DEFINING AND MEASURING SUCCESS IN THE URBAN FOREST

By

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ABSTRACT OF THE DISSERTATION

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An in-depth understanding of what constitutes success is needed in order to assess current management practices and improve to ensure a more stable urban canopy trajectory in the future. This dissertation is comprised of three studies that investigate three discrete time pieces in the urban forest in order to determine and measure their success.

The early transplant survival study found after two years post-transplant, urban trees had a 91.4% survival, whereas the survival declines 8-9 years post-transplant to 75.8%. Trees had the lowest survival in downtown areas and increasing survival as a residential gradient was reached.

The parking lot study examined trees approximately 20 years post-transplant in order to determine a size reduction based on amount of apparent available soil. Tree Diameter Breast Height (DBH) was fond to be a useful predictor of tree canopy area. There was a reduction in canopy area seen across all five species measured as apparent available soil decreased.

The maximum size study linked terminal size (stem diameter) to site type based on apparent available soil when trees were grouped into categories based on their published height expectations (small, medium, large). Maximum height was different in all three plating site types, irrespective of size class. Overall a reduced planting space resulted in a reduced maximum size, which serves as service endpoint for managers.

Dedication

To every student who has had what seems like endless questions, and to the patient teacher who guides them to the answers. Thank you for all the teachers who lead me through this journey, and my students who kept me on track.

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Chapter 1: Introduction

What is the Urban Forest?

The definition, much like the field of urban forestry is fluid, changing over time and circumstance, depending on the perspectives of the author and political climate. A few key concepts surround the evolution of a consensus definition of the practice and discipline of urban forestry. In 77 posited definitions, since its recent academic inception, common themes are displayed: resource management, urban areas, and forestry as management (Hauer et al. 2010). The most recent published definition of urban forestry revolves around the central idea that an urban forest is a dynamic ecosystem that can provide environmental services to its surroundings (NUFAC 2011). Simply put, the urban forest is an area of high population density where trees are seen as a shared community resource.

The practice of urban forestry is still fluid and with a limited research base, there has been a goal of standardization (e.g. *i*-Tree, standardized data initiative); however a standard practice has yet to emerge. Most management recommendations to date have been proposed as informed opinion rather than as researched areas of practice. A more systematic approach is needed as the stakes become higher rather than one of reliance of accumulated field wisdom of active practitioners. The urgent needs of practical application have dictated the direction of research in the field, but inadequate knowledge of survival rates, pest infestations, age, size expectations, and senescence size still exist. The effectiveness of management will depend on such information.

The Importance of Establishing a Forest in a Municipal Setting

The presence of trees in the urban environment often arouses emotion and passion in people who share that environment. Descriptions of trees as the "noblest expression of plant life" or the "aristocrats of the plant kingdom" (Wilson 1974 p v) convey a strength of emotional connection towards what are in effect, non-life-essential commodities. The lack of nature in urban environments is also expressed as an undesirable feature of urban living, as in Jim's (2000 p. 271) description of trees as "forgotten companions of the natural world," or Ng's description of trees (1981 p. 25) as the "unsung heroes of a cold concrete and steel environment."

Trees in the urban setting have a value. The value is declared around the world when people protest the removal of trees by chaining themselves to them (Munson, 1993) or tie yellow ribbons around branches (Dwyer et al. 1991). Many towns and cities have legislation to protect trees and restrict removal (Cooper 1996). In New Jersey, this is seen in the No Net Loss Reforestation Act, where state entities are required to replant trees when they are removed during development projects involving one-half acre or more. It is also demonstrated through the New Jersey Shade Tree and Community Forestry Assistance Act (s-591/A-926), which provides liability protection to participating municipalities to ensure a more livable community through the care and management of trees throughout New Jersey. New Jersey and New York have contrasting legislation for urban trees. New York has no current legislation on the removal of trees on private land, but does require permits for public tree removal.

Connecting with nature has often been purported as beneficial for urban dwellers (Ulrich 1981). The social benefits of urban trees extend to psychological well-being, reduced stress, increased enjoyment of everyday life, and the facilitation of a community (Ames 1980, Getz et al. 1982, Dwyer et al. 1992). Urban trees are also credited for influencing the physical and biological elements of the environment. These ecosystem services are illustrated through: energy conservation, carbon dioxide sequestration, improved air quality, increased rainfall percolation to name a few (Grey and Deneke 1986, Dwyer et al. 1992, Simpson 1998, Nowak and Crane 2002).

Historical Context

Cities arose early in human civilization, and the Egyptians, Phoenicians, Persians, Greeks, and Romans all held trees in high esteem, recognizing their aesthetic and emotional benefits (Lawerence 1988). The hanging gardens in Babylon placed vegetation in an urban setting (Philliips 1993). In the arid climate of ancient Egypt, trees provided shade and cooler environments for the ruling class (Zube 1973). Paintings in Egyptians tombs show trees in regularly spaced rows and geometric patterns (Dickins 1985). Despite the majority of early plantings in temples or as pleasure gardens for the ruling class, there is evidence of street tree plantings. Greek cities contained plane tree and poplars (Phillips 1993). The Romans valued trees along roadsides, because they were seen from a great distance, and therefore clearly indicated the course of the road (Nadel et al. 1977). The great Khan ordered trees lining the walkways of Shangdu. Although many acknowledge the idea of trees for the urban masses as relatively new (Neales, 1992, Miller 1997), trees had important roles in government not only in citizens perceptions, but also in ecological and environmental impact (Gobster 2001).

The concept of an organized urban forest is a relatively new academic discipline in North America, but as cities grow, the development of designs involving landscapes bring about innovation and creativity (Jorgenson 1970, Spirn, 1984, Botkin and Beveridge 1997, Grove and Burch 1997). Vegetation is essential to achieving the quality of life that creates a great city and makes it possible for people to live reasonable lives within urban developments (Botkin and Beveridge 1997). Whether trees provide amenity or emotional value, there seems to be an implicit understanding that the value of trees is somehow transferrable from the natural forest into urban settings.

The Urban Context

There has been a steady trend toward an increasing urban population since the inception of the industrial revolution (1843). By the 1900s, the United States population had risen to slightly more than 76 million, about 40% of who were living in urban locations (Makun 2009). Currently, the numbers of people living in urban areas is increasing, and the urban population is growing at a much faster rate than the population as a whole. In most developing regions around the world, the proportions of people living in the largest cities are also increasing (Nowak 1995, Nowak 2000, Makun 2009). More than 80% of the current United States population lives in urban and urbanizing areas (Makun 2009).

A Predictable Future

With demographic shifts across the globe causing rapid urbanization, especially concentrated in the developing world, the issue of urban development is likely to become a focal point in the near future. In China, rapid urbanization is urgent with the government erecting 100 new cities with a million residents and the corresponding amenities those populations need (World Bank 2009). Organized urban forestry will play an important role in meeting the needs of such urban populations, avoiding urban blight, improving air quality, lowering the heat island effect, while also addressing the social and environmental problems that arise from urbanization.

GUIDING CONCEPTS OF URBAN FOREST MANAGEMENT

Plant Selection and Strategy

There are many confounding factors into the selection of a tree for planting: overall physical attractiveness of the species, consideration of the debris the tree will create throughout its lifetime, the possibility of property damage from tree growth, and many other factors such as site condition for tree placement and finding the "perfect tree" for the site, shade quality, and overall growth pattern to name a few. In the face of complexity, there is a simplistic mantra of "right tree, right place". Yet public policy informs this recruitment of urban trees, and some trees will not be planted in the urban environment because the species is "not desirable" in the urban context. An example of this public policy can be found examining

Gingko biloba: female trees are very seldom planted in urban environments due to their large malodorous fruits that become messy when falling into pedestrian areas and create a slipping hazard. Perceptions of good and bad influence human requirement in designed ecological systems.

Management of the issues from selection become part of an urban manager's job, since trees need to withstand a wide variety of exaggerated stresses, all of which impact the overall growth and survival of a tree. Urban areas are quite heterogeneous, and stresses vary considerably, even among adjacent planting sites. Berrang (1985) studied over 80 variables for 375 trees planted near the Consolidated Edison facilities of New York City and found that excess soil moisture, mounding of soil on roots, soil salts, and overall root system size are the most important factors affecting a tree's overall health. Chacalo (1994) surveyed 1261 street trees in Mexico City and concluded that the problems with overall tree health could be attributed to planting in inappropriate location, overall species choice, and lack of adequate maintenance and planning. Both of these studies conclude planting selection is a site-specific challenge.

Frequently, as is illustrated in Mexico City, there is no careful attention devoted to choosing an appropriate species for the planting site and trees are planted as monocultures (Meza 1992). Moreover, in many cases, the plantings are not properly spaced, which in-turn causes excessive competition; pruning is performed too frequently as well as improperly, causing severe wounding of the tree. Tree type contributes to overall survival, but just as important is tree size at planting. Nowak et al. (2004) demonstrated that planting stock with smaller diameters have a longer life span than conspecific material with a larger diameter at planting. This is a common horticultural problem, planting seedling/small saplings is usually more efficacious already large. The quick gratification of the larger tree, is quickly diminished by the plant's ultimate survival.

Trees in the urban forest on public lands, on which managers have authority to act, only represent about 20% of the forest (Miller 1997). The challenge with the discipline of urban forestry is moving public behavior towards best practices. This dissertation will only focus on the 20% of the urban forest that is represented on public lands that would later inform best practices in urban areas. The 80% of trees that are found on private lands would be better managed and informed when urban foresters have a better idea of what constitutes a best practice to inform the public.

