut (*Juglans regia*), black mulberry, European bird-cherry (*Prunus padus*), and black willow were not found in the three forests in 1907 or 2007.

Forest Dynamics

Plantings of native trees in the estates that became Fairmount Park formed a forest dominated by native species in 1868 (Fig. 2.4) which reflected the pre-European settlement forest composition as indicated by witness tree records (Loeb 1987). In contrast, balsam fir, black spruce, eastern hemlock, red buckeye, and scarlet oak were commonly planted before 1800 (Adams 2004) but were not represented in Fairmount Park by specimens with dbh > 45.7 cm (Cresson 1868). Paul (1908) pointed out that air pollution caused the loss of some gymnosperm species, which may explain the absence of balsam fir, black spruce, and eastern hemlock in 1868 (the field surveyor tally sheet used in 1868 included eastern hemlock but none were reported). All of the species with an sph greater than two in 1868 were found in the three forests except silver-maple, English walnut, eastern red cedar, black mulberry, European bird-cherry, and black willow. The large population of black mulberry in 1868 could have been a remnant plantation for the silk industry in Philadelphia (plantings occurred from 1769 to 1843; Scharf and Westcott 1884). The three forests are far from the lake or stream environments usually associated with silver-maple and black willow. English walnut was probably a minor component of the pairing with black walnut. European bird-cherry is a species that reproduces well in cutover Pennsylvania forests but is not competitive when the canopy redevelops (Elliott 1927). The loss of a large population of eastern red cedar also occurred in Seton Falls Park, New York City, NY (Loeb 1989b), and Alley Pond Park, New York City (Loeb 1992b).

The forest dynamics of the Beechwood, Country Club, and Robert's Hollow can be differentiated based upon their history of disturbances: Beechwood was periodically burned and experienced trampling from pedestrian traffic; Country Club was not affected by railroad-engine-related fires but park visitors and horseback riders had access to the site, and the forest had the dead trees removed in 1905; and Robert's Hollow did have severe fires but no regular human visitors. Chestnut, tulip-tree, choke-cherry, northern red oak, and black oak being the dominant species for Beechwood in 1907 could be explained by the fires but the question of how frequently did burning occur comes to the forefront because the species had little or no reproduction, and as Paul (1908) observed the sprouts were short lived. The absence of railroad fires is reflected in Country Club by the large population of flowering dogwood and the presence of saplings or larger sapling sph than found in Beechwood for chestnut, tulip-tree, choke-cherry, and black oak. In all three forests, chestnut, black oak, white oak, rock chestnut-oak, and northern red oak had few or no saplings, which could have been caused by shading from the understory composed of the flowering dogwood, choke-cherry, white ash, and red maple (Lorimer et al. 1994). Although chestnut, black oak, white oak, and northern red oak were well represented in Robert's Hollow, the large population of rock chestnut-oak was unique. Perhaps relatively little trampling in the secluded location of Robert's Hollow permitted rock chestnut-oak to reproduce and grow more quickly as compared to chestnut, black oak, white oak, and northern red oak.

During the century from 1907 to 2007, two changes affected all three forests: chestnut was lost, which opened the canopy soon after 1907, and white-tailed deer began to browse after 1980. The cessation of railroad fires by 1955 changed conditions only in Beechwood and Robert's Hollow (no evidence of fire was found in the three forests during the 2007 field work). The amount of trampling in Beechwood and Country Club was reduced by the end of trolley service in 1946. Aughanbaugh (1935) predicted from observations in old growth rural forests of southern Pennsylvania that the canopy openings left by the loss of chestnut would be filled with the fast growing species northern red oak and tulip-tree. The total sph for northern red oak in each of the three old growth forests in Fairmount Park is nearly equal, which is not the situation for the landscaped old growth urban forests of New York City (Stalter and Kincaid 2008). Tulip-tree nearly follows the pattern of northern red oak but the sapling sph for Country Club and Robert's Hollow is less in 2007 than 1907. Again, New York City forests have very different sph values for tulip-tree with lowland forests having the highest sph (Fitzgerald and Loeb 2008).

In 1907, white oak appeared to be poised to disappear from the forests but instead the 2007 data revealed expansions in the subcanopy size class in all three forests and the first reported presence of saplings in Country Club and Robert's Hollow. The increases in white oak differ from the declines in rural forests (Abrams 2003), which may be related to the higher levels of disturbance in the three Fairmount Park forests then occurred in rural forests. Comparing 1985 to 2001 in the landscaped old growth urban forest of Inwood Hill Park, New York City (Fitzgerald and Loeb 2008), white oak trees and saplings had lower sph values; however, the total sph in 2001 was more than double the values for the forests of Fairmount Park in 2007. American beech almost disappeared from Beechwood by 2007 but the sph for saplings rose in both Country Club and Robert's Hollow. The American beech sapling increase is primarily from roots sprouts in reaction to root trampling (Busby et al. 2008) but also could be related to a preference against browsing by white-tailed deer (Krueger et al. 2009). A rise in American beech root sprouts occurred in Seton Falls Park (Loeb 1982) and the Hemlock Forest, New York Botanical Gardens, New York City (Rudnicki and McDonnell 1989) without the presence of white-tailed deer. The combination of pedestrian and horseback riding causes greater damage to tree roots (Landsberg et al. 2001) which may explain why American beech sprouting is more prevalent in Country Club than Robert's Hollow.

The shade-intolerant choke-cherry did not survive in the three forests and has not been reported in old growth urban forest studies in New York City (Stalter and Kincaid 2008). Black oak appears to be headed for loss from the Fairmount Park forests but continues to successfully reproduce in Inwood Hill Park (Fitzgerald and Loeb 2008). Flowering dogwood representation by just a few sapling size specimens in Country Club and Robert's Hollow in 2007 could have been caused by the recent dogwood anthracnose disease (*Discula destructiva* Redlin; Hibben and Daughtrey 1988). However, McCormick (1971) reported flowering dogwood was not present in Beechwood during 1969. Also, 90% of the flowering dogwood in Seton Falls Park was lost by 1979 (Loeb 1982), which is 4 years before the disease

was found in New York and Pennsylvania. Red maple, Norway maple, Hercules club, sweet birch, hornbeam, bitternut-hickory, shagbark-hickory, white ash, sweet gum, umbrella-tree, black gum, wild black cherry, staghorn-sumac, and sassafras took advantage of the canopy openings created by the loss of chestnut to attain gains in sph or to be first reported in 2007. Umbrella-tree seedlings are thriving in Beechwood when seedlings for all of the other species including American beech are virtually absent because of deer browsing.

In New York City landscaped old growth urban forests, the 14 species noted above had sph increases except for umbrella-tree which is a species not reported in forest composition change studies (Fitzgerald and Loeb 2008; Stalter and Kincaid 2008). Many saplings of wild black cherry and sassafras occurred in repeatedly burned areas of Seton Falls Park (Loeb 1982) but the two species became established in the unburned Country Club forest. The most puzzling change in 2007 was the virtual loss of the former dominant species in Robert's Hollow, rock chestnut-oak, even though the species is well represented in the canopy, and sapling size classes of Inwood Hill Park (Fitzgerald and Loeb 2008). Paul (1908) reported "the white and chestnut oaks are suffering severely from a green scale (*Asterolecanium variolosum*), one of the most difficult and injurious insects we have encountered When we have been able to give the trees repeated sprayings, no serious damage has resulted but many trees not so protected have succumbed." In the northeastern United States, rock chestnut-oak is the most favored host of green scale with saplings being severely affected and mature trees dying when other factors stress the tree (Parr 1940). The rock chestnut-oaks may not have been treated because of the relative isolation of Robert's Hollow and the combination of fire and green scale could have caused the dramatic loss for rock chestnut-oak trees and saplings.

Remnant Forest

Agricultural Cessation

Original Research: Saddler's Woods, Haddon Township, New Jersey

Saddler's Woods (39° 54' 8" N, 75° 3' 19" W) was named after Joshua Saddler, an escaped slave who joined a Quaker farming community. In 1868, Saddler legally prevented cutting in the forest that developed on his family farm in New Jersey (Saddler's Woods Conservation Association 2010). The old growth urban forest was inventoried in 2011 with a 60 m by 60 m plot placed to have a minimum of a 10 m separation from the bordering disturbed forests. Saddler's Creek runs through the plot. Canopy gaps are present including the one caused by the fall of a large-diameter (>100 cm dbh) black oak during winter 2011. Among the 15 species identified in the sample (Table 2.4), only American beech, northern red oak, white ash, and white oak had trees in the canopy size class (dbh>45.7 cm). American beech comprised 41% of the saplings (2.5<dbh<10.2 cm) and 37% of the subcanopy

Table 2.4 Stems per hectare for canopy trees, subcanopy trees, and saplings for Saddler’s Woods, Haddon Township, New Jersey, in 2011

Species	Saplings	Subcanopy	Canopy
	2.5 < dbh < 10.2 cm	10.2 ≤ dbh ≤ 45.7 cm	dbh > 45.7 cm
<i>Acer negundo</i>	83	56	0
<i>Acer rubrum</i>	278	417	0
<i>Betula papyrifera</i>	0	28	0
<i>Carpinus carolinana</i>	56	83	0
<i>Cornus florida</i>	28	0	0
<i>Celtis occidentalis</i>	83	0	0
<i>Fagus grandifolia</i>	834	723	83
<i>Fraxinus americana</i>	56	222	0
<i>Liquidambar styraciflua</i>	0	28	0
<i>Liriodendron tulipifera</i>	139	0	111
<i>Nyssa sylvatica</i>	28	56	0
<i>Quercus alba</i>	139	56	167
<i>Quercus prinus</i>	28	0	0
<i>Quercus rubra</i>	111	56	111
<i>Sassafras albidum</i>	167	222	0

(10.2 cm ≤ dbh ≤ 45.7 cm dbh) trees. Red maple, white ash, and sassafras had the second through fourth highest sph for subcanopy trees and no representation in the canopy.