Species Diversity

Tree species composition varies widely among cities, a reflection of variable geographies, histories, and cultures (Detwye 1972, Grey and Denke 1986, Miller 1997). Environmental conditions define biological tolerance and thus the range of credible species choices for any particular location (Nowak et al. 2004). Species selections by managers for public areas and preferences of property owners for their own properties have contributed to the current urban forest makeup. Species selection is critical when determining the success of a tree species in New Jersey versus California. The species diversity of the urban forest will tend to decrease as the number of trees to be planted rises, since availability varies inversely with market demand (Bond 2006). The implication is that very large cohorts are dominated by a small number of species unless careful advanced planning is conducted. This practice of limited diversity can cause serious problems in the long-term health of the urban forest. Planned diversity acts as an insurance policy against potential future losses from (currently unknown) pests and diseases.

One can plan for species diversity in the urban setting, though it is often the case that the array of plant species used is limited. Kielbaso (1989), reporting on a survey of urban foresters, found agreement in the diversity of street plantings to help insure the health of the collective investment of the urban forest. Santamour (1990) suggested a 10-20-30 rule for the urban planner; in which no more than 10% of the total planting shock should be from one species, no more than 20% from one genus; and no more than 30% from any one taxonomic family. A formulaic approach to diversity, as a management strategy, attempts to limit losses in an environment that is prone to frequent disease or pest outbreaks and has little in the ways of damage control against cosmopolitan pests. This strategy has guided urban plant selection for 20 years and has been examined in many inventory analyses. Some have suggested a 5-10-15 (no more than 5% from one species, 10% from one genus, and 15% from one family). While one could debate that more diversity is better, it is already difficult to maintain with proactive effort given site limits, market (supply and stock diversity) forces, and client demands. Unless there is direct market access, it is very hard to achieve a higher diversity. Site variables, aesthetic considerations, and maintenance requirements will always limit the list of species that may work in any particular urban location. Within those limitations, maintaining diversity is of the utmost importance. A growing number of municipalities have ordinances which follow the 10-20-30 rule, that specify the minimum number of tree species to be represented within a block (development) (Smiley 1988). Some have criticized Santamour's suggestion as overly simplistic (Lacan and McBride 2008) and have suggested other techniques to manage diversity in the urban forest; Santamour's stocking balance is used extensively in the Northeast for municipal tree planting programs.

There are several recent efforts to identify relationships between species diversity or vegetation distribution and a number of social and physical factors. Hope et al. (2003) found relationships between species diversity, current/former land use, elevation, family income, and housing age. Grove et al. (2005) examined plant diversity in relation to population, lifestyle behavior, and social stratification. Martin et al. (2003) related vegetation richness and abundance to neighborhood socioeconomic status. All of these studies found greater species diversity and survival in higher socioeconomic neighborhoods and suggested 'habitat fit' lays the groundwork for 'behavioral fit'. The available data also suggests that higher socioeconomic status leads to better designs and more expensive materials, while lower socioeconomic clients have lower designs with less healthy stock and less plant material selection. These affluent communities have a great tax resource, which lends the ability to a greater urban tree canopy. Further, neighborhoods with no trees had higher crime rates, while neighborhoods with highest species diversity had a higher sense of community and socioeconomic status. Increased canopy coverage has been seen as a predictor of an affluent community. Previous research has helped to link the idea of a "fit setting" providing for "fit habitats" in designed setting. Tree canopy coverage has been linked to a lower crime rate and heightened consumer traffic in commercial districts (McPherson et al. 1994, Prow 1999, Nowak et al. 2002, Kuo 2003, Wolf 2003). Although trees may not be the direct cause of this relationship there is undoubtedly a correlation. The need for an urban canopy is one that is not just meant for the wealthy, but for the general good of society.

Newly Planted Street Trees

When such care is made in tree selection, more knowledge is needed to develop expectations for the survival of newly planted street trees. An increasingly dense-residential population, shifts in service demands, changes in infrastructure design, increased vehicular traffic and poor air quality all translate into environmental stresses on plants. Buildings, pavement, and other hardscape features restricted above and below ground resource access, furthering challenges to newly planted urban trees. The first years are the most crucial to transplanted trees and early losses need to be countered. The obvious need to insert the 'right tree' into the 'right hole', a major challenge for cities developing million tree initiatives (PLANYC, Million Trees LA). Where site by site analysis is not practical in urban scenarios, so determining a common practical site index is crucial (Luley and Bond 2006). The development of an urban site index for trees is one of the central focuses of this dissertation.

Knowledge of why trees died is crucial in order to understand and prepare planning for cycles of tree removal and replacement, tree population modeling, and predicting urban ecosystem services accurately. Roman (2006) assessed the condition of street trees in Philadelphia, 2-10 years post planting. The study addressed trends in street tree mortality rates, variation in annual mortality over time, relationships between site conditions, and mortality and the size of the trees, and established the appropriate sampling frame to obtain an accurate account of morality, causes of street tree decline, and growth rates. She showed that the principle causes of street tree mortality and decline included poor soil, unfavorable atmospheric conditions, natural disturbances, and direct human interactions. The overall survival rate, 8-10 years post planting was 57%. Young street tree mortality is also examined in detail in Chapter 2, in New York City street trees. Better species and site choices are critical in order to avoid replanting at unpredictable intervals. We can go from reactive to proactive/adaptive management, but it is hard to practice proactive management if no information on growth or success is available. Roman (2006) has shown us that a more nutrient rich soil rather than urban fill would yield a higher success in tree survival.

Maintenance

Currently, most time is invested in the first period of a tree's life (planting) and considerably less time in the tree's overall maintenance. In 2002, the American Forestry Association and the US Forest Services conducted a follow-up study to a 1989 survey on the impact of tree size, condition, and history on overall tree health. Working from sample plots in 44 urban communities throughout Missouri the study found that communities needed to devote more resources and time to tree maintenance (Gartner 2002). With this lack of maintenance and in the absence of critical research findings, many city managers and citizen groups cite a 13-year average life span for trees planted in downtown urban settings (Moll and Gangloff 1987). While this striking number has raised awareness of the harsh growing conditions for urban trees, Moll admits that the 13-year figure has not been scientifically established and it has raised many concerns about tree mortality (Burns and Honkola 1990, Skiera and Moll 1992, Urban Forest 1992, Miller 1997, Iakovoglou et al. 2002, Johnson, 2008). Although the 13-year figure helped to raise awareness to the harsh growing conditions for life span of urban plantings, it also negatively affected the reality of urban trees.

Premature Senescence

Urban trees are often planted with an expectation of nearly zero attrition, in spite of ample experience to the contrary, thus skewing major portions of the urban canopy towards one age class. As a consequence we have little knowledge of what might be considered a reasonable life span for any number of common species, design situations, urban gradients, soil disturbances, or environmental ranges. Although it is hard to imagine a professional forester being naïve enough to believe this, urban planners with little input from foresters do much of urban management.

Sinclair and Hudler (1988) define "tree decline" as a progressive, premature loss of health, distinguished from the normal occurrence of senescence by premature debilitation. There is no current definition of premature death, however, due to the fact that there are no established definitions for normal rates of attrition and life expectancies. Although anecdotal field wisdom exists, lifespan has not been established for most tree species, particularly in the urban context.

Early studies suggest that elevated mortality and shortened life span are prevalent for urban trees, due to harsh urban conditions. Skiera and Moll (1992) have estimated an average lifespan of 13 years for downtown trees, 37 years for residential sites, 60 years for an ideal city site, and 150 years for rural sites. Anecdotal claims have engendered untested acceptance, through iterative citation within the urban forest community, to the effect that, "street trees live on average only seven to ten years" (Moll 1989, Craul 1992, Taylor 1993, NJTG 2005). This claim has been proved false by a number of studies (Foster and Blaine 1978, Polanin 1991, Nowak et al. 2004, Roman, 2006), but it persists as "street wisdom."

Sinclair and Hudler (1988) classify tree decline into four major types: chronic irritation by a single agent, damage by a secondary agent after injurious event, chronic irritation by one or more agents that reduce tolerance of other agents that then lead to a decline, and synchronous cohort senescence. Synchronous cohort senescence can be linked to early tree survival/mortality. Survival rates vary widely within the first ten years, from species to species, ranging from 34.7% to 99.7% (Roman 2006). However, little is known about the factors or significant relationships that ultimately contribute to tree mortality or survival. A systematic approach to the assessment of the demographic realities of urban tree species is necessary, with careful attention paid to the factors contributing to the urban tree life tables is needed.

Decline of Trees and Premature Mortality

While the growth of young trees is highly visible, well measured, and represents a strong industry investment, mortality is relatively unseen, insufficiently tallied, and easily overlooked, though substantial (Miller 1997, Bond 2006). The importance of estimating mortality in traditional forestry practice is well established, but the forces responsible for its prominence (light competition, nutrient limitation, pests) are much reduced in the urban context. Beyond site selection, tree growth in the wild is largely beyond practical influence for large populations since it is driven by genetic growth traits interacting with local environmental factors (Bond 2006). By contrast, early mortality in the urban context has a large human component, whose significance is often ignored in tree planning projects (Ip 1996), in spite of its importance for policy makers, planners, local tree managers, and even planters (Luley and Bond 2006). There is a physiological shock to the condition of younger trees; due to transportation and improper planting techniques, that lead to high first year mortality (Ferrini et al. 2000).

Removal/Replacement

While residents and managers value the health of an older tree canopy, there is little evidence that what is planted today will live as long as currently mature stock. Currently, although there is knowledge on how a tree would grow in its natural setting, there is limited knowledge as to how fast it will decline in urban settings. We need some information on that point, in order to inform best management practices (BMP), setting standards for tree removal. Whether a tree has been lost through natural or intentional causes, the inevitable question arises of whether or not it should be replaced at all (Cobham 1990). Different planting designs may require different replacement strategies; isolated trees may need to be treated differently than group plantings (Hannah and Yau 1993, Hitchmough 1994).