Among the New York City historical forest ecology studies, the best comparison to Saddler’s Woods is Seton Falls Park (Loeb 1992b) because Rattlesnake Creek runs through the Park and the site was a farmland before the American Revolution and then a protected woodland associated with a farm. The most striking difference is that the Saddler’s Woods canopy sph in 2011 is more than 13 times greater than the Seton Falls Park canopy (dbh > 50 cm) sph in 1979. Also, the Saddler’s Woods subcanopy sph is almost 10 times greater than the subcanopy of Seton Falls Park (10.2 cm < dbh ≤ 50 cm) in 1979. A visit by the author to Seton Falls Park in 2010 showed continued losses of trees from the canopy and subcanopy layers and little replacement occurring. Seton Falls Park had six taxa in 1979 not present at Saddler’s Woods, American elm, black cherry, hemlock, black birch, black locust, and hickory (*Carya* spp.), while only elder maple (*Acer negundo*) was found in Saddler’s Woods but not Seton Falls. The deep soils in Saddler’s Woods could contribute to an explanation of the higher sph than found in the shallow soils of Seton Falls Park. Although vandals cut down trees in both Saddler’s Woods and Seton Falls Park, evidence of fire was observed in the Park in 1979 and 2010 and is neither evident nor reported for Saddler’s Woods. American beech having the largest sapling and subcanopy populations in Saddler’s Woods appears to be related to preference against browsing by white-tailed deer (Krueger et al. 2009). Since many of the saplings and subcanopy trees appear to be root sprouts, the high population of American beech could be related, at least in part, to human trampling (Busby et al. 2008) as occurs in Seton Falls Park (Loeb 1982). Among the three Fairmount Park forests, Country Club forest is most similar to Saddler’s Woods in terms of the distribution of stems among the canopy, subcanopy, and sapling sizes class. However, Saddler’s Woods has twice

the total sph of the Country Club forest. Saddler's Woods and Country Club both have high American beech sapling populations apparently because of root sprouts and white-tailed deer preferences against browsing American beech. Red maple having the second highest number of sph for both saplings and the subcanopy in Saddler's Woods parallels the red maple population expansion in the Beechwood, Country Club, and Robert's Hollow forests.

Forest Cutting Cessation

The Mishow Marsh Watershed, Hunter Island, Pelham Bay, Park, Bronx, New York (40° 52' 40" N, 73° 47' 4" W) is the only remnant old growth urban forest with a known ecological history of a clear-cutting event and forest redevelopment (Loeb 1998). Paleopalynological research revealed the watershed was a native American corn field from 1105 to 1165 and then planted to become a hickory forest. The British Navy clear-cut Hunter Island in 1779, and subsequent plantings for estate development and park use of Hunter Island did not affect the watershed. From 1934 to 1989 tree (dbh > 30 cm) sph rose for hickory (2.6–11.2), oak (17.5–31.2), and sassafras (0–7.3), while apple (*Pyrus* spp.), American beech, black locust, and flowering dogwood were lost from the forest.

Limited Tree Harvesting

There are two remnant forests in the New York and Philadelphia metropolitan areas which are thought to be limited tree harvesting forests: the Hemlock Forest, New York Botanical Gardens, Bronx, New York (Rudnicky and McDonnell 1989), and the William L. Hutcheson Memorial Forest, New Jersey (Buell et al. 1954). The Hemlock Forest (40° 51' 51" N, 73° 52' 34" W) is located along the ravine of the Bronx River which provides the environmental conditions for hemlock to grow and reproduce. The forest has paved roads with water hydrants to enable responses to fires. Dead chestnut trees were cut and salvaged from the woods after the chestnut blight in 1908 (Pennsylvania Chestnut Tree Blight Commission 1912). From the 1880s to the 1910s, leaves were raked out of the forest which may have affected seedling survival. Rudnicky and McDonnell (1989) resampled the canopy (dbh ≥ 15 cm) with 227 plots (15% of the Hemlock Forest area) in order to compare the forest in 1985 to the inventory of the forest noted on the New York City Department of Parks and Recreation maps created in 1937. There was a small decrease in stems per hectare for the two dominants hemlock (52 sph in 1937 and 47 sph in 1985) and oak species (from 34 sph in 1937 to 31 sph in 1985). In contrast, there were large increases for red maple (4 sph in 1937 and 27 sph in 1985), black cherry (from 2 sph in 1937 to 14 sph in 1985), and white ash (1 sph in 1937 as compared to 16 sph in 1985). The history of trampling and annual fires causing loss of organic material in the soil of the Hemlock Forest that began with the founding of the Botanical Gardens in 1889 inhibits hemlock reproduction (Rudnicky and McDonnell 1989). Based on my personal observation in 2007 the canopy population of hemlock continues to decline in the Hemlock Forest, which

appears to be related to the effects of the introduced hemlock woolly adelgid (*Adelges tsugae* Annand; Orwig et al. 2002). In Seton Falls Park, hemlock declined from 1936 to 1979 (Loeb 1982) and was lost by 2007.

The William L. Hutcheson Memorial Forest (40° 29' 55" N, 74° 33' 48" W) is a remnant forest associated with a farm but the forest was not cleared for agriculture. Although no record exists, presumably chestnut trees were lost to the chestnut blight. There are records of the losses of trees to a severe storm in 1950 and the downed timber was recovered for sale. Some of the downed trees had colonial-period fire scars (Buell et al. 1954). Wind is the predominant cause of tree losses with tree falls occurring more frequently at poorly drained sites than well drained sites. Oppositely, trees lost to wind snapping the trunk occurred more frequently in well-drained sites than poorly drained sites (Reiner and Reiner 1965). Overall, wind caused more trees losses in the poorly drained sites than the well-drained sites and the resultant canopy gaps permitted the advance of red maple and white ash into the canopy (Monk 1961a). In 1961, the number of seedlings per tree for sugar maple and Norway maple were far greater than the other species in the forest (Monk 1961b). Tree deaths related to a severe summer drought in 1957 were greatest for red maple, dogwood, and white oak (Small 1957). A study of tree seedling survivorship over a 15 month period (1979–1980) including a severe summer drought in 1980 showed that more than 70% of the Norway maple, hickory, white ash, black cherry, and white oak seedlings survived while less than 30% of the red maple, dogwood, and sweet cherry seedlings survived (Davison 1981).

Sulser (1971) examined change in forest composition by resampling (sampling plot locations approximated from a field notebook) 20 years after unpublished measurements in 1950 were taken prior to the storm that extensively damaged the forest. The only significant differences found were sapling increases for red maple and white ash. In 2003, Aronson (2007) resampled the plots measured in 1950 and 1970. Flowering dogwood tree importance value (IV) dropped from 149.3 in 1950 to 17.4 in 2003 and the sapling IV also dropped from 131.9 in 1950 to 0 in 2003. White oak tree IV declined from 69.6 in 1950 to 4 in 2003 and saplings were not present in either year. Black cherry tree IV rose from 0 in 1950 to 52.2 in 2003 while the saplings IV only increased from 3 to 9.3 over the same period. Red maple tree IV jumped from 4.5 in 1950 to 97.4 in 2003 but in contrast sapling IV was lower with 16.8 in 1950 and 9.3 in 2003. American beech trees and saplings disappeared from the sampling plots. Finally, Aronson (2007) found the combination of deer browsing and the invasive species stilt grass (*Microstegium vimineum*) caused regeneration failure for the canopy species and permitted the seedling populations of tree of heaven and white ash to expand.

Regional Synthesis of Historical Continuity

The paleopalynology record of the past century indicated five major changes affecting forest dynamics in the oak chestnut region: release of lands from agriculture, loss of chestnut, decimation of hickory, urban expansion instead of rural forest

development, and urban tree plantings (Loeb 1989a). A paleopalynological study of one remnant old growth urban forest in New York City revealed pre-European settlement Native American plantings of hickory occurred soon after corn field abandonment (Loeb 1998). Floras (Barton 1818; Torrey 1819) published after the American Revolution, the regional forest resetting event, revealed 98 arboreal species with only 54% of the species being present in both the New York City and Philadelphia metropolitan areas. Even though alien species comprised only 5% of the total, more than 90% of the arboreal species were used for arboricultural plantings before the floras were published, which highlights the extensive use of native species in estate and garden plantings starting in the early colonial period (Fry 1996; Fitzgerald and Loeb 2008). Witness tree records (Loeb 1987) of pre-European settlement forests showed oak species were two-thirds of the forest and chestnut was the second dominant genus, which confirms the classification of the region as oak-chestnut forest. The most common oak species in the witness tree records were in descending order white oak, black oak, and northern red oak. The predominance of oaks indicates fire affected pre-European settlement forest, which is supported by the fire scar analysis of trees in the William L. Hutcheson Memorial Forest (Buell et al. 1954). However, twelfth-century native American plantings of hickory in the Mishow Marsh Watershed (Loeb 1998) points out the possibility of prehistoric arboricultural influences on species composition.

A treatise on street tree management and plantings in the New York and Philadelphia region (Solotaroff 1912) included the results of a 1911 street survey for East Orange, New Jersey. Three species, red maple, silver-maple, and sugar maple, comprised more than two-thirds of the 10,953 trees. By 2004, the street forest was down to 6605 trees with red, silver, and sugar maples being less than 10% of the total. Solotaroff indicated several problems with the growth of each species which made the trees unsuitable for urban conditions. In contrast, Solotaroff recommended Norway maple which had a slightly smaller population in 2004 than 1911 but became the second most common species behind pin oak. A comparison of surveys for the street old growth urban forests of Haddonfield and Moorestown in Camden County, New Jersey with the East Orange results indicates the species diversity of the two forests was more than three times greater than East Orange. Criteria for the assessment of whether a particular forest is a street old growth urban forest are the largest size class (dbh > 45.7 cm) has the highest sph for all species and the species with the greatest sph match the species in historical street forests (e.g., East Orange in 1911) or historical floras (such as found in Table 2.1). Haddonfield matched the criteria well but Moorestown did not which revealed how tree diseases and planting plans to increase diversity affect patterns in the historical continuity of street old growth urban forests.

In the three landscaped forests of Fairmount Park, the losses for chestnut, flowering dogwood, and choke cherry permitted American beech, black gum, northern red oak, Norway maple, red maple, sassafras, tulip-tree, and white oak to establish sapling populations and have increased presence in the subcanopy and canopy layers. The Robert's Hollow forest's former dominant species rock chestnut-oak appears to have been lost to intense fire and green scale infestation, which permitted black cherry, black locust, Norway maple, and red maple to become more

important in the forest. Studies of old growth landscaped forests in New York City parks (Loeb 1982; Fitzgerald and Loeb 2008; Stalter and Kincaid 2008) indicate rock chestnut-oak survives and reproduces in the presence of ground fires. However, the fires permitted black cherry, black locust, Norway maple, and red maple to become more important in the sapling, subcanopy, and canopy layers of the New York City forests. In contrast, the absence of fire in a Cleveland, Ohio, area old growth urban forest has resulted in dominance by sugar maple (not found in the three Fairmount Park forests) and American beech (Loeb 2001). American beech reproduction was greatest in the Country Club forest in 2007 because of root sprouting in response to trampling by people and horses. Although Beechwood forest has virtually no American beech trees, deer browsing appears to have poised umbrella-tree saplings to advance in the subcanopy and canopy without competition. In the New York City old growth landscaped forests noted above, American beech reproduces well through root sprouting in the absence of white-tailed deer and umbrella-tree.