Urban Ecosystems

Urban plant communities provide many environmental services. Environmental services can be broken down into smaller categories, including, but not limited to: carbon sequestration, water quality, air quality, and cooling effects. An urban forest meets all the definitions of an ecosystem, and may even contain smaller systems within it (NUFAC 2005, i-TREE 2006). Considering the entire city an ecosystem, then the system has embedded parks that contain meadow, stream, and forest communities. There are man-made infrastructure boundaries that separate components, say a paved roadway between a small grove of trees or an open grassland of a park. But by considering urban areas as part of a broader ecological system, investigation is facilitated on how urban landscapes function and how they affect other landscape interactions. Plants in urban ecosystems experience higher temperatures, CO_2 concentrations, and more nitrogen deposition than do rural areas; all other factors are equal (Gregg et al. 2003). Urban ecosystems can no longer be considered as separate from the environment. With the increasing urbanization of the world, a greater investment needs to be made to understand these complex ecosystems that an increasing population is a part of. If urban areas are to be sustainable, a more practical method to sustainability should be invested in infrastructure sooner rather than the complete retrofitting of the area.

Amenity Values

Trees are now a regular and major component of the urban landscape due to the development of cities and urban sprawl. When considering the future management of amenity trees, particularly in their removal and replacement, it is necessary to understand the effects of trees on the urban environment and their perceived value to residents. The benefits received from the tree will depend on the particular species, life cycle, location, size, and growth rate (McPherson and Rowntree 1989). Planning requires that potential effects of the loss of trees and the subsequent decrease in benefits for residents and communities during tree removal be taken into account.

Street trees are an important factor in the perceived attractiveness of residential streets, with larger trees providing the greatest contribution (Schroeder and Cannon 1983). In an evaluation of street trees in Sacramento, one resident responded, “a city without trees is a day without the sun” (Sommer et al. 1989). Following hurricane Hugo in 1989, a survey in Charleston, South Carolina revealed that 30% of residents thought the urban forest was the most significant feature of the community damaged, slightly more than the 27% who felt that churches were more significant (Hull 1992).

Residents of urban areas desire park and street trees. Residents of Detroit felt that both street and park trees needed increased municipal funding (Getz et al. 1982). In fact when asked about the importance of different urban forest areas, tree-lined streets were rated as highest, above open parks and wooded areas (Getz et al. 1982). In the two cities of Joensuu and Salo (Finland), benefits of the urban forest relating to nature and social/outdoor activities were valued the highest, indicating there were no negative impacts of urban forests (Tyrvainen 2001).

In addition to resident surveys, other studies indicate that the perceptions of aesthetic and scenic quality of urban streets can be predicted from measurable parameters such as: trunk diameter, canopy enclosure, and trees/km (Buhyoff et al. 1984, Lien and Buhyoff 1986, Schroeder et al. 1996). There are strong indications that residents in a number of different towns and countries desire and highly value urban trees (Nowak 2004). The valued benefits of urban forests include both the physical, environmental effects, and identifiable social effects. Urban forests have much to offer the communities they serve and as such a direction towards a sustained management of a stable urban canopy is necessary.

Cost/Benefit Model of Urban Trees

In order to weigh or monetize the effectiveness of any specific intervention, a cost-benefit analysis, typically used in economics and business, can be employed to evaluate the desirability of consequences. The aim of a cost-benefit analysis is to gauge the efficacy of the intervention relative to the status quo. Typically, the costs and benefits of the impacts of an intervention are evaluated in terms of the willingness to pay for them (benefits) or the willingness to pay to avoid them (costs). This very limited definition provides a framework for the discussion of a cost/benefit analysis. As a system has increasing outputs and complexity, the dialogue shifts to include values and perceptions. In its most realistic form, a cost/benefit model includes the possibility for inaction as well as waiting for further development. Inputs are typically measured in terms of opportunity costs: the value of the best alternative allocation. The guiding principle behind cost-benefit analysis is to list all the parties affected by an intervention and place a monetary value of the effect it has on their welfare, as they would value it. The entire process involves the monetary value of initial and ongoing expenses versus expected returns. For cost-benefit analysis, monetary values may be assigned less tangible affects, such as the various risks, which could contribute to partial or total project failure.

It is often difficult to assign monetary values for less tangible effects. For example: when examining the value of a dog, one would have a hard time putting an exact monetary value on companionship or protection. There are many possible factors that could go into the value of a pet: is there one individual value for a pet, or does it depend on age, socioeconomic factors, and social status? The difficulty of assessing the value of the life of a tree is similarly complex, depending on age and circumstances.

There are various accuracy problems in a cost-benefit analysis that should be acknowledged. The accuracy of the outcome is, for one, completely dependent on how accurately costs and benefits have been estimated. A second challenge to the analysis comes from determining which costs should be included in the analysis (a min-max strategy). These problems aside however, cost-benefit analysis can be very useful when determining the value of a tree.

If a tree is seen as an asset, then a dollar amount can be fixed for the cost of planting and maintaining that tree, as well as the benefits the tree provides. However, as opposed to a typical manufacturing product, whose value depreciates as time passes (Figure 1), trees gain overall amenity value as time passes (Figure 2). The idea is that the costs of establishing a tree in an urban area are extremely high; they include costs of planning, initial planting, and maintenance. The benefits received in the establishment period are fairly low (potentially zero), but could be very high politically. As time progresses, post establishment, the cost of a tree decreases (less maintenance) and the benefits increase. There comes a time, more accurately a size when the tree becomes over-mature, beyond which point the cost of further presence mounts steadily. The potential for damage to human life or human property increases if the tree or limbs fall due to decay or other factors. When the marginal costs of leaving the tree in place begin to outweigh the amenity benefits, it is time to harvest the tree and (possibly) replant the site.

The work of Hitchmough (1994), Cobham (1990), Hannah, and Yau (1993) all suggest that as a practical matter, a time arrives when the benefits of the tree no longer outweigh the costs of further maintaining that tree and the tree should be removed. When looking at the hypothetical model, Figure 2, the return of a tree increases through establishment, reaching a plateau at maturity, before decreasing through senescence. The benefits/outputs can be measured as any perceived benefit or the collection of all benefits. The second component of Figure 2 shows the management cost relative to age for the same tree. After initial high costs of planting and establishment, the costs are low. Costs increase again as the tree ages and require increasing arboricultural inputs. A tree should be removed at the point in decline phase where the lines cross, as it is difficult to justify increasing expenditure for decreasing benefits. Furthermore, if a tree is maintained for another 20-30 years past this point, those maintenance costs are accrued with decreasing amenity rather than on a replacement tree that increases amenity levels to those previously attained by the tree removed. However, determining where an actual tree fits this model is far from simple. Furthermore, different tree species will require different maintenance inputs and require removal at different times (McIntyre 1994). A further complication in determining when or whether the costs outweigh the benefits is that the life span of urban trees is not known with any certainty (Pescott 1968, Moore 1990). This uncertainty is reflected in the time frames used in tree replacement models.

Costs

There are many major costs in the urban forest. Some of these major costs include: planting, maintenance, and the costs associated with building infrastructure conflict, repairing or preventing power line breaches and sidewalk damage, and reduced visibility (D'Amato 2001, Gorman 2004). All inhabitants of the urban community may not perceive all costs, and the costs of the trees may be evaluated and appreciated differently between residents and managerial units within a municipality; there are always inherent costs.

Several studies have examined residents' feelings toward urban trees. A Chicago study more specifically detailed street trees perceived by residents. The main annoyance of street trees was leaves falling in autumn (Schroeder and Ruffolo 1996). Other less notable annoyances included: roots too close to the surface and pavement cracking, both of which were considered to be minor to moderate annoyances.

Quantifiable/Unquantifiable Benefits

Trees in the urban forest make a strong contribution to the quality of life and provide significant outdoor leisure and recreational opportunities for urban residents (Dwyer et al. 1992). Chicago metropolitan area residents described the important benefits of street trees, stating that they were "pleasing to the eye" and that it "enhances the look of my yard and house" (Schroeder and Rufflo 1996 p. 30). "Bringing nature closer" was also a perceived benefit. Eye pleasing also rated the highest in a Detroit survey, followed by shade, increased property values, and autumn color (Getz et al. 1982). A recent study by Lohr et al. (2004) ranked "shade and cool surroundings" as the highest reason to have trees in cities.

There are many perceived benefits of urban trees; some more easily monetized then others. Urban trees aid in carbon sequestration, reduce storm water runoff and increase soil infiltration, and provide shade and counteract urban heat island effects (Coder 1996, Botkin and Beveridge 1997, McPherson 2000). Additionally, urban trees can contribute to the physical, psychological, and mental well-being of urban residents and workers, with reduced stress attributed to the presence of urban trees (Dwyer et al. 1992). It has also been suggested that hospital patients with window views of trees recover faster, with fewer complications, than patients without such views (Ulrich 1984). Dwyer et al. (1992) indicated that urban forests provide increased enjoyment of everyday life and a deeper connection between people and the natural environment. Barro et al. (1997) asked Chicago residents to report examples of "big" trees. In addition to the physical dimensions requested, they also received numerous descriptive notes and letters about various aspects of the trees, including comments relating to physical size, and aesthetic and functional values. These emotional and psychological connections felt with the urban forest are often difficult to quantify.

A significant impact of trees to the environment occurs through the mitigation of climatic effects (McPherson et al. 1995, Botkin and Beveridge 1997). For example, trees in urban landscapes moderate the heat island effect imposed by the absorption of solar radiation by pavement on streets and parking lots and buildings (Whitlow and Bassuk 1987, Whitlow and Bassuk 1988). Trees provide shade that also reflects solar radiation from paved surfaces, reducing temperatures in city centers by 3 – 5^0 C (USDA Forest Service Northeastern Area). This effect is achieved through evapotranspiration of the leaves, causing the tree to act as a natural air conditioner, by creating a reduction in the immediate air temperature and alleviating the impact of the heat island effect (Botkin and Beveridge 1997, USDA Forest Service Southern Region).