The Mishow Marsh Watershed Forest recovered after a clear-cut in 1779 but more important to the forest recovery process was the planting of hickory by native Americans in the twelfth century. Hickory and oak increased in the forest canopy from 1934 to 1989 but sassafras appeared and became a canopy species since 1934 because of arson (Loeb 1998). American beech was lost from the forest, which could be related to the absence of white-tailed deer on the island. For Saddler's Woods, New Jersey, white-tailed deer and human trampling have led to American beech becoming the dominant species among saplings and subcanopy trees. Also white ash, red maple, and sassafras have advanced in the sapling and subcanopy layers but evidence of fire is not obvious in Saddler's Woods.

Researchers in the two limited cutting remnant old growth urban forest, the Hemlock Forest (Rudnicky and McDonnell 1989) and William L. Hutcheson Memorial Forest (Aronson 2007), predicted the oak species dominance of the canopy will be lost to black cherry, red maple, and white ash, as well as the possibility of tree of heaven becoming part of the canopy. However, the reasons for the changes in each forest are different. In the William L. Hutcheson Memorial Forest, regeneration is succumbing to deer browsing and stilt grass invasion, but neither of these species is a concern for the Hemlock forest. Instead, the loss of canopy species regeneration is due to trampling and periodic fire destroying the organic layer of the soil. An explanation for the changes in the Hemlock forest comes from studies of the seed bed germination in urban soils. In two New York City parks, high germination rates were found for tree of heaven, black birch, sweet gum, and tulip tree but there was no evidence of seed germination for maple species, oak species, white ash, flowering dogwood, and sassafras was found (Kostel-Hughes et al. 1998b). Also, Kostel-Hughes et al. (1998a) found that small-seed trees (red maple, sweet gum, tree of heaven, tulip-tree, white ash, witch hazel, sugar maple, sassafras, black cherry, and striped maple) germinated in urban soil leaf litter that was less than half the depth of the urban leaf litter in which large-seed trees (species of oak and hickory) germinated.

The forest resetting event for the old growth urban forests of the New York and Philadelphia metropolitan areas was the American Revolution. Witness tree records provided evidence for the existence of the oak-chestnut forest region prior to the war and the historical floras indicated intraregional variation in species distribution and frequency of occurrence. Arboricultural use of native species as early as 1600 and through 1850 brings into question the origin of trees in old growth urban forests. Are the trees from arboricultural plantings or spontaneous regeneration of local trees? Street old growth urban forests shift from dominance by species of maple at the turn of the twentieth century to dominance by oak, linden, and plane trees at the start of the twenty-first century. Landscaped forests composed of native and alien species, such as the Centennial Fairgrounds in Fairmount Park, have had large species diversity losses. The historical continuity of landscaped and remnant forests dominated by native species was disrupted by species losses or tree population drops to the chestnut blight, Dutch elm disease, hickory bark beetle, hemlock woolly adelgid, and green scale. Choke cherry was lost to shading by canopy trees. Major population declines for flowering dogwood are not apparently related to known diseases or insect infestations. Fire from railroad engine sparks and vandals has enabled the advance of black cherry, red maple, black locust, and sassafras. Human trampling and white-tailed deer browsing has decimated the seedlings and sapling populations except for American beech and umbrella-tree which are the species that white-tailed deer avoid browsing. Invasive species are not yet a widespread threat in old growth urban forest but there is great potential for the displacement of native tree species (Loeb 2009). The effects of tree deaths and sapling losses are not uniform across the landscaped and remnant forest sites in the New York and Philadelphia region as evidenced by an order of magnitude difference for canopy and subcanopy stems per hectare between Saddler's Woods, New Jersey, and Seton Falls Park, New York.

Looking into the future, research on more sites in the Philadelphia and New York metropolitan areas is required for statistical analysis of forest dynamics on the regional level. The focus of historical continuity research needs to be expanded to include abiotic factors such air pollution, climate warming, and progressive soil fertility exhaustion through the analysis of tree ring wood. Monitoring of sites is essential to maintain the flow of data on the changing biotic and abiotic conditions in the old growth urban forests. Finally, research on old growth urban forest must be done across the earth in order to gain an understanding of whether the factors affecting historical continuity in the oak-chestnut forest region have parallels in other forest regions of the planet.