By modifying the urban environment, trees can reduce the cost of heating and cooling buildings, commonly known as the urban cooling effect. A study in

Sacramento, California demonstrated the urban forest saved approximately 157 giga-watt hours (GWh) annually on cooling which saved $18.5 million (Simpson 1998) in costs. The net effects on heating were 40.2 GWh which saved $1.3 million annually in costs. Akbari et al. (1998) estimates that 100 million mature trees in US cities could reduce annual energy use by 30,000 GWh, which would save $2 billion per year. These mature trees could also avoid the planning and construction of new power stations to supply the city, which would result in further savings.

A further benefit of urban trees is pollution control and improved air quality. As carbon dioxide in the atmosphere rises, environmental temperature increases. Trees act as a sink for this $C0_2$, storing the fixed carbon as biomass (McPherson et al. 1995, Nowak and Crane 2002). Based on field data from 10 cities in the US and national urban tree cover data, urban trees store 770 million tons of carbon currently, with an annual sequestration rate of 22.8 million tons per year, equal to the US population emissions over 5 days (Nowak 2002).

Problem Statement

Urban forestry is a practical discipline in which the collective whole is more important than a single tree. The urban environment presents arboricultural and management challenges, but is influenced by social perceptions. Social issues and perception can severely impact the support and funding, as well as the legal frame and policies, for maintaining trees on public and private land. Tree populations in urban landscapes are exposed to elements much different from those in the wild, however, and it is important that trees are managed accordingly.

With demographic shifts across the globe causing rapid urbanization, especially in the developing world, urban development is likely to continue as a focal point of environmental and social issues for the foreseeable future. Urban forestry has a potentially important role to play in meeting the needs of urban populations, while also addressing the social and environmental problems that arise from urbanization. In a traditional forestry context, research has guided routine management decisions for many decades, but because urban forestry is a relatively new discipline, management decisions are often made in the absence of prior research in the urban context. Practical applications have dictated the direction of what research has been done in the field, but lacking fundamental information on survival rates, pest rates, and age structure of the forest, there is very little solid underpinning for current management techniques that allow us to judge efficacy.

Aim of Research

The research aim of these studies was to determine site typologies and median species size response to maximize the benefits received from the urban forest, while limiting the amount of time over which costs prevail. Also, we will attempt to examine and improve the current techniques of urban forest management, while within an organizational rubric. The research explores on the overall current state of the urban forest in the New York City metropolitan area, focusing specifically on how using the organization of the cost/benefit links to other infrastructure and imposes traditional forest measurements in an urban setting.

This research provides an organizational rubric to be employed to achieve maximum terminal size and determines a stocking plan scenario.

Research Approach

The studies that follow inform different aspects of a modified cost-benefit curve. Chapter 2 details a study of early urban tree transplant survival. The research is discussed as an application, within the context of the early stages of the suggested cost/benefit curve perspective. At initial installation, trees are planted and the municipality incurs the planning, purchasing, and installation costs with the anticipation of a specific benefit(s) at some point in the future. This early transplant survival study improves our knowledge about the urban and community forest ecosystem and the dynamic nature of tree mortality and survival, by examining an unprecedented number of trees in a wide variety of growing conditions, using a multi-disciplinary approach. This study reviews the aspects of traditional forestry that can be slightly modified for use in the urban setting and provides clear expectations of factors that influence early transplant survival. Chapter 2 also introduces Bayesian analysis, as a way of interpreting large data sets that were not originally designed for research.

Chapter 3 focuses specifically on the middle of the cost benefit curve and examines trees that are approximately 20 years post-transplant. These trees are in different soil volumes, and this study examines their growth as a reflection of their potential, according to soil access. Little knowledge is currently available about diameter, height, and canopy spread in the urban context. This chapter serves as a way of assessing growth in the urban forest, 20 years post-transplant.

Chapter 4 examines maximum terminal DBH in urban trees, and focuses on the intersection end of the curves to determine a point for optimum removal. This study examines mature and over-mature trees in central and northern New Jersey in order to develop a cropping rotation of mature trees at senescence when they reach a point of decline in the urban forest. No previous work has examined the DBH expectations of trees in the urban forest. This study outlines growth expectations that could be used to inform current models that evaluate ecosystem services.

Chapter 5 synthesizes the studies within the context of the overall urban forest, not just the US Northeast. Emphasis is placed on the cycle of the urban forest, from the site indices for species and stand density, early transplant survival rate, and the removal of species to preserve the form and function of the urban forest. This chapter further discusses the research results in relation to previous findings and offers recommendations for future research while providing an overall conclusion.

This research advances the understanding of this central issue in three areas characterized by the following questions:

- What is the current survival rate of transplanted trees in the urban forest, both at two years and eight-nine years post planting? What factors affect the success of these trees? If diversity is planned for in the urban forest, is it still maintained after eight-nine years? (Chapter 2)

- What is the growth expectation of trees 20 years post planting. Is there a canopy reduction, based on amount of apparent available soil? (Chapter 3)

What is the maximum terminal size expectation of urban trees in different apparent available soil? (Chapter 4)

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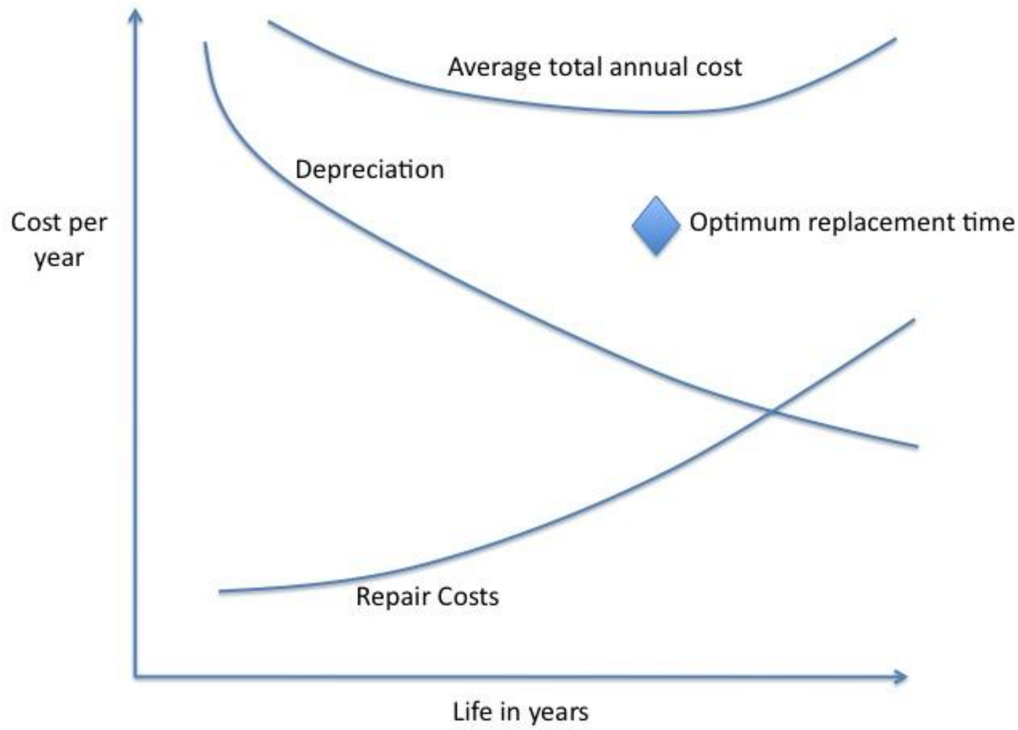

Figure 1: A model for replacing machinery based on depreciation and repair costs. (Adapted from Fraken and Wallen 1971)

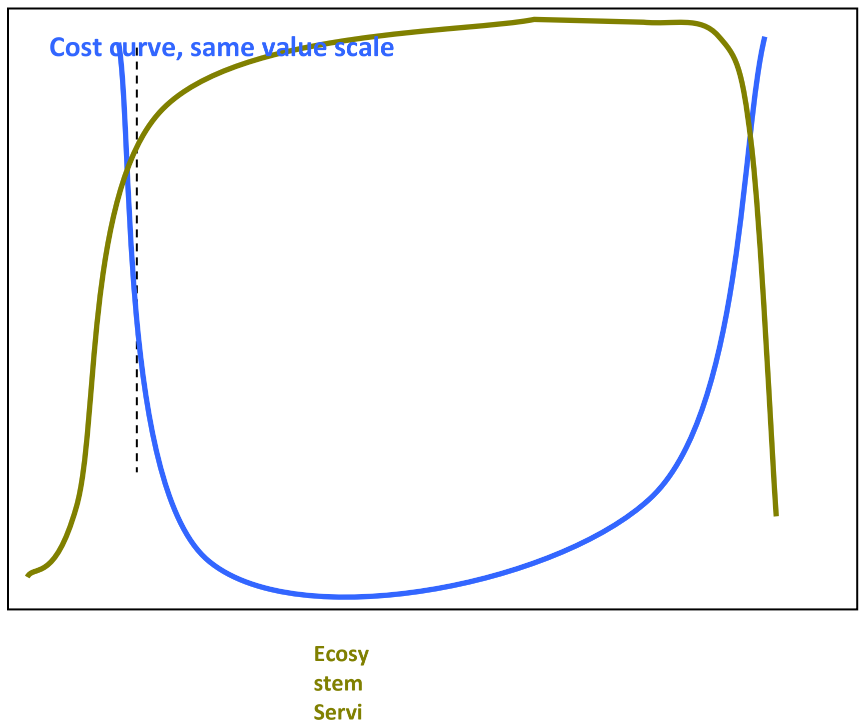

Figure 2: A graph showing the hypothetical relationship between time, management costs, and ecosystem services

CHAPTER 2

Factors influencing Early Survival of New York City Trees

ABSTRACT

A young tree survival study was developed to analyze the success of owner-requested trees planted in New York City. Data were collected throughout the five boroughs of New York City, with representatives from all planting typologies (tree pit, planting strip, unlimited soil, lawn plantings) used in northeastern United States urban forests. The focus of this study was to better understand how biological and urban design factors affect survival rates of newly planted urban trees in their first decade of establishment. Analysis of contract verification records for 43,500 second year inspections of trees planted between 1999 and 2003 was examined to study the factors that affected the urban forest *as a whole*, as well as on a borough or management unit basis. Overall, second year survival, averaged over species, was 91.4% survival. Additional field data were collected (2006-2007) from a subset of 13,405 trees. Land use, planting site type, and borough all had significant effects on survivorship. The survival of the entire original cohort to age 8-9 had declined to 75.8%. Urban forest managers will benefit from the results in their ability to optimize survival for their planting stock.