References

- Abrams MD (2003) Where has all the white oak gone? *Bioscience* 53:927–939
Adams DW (2004) *Restoring American gardens*. Timber Press, Portland, Or
Agnoletti M, Anderson S (2000) *Methods and approaches in forest history*. CABI Publishing, New York, NY

- Anonymous (1868) Map of farms and lots embraced within the limits of Fairmount Park as appropriated for public use by act of Assembly approved 14th day of April A.D. 1868. No publ, Philadelphia, Pa
- Anonymous (1916) The dying hickories on Long Island. Branch For Shade Tree Insects, Bur Entomology, US Dep Agri, Washington DC
- Anonymous (2006) Comprehensive master plan for East Orange, New Jersey. Unpubl, East Orange, NJ
- Aronson MF (2007) Ecological change by alien plants in an urban landscape. Diss, Rutgers Univ, New Brunswick, NJ
- Aughanbaugh JE (1935) Replacement of the chestnut in Pennsylvania. Bull 54, Pa Dep For Waters, Harrisburg, Pa
- Baker CA, Moxey PA, Oxford PM (1978) Woodland continuity and change in Epping Forest. Field Stud 4:646–669
- Bard GE (1952) Secondary succession on the Piedmont of New Jersey. Ecol Monogr 22:196–215
- Barton WPC (1818) Compendium floræ Philadelphicæ. Vols 1–2. M. Carey and Son, Philadelphia, Pa
- Behre K-E (ed) (1986) Anthropogenic indicators in pollen diagrams. Balkema, Rotterdam, Neth
- Birks HH, Birks HJB, Kaland PE, Moe D (eds) (1988) The cultural landscape: past, present, and future Cambridge Univ Press, Cambridge, UK
- Brande A, Böcker R, Graf A (1990) Changes of flora, vegetation and urban biotopes in Berlin (west). In: Sukopp H, Hejný S (eds) Urban ecology. Plants and plant communities in urban environments. SPB Academic Publishing, The Hague, Neth
- Braun EL (1950) Deciduous forests of eastern North America. Hafner, New York, NY
- Brown CS (2005) Finding of no significant impact and decision. Environmental assessment – shooting white-tailed deer to assist the city of Philadelphia, Fairmount Park Commission in achieving deer population reductions on park properties located in the Pennsylvania counties of Delaware, Montgomery and Philadelphia. US Dep Agric, Anim Plant Health Insp Serv, Wildl Serv, Harrisburg, Pa
- Buell MF, Buell HF, Small JA (1954) Fire in the history of Mettler's Woods. Bull Torrey Bot Club 81:253–255
- Busby PE, Motzkin G, Foster DR (2008) Multiple and interacting disturbances lead to *Fagus grandifolia* dominance in coastal New England. J Torrey Bot Soc 135:346–359
- Cho D, Boerner REJ (1991) Structure, dynamics, and composition of Sears Woods and Carmean Woods State Nature Preserves, north-central Ohio. Castanea 56:77–89
- Collins S (1956) The biotic communities of the Greenbrook Sanctuary. Diss, Rutgers Univ, New Brunswick, NJ
- Corson A (1937) Report of landscape gardener for the year of 1936. In: (no ed) Fairmount Park annual report of the chief engineer for the year 1936. Fairmount Park Com, Philadelphia, Pa
- Cox HE (1970) The Fairmount Park trolley, a unique Philadelphia experiment. Harold Cox, Forty Fort, Pa
- Cresson J (1868) Report of the chief engineer of Fairmount Park. In: (no ed) Second annual report of the Fairmount Park Commission. Fairmount Park Com, Philadelphia, Pa
- Crowder MJ, Hand DJ (1990) Analysis of repeated measures. Chapman and Hall, London, UK
- Davidson SE (1981) Tree seedling survivorship at Hutcheson Memorial Forest. William L Hutcheson Meml For Bull 6:4–7
- DeCandido RV, Lamont EE (2004) The historical and extant vascular flora of Pelham Bay Park, Bronx county, New York 1947–1998. J Torrey Bot Soc 131:368–386
- Dimbleby GW (1985) The palynology of archaeological sites. Academic Press, Orlando, FL
- Egan D, Howell EA (eds) (2001) The historical ecology handbook: a restorationists' guide to reference ecosystems. Island Press, Washington, DC
- Elliott HE (1927) What follows pulp and chemical wood cutting in northern Pennsylvania. Bull 43 Spec Stud Ser, Comm Pa, Dep For Waters, Harrisburg, Pa
- Fairmount Park Commission (1868) Fairmount Park, Philadelphia with limits, as prescribed in Act of Assembly, approved March 26th, 1868 showing the trees and woods nearly as now existing with a study for roads and paths. Worley & Bracher, Philadelphia, Pa