INTRODUCTION

Urban trees provide ecosystem services such as improved air and water quality, carbon sequestration, improved energy efficiencies, aesthetic values, and social and economic gain (Dwyer et al. 1992; Schroeder and Ruffolo 1996; Botkin and Beveridge 1997; Miller 1997; Nowak and Crane 2001; Nowak and Dwyer 2007; Tzoulas et al. 2007). The capital and management investments required are quite substantial, however, and the benefits have to be rationalized relative to the costs, in order to justify planting expenditures and motivate the public toward investment for environmental quality (Nowak et al. 2000; Nowak and Crane 2002; Nowak et al. 2002). The many "million tree" planting programs in the United States are programs designed to achieve larger environmental planning objectives (Million trees LA 2006; PlaNYC 2010; 4H million trees 2010; Million trees Miami 2010).

In order to ensure success, questions of stocking age (age at transplant), species composition, species selection, and transplant survival must be considered. Earlier observations of the general urban forest have suggested low average life spans (7 years) in street tree plantings (Moll 1989; Kielbaso 1989). The abundance of such studies (Burns and Honkola, 1990; Skiera and Moll 1992; Nilsson and Randrup 1997; Johnson 2008) has led some practitioners to expect a 7-year life span for urban trees. A more recent meta-analysis, based on empirical data from 16 different studies, has shown that estimated mean life expectancy for street trees ranged from 19-28 years (Roman and Scatena 2011). A Sacramento, California shade tree program reported that 43% of trees remained alive after 10 years (Lindeleaf 2007). The average annual street tree survival was 90% in Gainesville Florida (Lawrence et al. 2012), 93.4% in Baltimore (Nowak et al. 2004), and 95.3% in Houston (Staudhammer et al. 2011), so survival was considerably greater than the figures and popularized expectations attributed to Moll (1989).

Most previous studies on tree plantings have had limited sample sizes. A Boston, MA study examined socio-economic and demographic factors for 136 trees that were planted on one street (Foster and Blaine 1978). A study in Oakland, CA observed street tree growth and survival of 480 plantings along a 5.4-mile boulevard (Nowak et al. 1990). A NYC study reported environmental factors that influenced 1,000 street trees (Berrang et al. 1985). The largest sample size previously used was 10,000 newly planted street trees in northern New England, a study focused on factors affecting survival after one year (Gilbertson and Bradshaw 1985).

Apart from the normal losses of urban canopy due to abiotic site and environmental limitation, there are non-trivial concerns about native and exotic pests and diseases. Santamour (1990) proposed a target stocking balance for municipal management units, 10% of any single tree species, no more than 20% of any tree genus, and no more than 30% from any one family. A stocking balance is commonly used as protection against unanticipated introductions of invasive pests such as Asian Longhorn Beetle or Emerald Ash Borer, largely in reaction to massive municipal tree canopy losses to Dutch Elm Disease. This practice also avoids the use of a monoculture of the "perfect urban tree", because the urban setting is best viewed (and treated) as a series of heterogeneous microclimates (Bassuk 1990). Although there are other techniques to manage the diversity of the urban forest, and Santamour's rule has been criticized as overly simplistic (Lacan and McBride 2008), stocking balance has been used extensively in the Northeast for municipal tree planting programs.

We present two components of a larger study (Lu et al. 2010) to investigate species diversity and early mortality within the NYCPRD. The goals of this study are to answer questions concerning traditional conceptions of premature death of young trees within the urban context (7 year average lifespan) and to determine the causes or risk factors for premature urban tree mortality, while examining the stocking balance of the urban forest. Risk factors were assessed for newly planted street trees, both across NYC as a whole and within (borough) management units. The specific mortality issues addressed included: The survival rate after the two inspections, as well as the identification of factors associated with mortality in this data set. We also addressed the question of whether the initially planned diversity had survived the attrition.

MATERIALS AND METHODS

Tree planting and inspection database

The New York City Department of Parks & Recreation (NYCDPR) assembled a data set of 43,500 trees, planted by owner request between 1999 and 2003 throughout the five boroughs of NYC. Data records were collected from a representative cross section of all planting typologies in the Northeast Urban forest. Site types (such as lawn, sidewalk opening, and planting strip) represented all commonly occurring planting situations. The database contained planting information and second year inspection data, to verify contract compliance. Planting sites were characterized as containing live trees, dead trees, or as 'tree missing' during inspections. Species frequencies recorded in the contract inspection inventory database were compared with the Santamour (1990) diversity recommendations to determine whether planned diversity was maintained after planting losses. All contract verifications were recorded at two years post planting (2 year survivorship data). This study does not follow a specific cohort, but rather a group of cohorts that are bundled together for analysis, in order to evaluate the effectiveness of the tree-planting program.

Managerial Units

This study broke the urban forest into boroughs as logical managerial units, reinforced by the organizational structure of the NCYPRD. The impacts and history of construction events develop some similar soil situation types in heavily urbanized sites such as New York City. Additionally, the underlying geologic patterns, separation of boroughs as islands, and different characteristic land use patterns between boroughs suggested we should treat the obvious borders in our development of a re-sampling plan.

Diversity Composition

Descriptive statistics for occurrence of species (population counts, percent of total) were developed to examine the stocking diversity at the species, genus, and family levels, relative to Santamour's (1990) 10-20-30 recommendations. We analyzed data from the entire city and from each of the five boroughs separately. Nested plot analysis treated the five boroughs as subsets, while the entire city served as the meta-level. Survival was estimated as the percentage of total count planted for each species, and the numbers were aggregated across species and then examined by borough and by year planted, in order to ensure the logic in a sampling stratification strategy in the second part of the study effort.

Species of Interest

A data set representing 109 species of planted trees was available. We focused on those species with total counts greater than 50 individuals. Of those species, we identified those that exhibited less than 90% survival for NYC reevaluation, in planning future projects, arbitrarily set as an acceptable survival level of planting success. When computing overall survival, either across the city or within any particular borough, all trees were used for aggregate survival statistics. Six species (*Gleditsia tricanthos, Prunus virgianana, Pyrus calleryana, Tilia cordata,Tilia tomentosa, Zelkova serata*) comprising 51% of the city-wide data, and were considered to be important enough components of the planting mix to warrant calculation of species-specific survival data.

Stratified subsample of data

The second part of the study utilized a stratified sample from the 2-year dataset to establish a longer trajectory of installation mortality. Site parameters were used as correlated variables that might usefully inform future stocking decisions, and were evaluated for their predicted efficacy in forecasting survival. Given knowledge of the highly uneven species distributions among the five boroughs from the first phase of the study, a stratified subsample of 14,090 trees (32%) was chosen, providing an opportunity to assess critical variables such as borough, time in-ground, and land use (Sun and Bassuk 1991; Jaenson et al. 1992). Information on 57 variables was collected, providing detailed information on four general types: tree condition, municipal aspects (hardscape features, landuse, and borough of planting), planting site considerations (planting environment, tree species, and survival rate), and social effects (tree ownership/caretaking and socio-economic factors) (Lu et al. 2010). Social effects were not examined in depth for this paper (Lu personal communication). There were multiple instances where the tree in the phase two sampling did not match expectations from the phase one inspections. For example, inspectors would find a 40 cm tree in a 5 year location or have some uncertainty in the location or positive previous identification (no planting space for a tree or much larger tree then could have grown in the time period). All such individuals were eliminated from analysis, reducing the final sample to 13,405. The discrepancies suggest a 4.9% sampling error in the larger (2 year inspections) data set. Survivorship statistics were developed for the second data collection to ensure that the composition of the data was preserved, enabling a direct comparison of species diversity balance over time.

Bayesian Analysis

To examine factors associated with tree mortality, a Bayesian analysis approach was used, since the data did not meet assumptions for a traditional frequentist statistical approach. This planting program was respondent to owners request, so the trees are not a random sample from the urban forest. The initial dataset (1999-2003) was used as a prior distribution to inform our second phase, by virtue of its large sample size and explicit connection to the second data set. Bayesian analysis was employed; observations from an early (the 2nd year analysis?) phase of the study were used to develop a hypothetical survival model, and later observations were used to calculate the probability that our model was likely to be true. For ease of comparison to traditional statistical output, a Bayesian Score Statistic is reported and considered to be an analog to the *p* value output of traditional statistics and as such can be interpreted with the same significance value (Kleibergen et al. 2000). A BSS score 0.05 or less is considered significant.

The Bayesian model was used to conduct a focused assessment on survey categories that potentially affected tree mortality. Data were analyzed using the WinBugs package in the R software (R Project 2010) MCMC (Markov chain Monte Carlo), simulations were run with 120,000 iterations. Analysis on the data was conducted on the 100,000 iterations, after trimming the first and last 10,000 runs. All data were analyzed as single components, as well as in a multivariate model, to test for confounding and interaction factors.