- Fairmount Park Commission (1895) Proposal for cutting grass in Fairmount Park for the season of 1895. MC 999.13, Eli Kirk Price Family Papers, Arch Spec Collect, Dickinson College, Carlisle, Pa
- Fairmount Park Commission (1900) Topographical map of Fairmount Park Philadelphia excepting Wissahickon Valley. Fairmount Park Com, Philadelphia, Pa
- Fitzgerald JM, Loeb RE (2008) Historical ecology of Inwood Hill Park, Manhattan, New York. *J Torrey Bot Soc* doi:10.3159/07-RA-046.1
- Fry JT (1996) An international catalogue of North American trees and shrubs: the Bartram broadside, 1783. *J Gard Hist* 16:3–66
- Gibson J, Eck RO, Jones T, Sterling R, Hartman J, Hannum W, Nichols RE, Thomas W, Leusner B (2010) Community forestry management plan 2010–2014 township of Moorestown. Unpubl, Moorestown, NJ
- Gleason HA (1939) The individualistic concept of the plant association. *Am Midl Nat* 21:92–110
- Gleason HA, Cronquist A (1991) Manual of vascular plants of the northeastern United States and adjacent Canada. New York Bot Gardens, New York, NY
- Graney DL (1987) Ten-year growth of red and white oak crop trees following thinning and fertilization in the Boston Mountains of Arkansas. In: Proceedings of the fourth biennial southern silvicultural research conference. Gen Tech Rep SE-42. SE For Exp Stat, For Serv, US Dep Agri, Asheville, NC
- Greller AM (1972) Observations on the forests of northern Queens county, Long Island, from colonial times to the present. *Bull Torrey Bot Club* 99:202–206
- Hannah PR (1968) Topography and soil relations for white and black oak in southern Indiana. Res Pap NC-25. NC Res Stat, For Serv, US Dep Agri, St. Paul, Mn
- Hepting GH (1977) The threatened elms: a perspective on tree disease control. *J For His* 21:90–97
- Hibben CR, Daughtrey ML (1988) Dogwood anthracnose in northeastern United States. *Plant Dis* 72:199–203
- Hoglund AW (1962) Forest conservation and stove inventors-1789–1850. *J For His* 5(4):2–8
- Holton JL (1989) The Reading Railroad: history of a coal age empire. Garrigues House, Laury's Station, Pa
- Kays R, Wilson D (2009) Mammals of North America. Princeton Univ Press, Princeton, NJ
- Keller IA, Brown S (1905) Handbook of the flora of Philadelphia and vicinity, containing data relating to the plants within the following radius: eastern Pennsylvania; all of New Jersey except the northern counties; and New Castle County, Delaware, with keys for identification of species. Philadelphia Bot Club, Philadelphia, Pa
- Kostel-Hughes F, Young TP, Carreiro MM (1998a) Forest leaf litter quantity and seedling occurrence along an urban-rural gradient. *Urb Ecosyst* doi:10.1023/A:1009536706827
- Kostel-Hughes F, Young TP, McDonnell MJ (1998b) The soil seed bank and its relationship to the above ground vegetation in deciduous forests in New York City. *Urb Ecosyst* doi:10.1023/A:1009541213518
- Krueger LM, Peterson CJ, Royo A, Carson WP (2009) Evaluating relationships among tree growth rate, shade tolerance, and browse tolerance following disturbance in an eastern deciduous forest. *Can J For Res* doi:10.1139/X09-155
- Landsberg J, Logan B, Shorthouse D (2001) Horse riding in urban conservation areas: reviewing scientific evidence to guide management. *Ecol Manag Restor* doi:10.1046/j.1442-8903.2001.00067.x
- Loeb RE (1982) Reliability of the New York City Department of Parks and Recreation forest records. *Bull Torrey Bot Club* 109:537–541
- Loeb RE (1987) Pre-European settlement forest composition of East New Jersey and southeast New York. *Am Midl Nat* 118:414–423
- Loeb RE (1989a) Lake pollen records of the past century. *Palynology* 13:3–19
- Loeb RE (1989b) Historical ecology of an urban park. *J For Hist* 33:134–143
- Loeb RE (1990) Measurement of vegetation changes through time by resampling. *Bull Torrey Bot Club* 116:173–175
- Loeb RE (1992a) Will a tree grow in Brooklyn? Developmental trends of the New York City street tree forest. *J For* 90(1):20–24
- Loeb RE (1992b) Long-term human disturbance of an urban park forest, New York City. *For Ecol Manag* doi:10.1016/0378-1127(92)90142-V

- Loeb RE (1998) Urban forest management and ecosystem change during the past millennium: a case study from New York City. *Urb Ecosys* doi:10.1023/A:1009545331265
- Loeb RE (2001) Fire in the urban forest: long-term effects in old growth stands. *Arboric J* 25:307–320
- Loeb RE (2009) Biogeography of invasive plant species in urban park forests. In: Kohli R, Jose S, Batish D, Singh H (eds) *Invasive plants and forest ecosystems*. CRC/Taylor and Francis, London, UK
- Loeb RE (2010) Diversity gained, diversity lost: long-term changes in woody plants in Central Park, New York City and Fairmount Park, Philadelphia. *Stud Hist Gard Des Landsc* doi:10.1080/14601170903040819
- Lorimer CG, Chapman JW, Lambert WD (1994) Tall understorey vegetation as a factor in the poor development of oak seedlings beneath mature stands. *J Ecol* 82:227–237
- Marx TG (1976) Technological change and the theory of the firm: the American locomotive industry 1920–1955. *Bus Hist Rev* 5:1–24
- McBride JR, Nowak DJ (1989) Urban park tree inventories. *Arboric J* 13:345–361
- McCarthy BC, Quinn JA (1990) Reproductive ecology of *Carya* (*Juglandaceae*): phenology, pollination, and breeding systems of two sympatric tree species. *Amer J Bot* 77:261–273
- McCormick J (1971) An ecological inventory of the West Park, Fairmount Park, Philadelphia, Pennsylvania. Jack McCormick and Associates, Philadelphia, Pa
- Miller JA (1965) The changing forest: recent research in the historical geography of American forests. *J For His* 9:18–25
- Monk CD (1961a) The vegetation of the William L. Hutchison Memorial Forest, New Jersey. *Bull Torrey Bot Club* 88:156–166
- Monk CD (1961b) Past and present influences on reproduction in the William L. Hutchison Memorial Forest, New Jersey. *Bull Torrey Bot Club* 88:167–175
- Nriagu JO (ed) (1978) The biogeochemistry of lead in the environment Part A. *Ecological Cycles*. Elsevier, Amsterdam, Neth
- Orwig DA, Foster DR, Mausel DL (2002) Landscape patterns of hemlock decline in New England due to the introduced hemlock woolly adelgid. *J Biogeogr* doi:10.1046/j.1365-2699.2002.00765.x
- Parr T (1940) *Asterolecanium variolosum* Ratzeburg, a gall-forming Coccid, and its effect upon the host trees. Number 40, *Bull Yale Sch For*
- Paul O (1908) Report on the trees of Fairmount Park; a study of the trees growing naturally in the park forests and of those planted for shade or decorative purposes, including the outline of a general forestry policy suggested for their future care. Fairmount Park Com, Philadelphia, Pa
- Peglar SM, Fritz SC, Birks HJB (1989) Vegetation and land-use history at Diss, Norfolk, U.K. *J Ecol* 77:203–222
- Pennsylvania Chestnut Tree Blight Commission (1912) The chestnut blight disease. *Bull* 1, C E Aughinbaugh, Harrisburg, Pa
- Peter D (2008) Tree succession planning: modelling tree longevity in Tutangga/Park 17, the Adelaide park lands. Diss, Univ Adelaide, Adelaide, Aust. <http://digital.library.adelaide.edu.au/dspace/handle/2440/48538>. Accessed 12 March 2011
- Peterken GF, Backmeroff C (1988) Long-term monitoring in unmanaged woodland nature reserves. *Res Surv Nat Conserv Ser* 9, Nat Conserv Coun, Peterborough, UK
- Rackham O (2003) *Ancient woodland its history, vegetation and uses in England*, 2nd edn. Castlepoint Press, Colvend, UK
- Rehder A (1940) *Manual of cultivated trees and shrubs hardy in North America: exclusive of the subtropical and warmer temperate regions*, 2nd edn. Macmillan, New York, NY
- Reiner NM, Reiner WA (1965) Natural harvesting of trees. *William L. Hutchison Meml For Bull* 2:9–17
- Rotherham ID, Jones M, Smith ML, Handley C (eds) (2008) *The woodland heritage manual a guide to investigating wooded landscapes*. Wildtrack Publishing, Sheffield, UK
- Rothrock JT (1880) Catalogue of trees and shrubs native of and introduced in the horticultural gardens adjacent to Horticultural Hall, in Fairmount Park. No publ., Philadelphia, Pa
- Rudnicki JL, McDonnell MJ (1989) Forty-eight years of canopy change in a hardwood-hemlock forest in New York City. *J Torrey Bot Soc* 116:52–64

- Russell EWB (1979) Vegetational change in northern New Jersey since 1500 A.D.: a palynological, vegetational, and historical synthesis. PhD thesis, Rutgers Univ, New Brunswick, NJ
- Saddler's Woods Conservation Association (2010) History. <http://www.saddlerswoods.org/35807.html>. Accessed 21 March 2011
- Scharf JT, Westcott T (1884) History of Philadelphia, 1609–1884. Everts, Philadelphia, Pa
- Seppä H (1997) The long-term development of urban vegetation in Helsinki, Finland: a pollen diagram from Töölönlahti. *Veg Hist Archaeobotany* doi:10.1007/BF01261957
- Sidney JC, Adams A (1859) Description of plan for the improvement of Fairmount Park. Merrihew and Thompson, Philadelphia, Pa
- Small JA (1957) Drought response in William L. Hutcheson Memorial Forest. *Bull Torrey Bot Club* 88:180–183
- Solomon AM, Kroener DF (1971) Suburban replacement of rural land uses reflected in the pollen rain of northeastern New Jersey. *Bull NJ Acad Sci* 16:30–44
- Solotaroff W (1912) Shade-trees in town and cities their selection, planting, and care as applied to the art of street decoration; their diseases and remedies; their municipal control and supervision. John Wiley & Sons, New York, NY
- Stalter R, Kincaid D (2008) A 70-year history of arborescent vegetation of Inwood Park, Manhattan, New York, U.S. *Arboric Urban For* 34:245–251
- Sulser JR (1971) Twenty years of change in the Hutcheson Memorial Forest. *William L Hutcheson Meml For Bull* 2:15–25
- Taylor N (1915) Flora of the vicinity of New York: a contribution to plant geography. Vol 5 *Memoirs New York Bot Gard*, New York, NY
- Torrey J (1819) Catalogue of plants growing spontaneously within thirty miles of the city of New-York. Websters and Skinners, Albany, NY
- Traverse AT (2007) *Paleopalynology*, 2nd edn. Springer, London, UK
- Vera FWM (2000) *Grazing ecology and forest history*. CABI Publishing, New York, NY
- Ward JS, Parker GR, Ferrandino FJ (1996) Long-term spatial dynamics in an old-growth deciduous forest. *For Ecol Manag* 83:189–202

Old Growth Urban Forests

Loeb, R.E.

2011, XI, 78 p. 15 illus., 7 illus. in color. With online
files/update., Softcover

ISBN: 978-1-4614-0582-5