Missing Trees

There were trees in the second phase that were missing in the first phase; the data collectors were at a correct location, but there was an empty pit. A missing tree could indicate that the tree had possibly been vandalized or deliberately removed, because it was unwanted by the local community members (as opposed to the owner who requested it), whereas a standing dead tree could imply the tree had failed, due to biological or environmental factors. For example, if a tree died in year 1, and was then removed, then no inference or explanation can be determined. These trees were considered to be "dead" for the purpose of our analysis. Seventyfour percent of the surveyed were still alive and standing in the 2nd round; 6% were dead but present, and 20% were missing. Analyses were run with and without the missing trees, to determine whether the patterns emerging from the results were sensitive to the difference. There is no statistical significance between the factors associated with missing and dead trees (BSS = 0.95), so both were treated as dead for ease of presentation.

RESULTS

Phase I Survey

In the first part of the study, species diversity generally conformed to the 10-20-30 program target. *Gledistia tricanthos* (GLTR) and *Pyrus callaryana* (PYCA) were over-represented as components of the total tree species inventories (11.5% and 16.3% respectively). Representation was <20% for each genus. The Rosaceae (31.1%) are over-represented as a family, relative to the 30%, but in general, the plantings are reasonably compatible with the Santamour (1990) recommendations (Table 1). Survival at the two-year post-planting inspections over the entire dataset was 91.5%.

Counts of listed taxa varied by borough, from 50 in the Bronx to 99 in Queens (Table 2). Breaking down the data by borough and year, survival levels in 1999 inspections were lower across the entire study area (Table 3). Brooklyn had consistently lower second year survival rates in the first four years (data not shown). When looking at the species codes representing greater than 50 trees each, several oaks, particularly *Quercus phellos* (196 trees; 71.4% survival) and *Quercus rubra* (527 trees; 83.1% survival) exhibited lower survival rates compared with the total test population. *Tilia tomentosa* (2850 trees; 89.2% survival) was the only major species in this dataset that did not meet the arbitrarily assigned 90% threshold survival level from the time of transplant, with anecdotal reports of seasonal transplanting sensitivity. Both deciduous conifers, *Taxodium distichum* (68 trees; 88.2% survival) and *Metasequoia glyptostroboides* (227 trees; 78.9% survival) survived poorly, in spite of suggestions of their desirability for use in urban landscapes (Gerhold et al. 1993;Bassuk et al. 2009).

Phase 2 Survey

At the end of Phase 2 survey, there was still a balanced stocking design, relative to the program target 10-20-30 ratio. Of all survey variables, only a limited number affected survival of the trees in the dataset. Although all the categories were examined using the MCMC, there were numerous categories that showed no conclusive effect on the overall mortality of the entire urban forest.

There were some minor inconsistencies in the stocking balance among boroughs, but the general stocking diversity was stable over the nine-year period. As in the Phase 1 survey, trees that encompassed greater than 5% of the initial stocking was documented (Table 4).

At the end of phase two, overall survival was 74.3% across the entire city. Total survival did vary among boroughs (Table 4), however, with 62.5% (Bronx), 70.8% (Staten Island), 74.8% (Manhattan), 76.4% (Queens), and 79.6% (Brooklyn). When considering the entire city, but stratifying by number of years post planting, survival was 91.5% after 2 years, 78.2% 3-6 years post planting, 73.0% 7-8 years post planting and 72.8% after 9 years.

Factors Affecting Survival

A city-wide Bayesian analysis demonstrated the factors that had a demonstrable impact on survival (Table 5). The model did not improve with interaction effects, except in the case of Traffic volume, when associated with land use, had a negative effect on survival (BSS 0.017). Residential land use yielded higher survival (82.7%), than was typical of open space and vacant land (60.3%). Metal tree trunk guards, installed at planting, were still found on a majority of trees at the date of the second inspection, but their presence or absence had no effect on survival (BSS = 0.47). Infrastructure conflict, scored as a yes/no category, covered a number of possible conflicts, and appears to have a borderline significance on tree survival (BSS = 0.067). Not all infrastructure conflicts were as detailed as needed for the purpose of this analysis, but the impact of such conflicts should be examined further in future studies. As available soil in the planting pit decreased, survival decreased (BSS = 0.020), 70.7% survival in 16.8 m^2 decreasing to 67.3% survival in 7.6 m^2. Trees planted in lawn settings had a survival rate of 78.1%; those planted in sidewalk vegetation strips, with intermediate soil amounts, had a survival rate of 72.9%; trees planted in extremely limited sidewalk cut outs had a rate of only 67.3%.

DISCUSSION

Previously published urban tree mortality studies, although much smaller in sample size, reached similar conclusions. A 26% mortality rate on urban street trees (over a 2-4 year period) Boston, MA (Foster and Blaine 1978), as compared to 37.1% mortality on trees planted 8-9 years in the current study. Nowak et al. (1990) reported a higher mortality rate after two years (34%) in an Oakland, CA, but could not attribute the cause of death to any specific factor. A previous study in NYC (Berrang et al. 1985) focused on environmental factors affecting survival of urban trees in the vicinity of electric power facilities, but it is difficult to extrapolate his study across the urban gradient, due to the role the land use may have. In the Berrang study, land use was not interpreted to the same standard of the current study. The largest sample size to date, following 10,000 street trees in northern England, found 9.7% mortality after one year , , and while the researchers drew attention to various potential factors, they made no not attempt to link any specific factor to the mortality rate (Gilbertson and Bradshaw 1985).

Although no mean life expectancy has yet been calculated for the current study, the population half-life (survivorship = 50%) has not yet been reached. The observed mortality rates, based on half-life, exceed lead one to expect greater than seven years from transplant (Moll 1987). Overall, survival rate can be related to borough, as a first-level geographic parameter. However, there may be small groups of species, which can impact these numbers. For example, *Juniperus virginiana* was planted in the relatively diverse species code zone of Staten Island, with 0% survival, but was not listed elsewhere. The potential to skew the data set for the smallest community population from one planting event was evident. When looking at lower survival rates among *Quercus spp.*, it seems obvious that *Quercus* *phellos* may be negatively impacted, due to planting into difficult environments, when installed beyond the northern limit of its natural range. A few problematic species (*Juniperus virginiana, Quercus phellos, Metasequoia glyptostroboides)* had very low survival rates. Those species tended to be planted infrequently, and therefore had minimal overall effect on the urban forest canopy.

There was a correlation between land use and borough, which probably explains the borough effects. Boroughs with high traffic volumes and large quantities of open space/vacant land use had lower survival rates, relative to those with more residential land and lower traffic volume. Tree survival was highest in Queens and lowest in the Bronx, which could be due to Queen's low population density and suburban land use pattern. The two commonly planted species *Gledistia tricanthos* (GLTR) and *Pyrus calleryana* (PYCA) have average survival rates across all the boroughs of 93.4% and 94.0%, respectively, but the other species that encompass more than 5% of the data have a slightly lower survival rate of 92.4%. In relation to tree survival, none of the most commonly planted species stand out as being problematic; which was not surprising. The most common species, including those considered over-used (Richards 1979), are chosen for their tolerance and reliability in urban environments, which causes preferential use in urban planting plans. Among plant family groupings, there is a suggestion that *Rosaceae* are slightly over-represented, but the balance in all levels is very similar to the stocking rules many programs have used for over 20 years (Richards 1979; Santamour, 1990; Miller 1997). When examining specific species, care must be employed. For example, in 1999 Brooklyn plantings there was a low survival in *Ginkgo biloba,* nearly half of the 153 trees. Over the next 4 planting years, there was only 178 *G. biloba* planted, which caused an extreme skew in overall survival.

According to Santamour's recommendation, the urban canopy that was surveyed in NYC has a stocking balance. This is an advantage in New York City, which is a potential entry point to many biological pests that have the potential to cause great disaster on the city's urban forest population.

It was observed that in the industrial setting, only certain species have high survival rates. We would recommend consideration of environmental tolerances for targeted selections, to be used sparingly throughout the rest of the urban forest, where other species may do just as well. Use “proven performers” for planting in the more challenging overall harsher areas (high traffic, high construction, etc.). The survival rate between cohorts (year of planting groups) stabilized between age groups. Although we do not yet know the end of the curve and will not for decades, the mortality curve is suggestive of a U shaped or bathtub mortality curve postulated to be characteristic of tree communities (Harcombe 1987). Although Harcombe (1987) never directly spoke of urban tree communities, and spoke rather of mortality tables in natural communities, a connection can be made from natural systems to a designed urban system. A type III curve (U-shaped) would show an early mortality that would level out as time progresses that is characteristic of our findings. Very small trees in the establishment phase typically have a lower survival rate (Richards 1979; Miller and Miller 1991). We suggest that survival rate has leveled until the next development event or infrastructure transition to examine when the survivorship rate decreases.

Overall, residential areas of low population density have the highest street tree survival rates, while industrial, open space (undeveloped property), and vacant lands have the lowest survival rates. This makes inherent sense, given that these plantings were generated by requests from owners. One/two family residential areas have the highest survival rate of 82.7%. When examining residential areas with a higher occupancy rates, such as a high rises, survival rate is still higher than for other land use categories, at 72.3%. In areas with a lower residential population density, the trees appear to be better maintained, with less damaged noted, and this will be investigated in a later paper. When examining mixed land uses (mixed, commercial, and public institutions), there is a 62.9% survival rate across the entire city. The industrial land use category exhibits a mortality rate of 66.2%, but if the industrial areas were planted with the trees that survived in that environment, with elevated diversity in other land use areas, overall survival rate across the city would increase. The landuse with the lowest survival in this dataset was open space and vacant land (61.3%). A reasonable explanation for this could be the lack of planted tree ownership in vacant lots or on undeveloped property. The low survival rate could also be attributed to the increasing development of these areas, relative to that in other areas in the city. As infrastructure changes around trees, they may be removed so that the area can be revitalized or developed. The industrial sites could also have soil of an inferior quality, when compared to the residential soil.

Although traffic volume was not significant on its own, when combined with land use, the two had a significant additive effect (BSS = 0.017). When there was a higher traffic volume, combined with a higher population density, there was a lower survival rate overall. The additive effect indicates greater flow of people into an area, and more people living in the area, so there is a lower survival rate.

As the available soil volume increases, survival rate increases. Trees in an extremely limited soil situation have a survival rate of 67.3%; while trees with more than adequate soil amounts (lawn settings) (Grabosky and Gilman 2004) have a survival rate of 78.1%. Trees placed in sidewalk strips, which is the medium amount of available soil, have a survival rate of 72.9%. This suggests that soil surface could be used as a predictor of early survival.

LIMITATIONS OF THE STUDY

The trimming of the data set to 13,405 trees, accounted for 4.5% sampling error that could have occurred. This collection error does not negate the overall findings, as the analysis was performed with the missing trees, as well as without them, and yielded the same results. Asynchronous cohorts over time limit the data interpretation; performance within the time frame of this study does not predict performance at later time frames. Lumping of different years of observation (8-9 rather than year 8 and year 9) limits how the data can be interpreted. Binning different age cohorts is not the same as following trees over multiple observations, so while survivorship curves over time are suggestive, they are not definitive and are limited to the first decade post-transplant. This dataset nevertheless provides a consistent baseline suggesting consistent survival patterns. The next infrastructure cycle is now in place, and should provide a vantage point from which to examine the survival and mortality patterns at a more advanced age.

CONCLUSION

The life expectancy for street trees held as a popularized belief attributed to G. Moll of 7 years (1989) is too short. The New York City data set shows survival rates at 8-9 years post planting that are substantially higher than 50%. This study does follow the same survival structure as found in Moll's (1989) study, with trees having the lowest survival rate in downtown areas and an increasing survival as one moves into residential and the best city sites but the average age at which the trees will die is elevated. This study encompasses one of the most diverse urban areas, marked by a high rate of change. We now have empirical data demonstrating the level of post-planting survival is not necessarily as dismal as frequently suggested.

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Table 1: Species, genera, and family levels in New York City (NYC) study area, which comprise over five percent of the total study population (trees planted on request by contract from NYC DPR) 1995-2006.

Species Code	population %	Survival %	Genus	Population %	Family	population %
			Acer	6.7	Aceraceae	7.2
GLTR[1]	11.5	93.4	Gleditsia	10.8	Fagaceae	9.6
PRVI[2]	5.1	93.2	Prunus	11.9	Leguminosae	13.7
PYCA[3]	16.3	92.7	Pyrus	18.0		
			Quercus	9.5	Oleaceae	5.0
					Rosaceae	31.1
TICO[4]	5.9	90.9	Tilia	12.8	Tiliaceae	12.7
TITO[5]	6.8	89.2				
ZESE[6]	5.2	92.1	Zelkova	5.1	Ulmaceae	5.1
Average survival		92.1%				

[1] GLTR: *Gleditsia triacanthos*
[2] PRVI: *Prunus virginiana*
[3] PYCA: *Pyrus calleryana*
[4] TICO: *Tilia cordata*
[5] TITO: *Tilia tomentosa*
[6] ZESE:*Zelkova serrata*

Table 2: Species counts and survival rates developed in contract document for 2nd year inspections (1997-1999) and End (2006-2007) inspections

Borough Code	2nd Year Taxa Count	2nd Year Tree Count	2nd Year Number Dead	2nd Year Percent Survival	End Tree Count	End Percent Survival
Bronx	53	4409	510	88.4	1,994	62.5
Brooklyn	74	6113	838	86.3	3,476	79.6
Manhattan	70	10,946	888	91.9	1,631	74.5
Queens	99	17,149	1016	94.1	4,737	76.4
Staten Island	80	6,321	653	89.7	1,567	70.8
City Wide				91.5		74.3

Table 3: 2nd Year (1997-1999) Inspection counts of NYC Street Trees installed at owner request. By borough and entire city. % occurrence % survival at 2nd year inspection. (Numbers are rounded to the nearest whole tree)

	Brooklyn		Bronx		Manhattan		Queens		Staten Island	
Acer campestre	5%	89%	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Acer rubrum	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	9%	95%
Ginkgo bilboa	6 %	71%	n/a	n/a	6 %	88%	n/a	n/a	n/a	n/a
Gleditsia tricanthos	11%	89%	13%	91%	16%	93%	11%	96%	5%	94%
Prunus virgianana	n/a	n/a	n/a	n/a	5%	91%	7%	96%	n/a	n/a
Pyrus calleryana	18%	91%	12%	89%	14%	92%	17%	94%	19%	94%
Quercus palustris	n/a	n/a	7%	91%	9%	96%	5%	94%	n/a	n/a
Tilia cordata	6%	85%	7%	86%	n/a	n/a	6%	95%	n/a	n/a
Tilia tomentosa	n/a	n/a	5%	80%	8%	87%	9%	93%	n/a	n/a
Zelkova serata	n/a	n/a	11%	95%	7%	94%	n/a	n/a	n/a	n/a
Total Survivorship	n/a	86%	n/a	88%	n/a	92%	n/a	94%	n/a	89%

Table 4: Phase 2 (2006-2007): New York City Young Street Tree Survival by land use

Land Use Category	Alive		Dead	
	# Trees	%	# Trees	%
Residential				
One/Two Family	4,821	82.7	1,009	17.3
Multi-Family	2,232	72.3	856	27.7
Mixed				
Commercial/public	388	62.9	229	37.1
Industrial/Utility/Parking	1,903	66.2	972	33.8
Open Space/Vacant Land	545	60.3	359	38.7

Table 5: Bayesian Analysis Questions Asked (of trees planted by contract at owner request)

Question Asked	**Effect on Survivorship**	**BSS**
Does landuse affect survival?	Yes	0.015
Does the tree guard affect survival?	No effect	0.047
Does infrastructure conflict affect survival?	Not conclusive	0.067
Does traffic volume affect survival?	When associated with landuse, the two have an additive value.	0.017
Does the pit type affect survival?	Yes	0.020
Does the presence of stakes affect survival?	Yes	0.021
Does the presence of irrigation bags affect survival?	No	0.57
Does pruning type survival?	No	0.55
Does borough affect survival?	Yes	0.018
Does land use affect survival?	Yes	0.014
Is there still the same stocking balance	Yes	0.002

Chapter 3

Measurement and prediction of tree growth reduction in parking lots based on apparent available soil

ABSTRACT

Urban conditions have been thought to affect tree growth, but there is little conclusive evidence as to the severity of those influences or whether different species respond differentially to urban stress. It is important to understand reduced growth expectations, because they affect our design choices for the urban tree canopy, particularly as required by legislative mandate. Five tree species (*Acer rubrum*, *Prunus serrulata*, *Pyrus calleryana*, *Quercus palustris* and *Zelkova serrata*), growing in parking lots ranging from 18 to 23 years old in central and northern New Jersey were measured. Tree height, diameter at breast height (DBH), and canopy radius were measured, as was apparent plant available soil (nonpaved planting zone area). Tree DBH, commonly recorded for many municipal inventories, was found to be a useful predictor of canopy area. Data were normalized within site, to facilitate multiple site analysis. Across different parking lots, reductions in tree size were consistently associated with reduced apparent soil access. A previous study from Florida was used for comparison of regional data, permitting conclusions on canopy reductions, relative to specification of design space for tree establishment.

INTRODUCTION

New Jersey has been described as either entirely urban or completely occupied (Nowak and Walton 2005). Many areas of New Jersey that are not considered urban, on a resident per square mile basis, are largely dominated by suburban sprawl and infrastructure – the majority being large parking lots for shopping malls. In urban areas, parking lots are also a dominate feature of the landscape (Davis et al. 2010). Parking lots usually provide for tree canopy establishment in design, albeit limited by space concerns and maximum parking capacity issues (Hill et al. 2010).

Environmentally sustainable development and legislation continues to increase with the continued urbanization in the United States. A large part of environmentally sustainable development has been targeted towards parking lot design; for instance, formulae exist for the minimum number of trees to plant, required for ordinances or credits for design goals (Sustainable Sites Initiative 2009; US Green Building Council 2009; Windhager et al. 2010). Such formulae tend to use percent canopy cover, trees per number of parking spaces, or numbers of trees per paved area (Harris and Dines 1998; Kuser 2000), and there is typically a time frame associated with these requirements (Arlington County Chesapeake Bay Preservation Ordinance, Undated; McPherson 2001). When predicting the canopy area coverage in a design plan, it is uncommon to take into account the diminishing returns on tree growth, due to the smaller biotic capacity of the planting site when small places are used in design, as compared with large lawn-type planting spaces (McPherson 2001; Grabosky and Gilman 2004; Celesitan and Martin 2005).

In designed ecosystems, such as heavily paved urban cores, where environmental stresses are often exaggerated, more data is needed on growth expectations and service life of trees. To date, lacking hard data on the environments in question, designers have relied on published botanical observations, obtained under garden conditions, or have used similar estimates from indexed texts (Gerhold et al. 1993; Gilman 1997; Bassuk 1998; Porter 2000;

Dirr 2009) on landscape plant materials, nursery trade sales literature, or commercially available software based on horticultural growth expectations.

It is unreasonable to assume that trees planted in parking lots will reach the same size dimensions as forest trees, or trees in park settings, or even published expectations. The design vision or planting plan may meet a proposed benchmark for expected canopy coverage minima, but the reality over time is infrequently in line with the design expectation. Few trees in paved environments reach their intended canopy dimensions prior to being replaced (Schwets and Brown 2000). A reduction of size over time had been observed even when well-adapted species such as *Ulmus parvifolia* (Chinese elm) are planted as parking lot trees. In Gainesville, Florida, the canopy size of *Ulmus parvifolia* was restricted when the unpaved surrounding fell below 80 m^2 (Grabosky and Gilman 2004).

The goal of this study was to determine the relationship between canopy area to diameter breast at height (DBH) and attempt to determine the reduction of growth expectations based on site restrictions as represented by apparent available soil in the planting zone of the parking lot. This information was used to evaluate the expectation of growth in varied design details for planning parking lot tree planting spaces. We compared the results in two planting scenarios with those seen in Florida (Grabosky and Gilman 2004), in an attempt to extract general patterns across different regions in the United States.

MATERIALS AND METHODS

Fifteen parking lots throughout central and northern New Jersey (Table 1) were used to test the hypothesis of a decrease in expected canopy volume as non-paved soil area decreases. Sites were selected on three criteria: age, variety of plant spaces (described below), and the presence of species common to other lots in the study. Eighteen to twenty-three year old parking lots throughout the study area were selected. All study sites had trees in the paved zone, as well as trees on the exterior of the lot (non-limited soil areas). Tree species selected are commonly used by landscapers as acceptable parking lot trees as demonstrated by their frequent use in New Jersey: *Acer rubrum* (ACRU), *Prunus serrulata* (PRSE), *Pyrus calleryana* (PYCA), *Quercus palustris* (QUPA), and *Zelkova serrata* (ZESE).

Tree planting sites were classified as: (a) planting strip (soil limited on 2 sides of the tree, average width of 4 m with varying lengths), (b) tree pit (soil limited on all sides of the tree, average surface area of 6 m^2), (c) or nonlimited (tree not restricted by amount of soil), as well as by (d) the measured open soil area of the planting site. All trees of the selected species were measured, with the exception of trees that were known to have been replaced (and therefore not old enough to have reached the target age class, 18-23 years from planting). Tree trunk diameter (DBH) was measured with a diameter tape at 1.37m elevation and to the nearest 0.1 cm. Tree height was measured using a LaserAce 501 (MDL laser, Aberdeen UK) to the nearest 0.31 m. Canopy radius was measured in four directions, North, South, East and West using a linear tape from the center of the trunk to the branch tip; the four measures were averaged to determine radius. Apparent available soil was measured as the non-paved soil surface area available to the tree in question. When trees shared a non-paved area, such as a linear planting strip, the total available soil was measured for the entire linear strip, and then divided by the number of trees that could possibly use that soil, to determine average available soil per tree. When trees fit into a non-limited soil category, the soil area was defined as twice the area within the drip line of the canopy from the trunk of the tree.

All trees within each species were combined to generate simple linear regression models relating DBH to canopy radius and DBH to height. Measured tree parameters were normalized within their respective sites for each species. The normalization allowed for multiple site analysis and comparisons. It also allowed a simple method if meaningful relationships were detected because data would represent a percent reduction from expected growth. An average DBH by species was developed within each parking lot opening category (as openings tend to repeat in dimensions within but not necessarily among parking lots). Simple linear regression models were used to test differences in tree DBH, based on apparent available soil, as determined by area of non-paved surface for all parking lots. Using the average canopy radius of all trees in non-limited soil as the upper limit (set at 100%) for any given lot, we calculated relative canopy radius for all the other trees in the lot to gauge the differences in tree canopy size among the various planting situations. Data were then examined over all sites to determine the mean canopy radius, relative to non-paved soil surface area. Note, the data points in regressions representing the non-paved soil area represent average values for the species in a given parking lot using that specific sized opening, and thus represent varied population counts and variances for each data point.

Two planting zone areas were established as a scenario for comparison with the Florida study, which had 20 m^2 for linear strips, as a 2 m wide by 10 m spacing of trees and 6 m^2, as a spacing of 2 m by 3 m. These represented average openings in linear strips and tree pits, respectively. Results were compared to establish benchmarking for canopy growth reduction. All analyses were performed in MiniTab 14.2 (2005).

RESULTS

DBH was closely related to tree height and canopy width across all the site types and for all five species. Table 8 illustrates the reduction of canopy area calculated like a circle, based on observed radii. Inventory data on DBH for these species could potentially function as surrogates for canopy coverage, if the latter were not directly available, at least for younger trees within the size range observed (r^2 = .93). There is a positive correlation between space available and tree size in all species (Figures 1, 2, 3, 4, 5).

Acer rubrum

DBH was positively correlated with both canopy radius (r^2 = 0.83, p < 0.02, Figure 3a) and height (r^2 = 0.81, p < 0.015, Figure 3b). Maximum height observations in the range of 12 m and radius in the 5-6 m range are below (Hightshoe 1988) or on the low end (Dirr 2009) of published height and width expectations. Soil openings less than 50m^2 displayed a reduction in trunk diameter

(r^2 = 0.923, p < 0.02, Figure 3c), but the relationship between soil opening and canopy radius was even more definitive (r^2 = 0.959, p < 0.001, Figure 3d); a 50 m^2 opening was associated with a 45% reduction in canopy radius and a 70% reduction in canopy area (Table 8).

Prunus serrulata

DBH was positively correlated with both canopy radius (r^2 = 0.908, p<0.025, Figure 4a) and height (r^2 = 0.744, p<0.045, Figure 4b). Maximum height observations of 7 m and radius in the 6 m range are below the 15-22 m height and width expectations for this vase-shaped form (Dirr 2009). Soil openings less than 60 m^2 displayed a 45% reduction in trunk diameter (r^2=0.92, p<0.036, Figure 4c). The relationship between soil opening and canopy radius was less sensitive (r^2 = 0.87, p < 0.047, Figure 4d), and a 100 m^2 opening was associated with a 22% reduction in canopy radius, or 40% reduction in canopy area (Table 8).

Pyrus calleryana

Canopy radius and height were positively correlated with DBH (r^2= 0.843, p<0.024, Figure 5a and r^2=0.902, p<0.019, Figure 5b respectively). Maximum height observations in the range of 12 m and radius of 8 m range are in line with published size expectations of 12.5 m (Dirr 2009). From the significant (r^2 = 0.92, p<0.027) regression relationship between DBH and planting soil access (Figure 5c) soil openings less than 60 m^2 displayed a reduction in trunk diameter. The similarly strong correlation relationship (r^2 = 0.91, p<0.032) between soil opening and canopy radius showed a 60 m^2 opening associated with a 40% reduction in canopy radius (Figure 5d), or a 64% reduction in canopy area (Table 8).

Quercus palustris

DBH was positively correlated with both canopy radius (r^2 =0.80, p<0.04, Figure 6a) and height (r^2=0.88, p<0.03, Figure 6b). Maximum height observations in the range of 16 m (Hightshoe 1988) and radius of 5-6 m range are below published height and width expectations (Dirr 2009). Despite the fact that trees in this species were the largest in size expectation behind *Zelkova serrata* (Dirr 2009), soil openings were limited to smaller areas in this species across the 9 parking lots in the study. There were robust relationships between soil opening and both DBH (r^2=0.89, p<0.027, Figure 6c) and canopy radius (r^2=0.88, p<0.032 Figure 6d). These showed decreases canopy radius reductions of 30% in 30 m^2 non-paved surface area.

Zelkova serrata

Canopy radius was correlated with height (r^2=0.811, p<0.036, Figure 7a) and strongly correlated with DBH (r^2=0.925, p<0.001, Figure 7b). Maximum height observations of 14 m and radius of 5 m are below published height expectations (Dirr 2009), however the canopy radius in texts are often representative of a large vase-shaped growth form which had not fully developed in the trees included in this study. DBH and soil openings are highly correlated (r^2=0.96, p<0.001, Figure 7c).

Soil openings of less than 50 m^2 was associated with a 30% reduction in canopy radius (r^2 = 0.96, p<0.002, Figure 7d) or a 50% reduction in canopy area (Figure 8).

Comparison with the Florida study

Both Florida and New Jersey studies show reductions of at least 19%, with the majority of reductions greater than 49% in 20 m^2 of soil (Table 9). Larger trees, as described by Dirr's manual, are seen to have a proportionately larger reduction in canopy size both in radius and area. This is further illustrated at the 6 m^2 size with all reductions greater than 21% in canopy area, with the majority of reductions greater than 50% (Table 10).

DISCUSSION

Increases in the area of open soil provided in the installation design were associated with increased size of the trees 18-23 years after installation. All species exhibited a smaller size than published expectation, which suggests a lower size expectation for the urban canopy within the first 20 years (Dirr 2009). This is reasonable if we expect trees to live longer than 20 years to a full mature size. The data exhibited that the current legislative and design growth canopy expectations are not being met if the published mature size is expected within 20 years. Furthermore, the common planting zone soil access provisions resulted in much smaller tree sizes. In order to meet realistic expectations urban tree planting design, the influence of soil resource provision must be acknowledged. Either design choices could include increased planting spaces to yield greater size, or continue with current designs and lower size expectations, and compensate with an increase the total amount of trees planted in order to meet the canopy requirement for canopy legislation.

The largest trees observed were found in non-limited soil in the edge of the parking lots in all species, which is not surprising. Regression analysis of tree size, demonstrated significant relationships between DBH and both canopy radius and height. The largest trees in each species, although slightly smaller than published expectations, were still reasonable for the amount of time in ground. We compared the results from this study to a similar study done in Florida, US (Grabosky and Gilman 2004), and although the species are different, our data in canopy size reduction mirror those of thee earlier study (Table 9 and Table 10).

There were no data collected to determine the exact cause of the diminished size in this study; however, a previous study has shown a decrease in maximum terminal DBH as apparent available soil decreases (Sanders et al. 2012). Other researchers have investigated factors influencing diminished size, such as elevated soil temperature (Graves 1998), tree gas exchange (Celestian and Martin 2005), leaf chlorophyll concentrations (Celestian and Martin 2005), soil limitation (Kristoffersen 1999), soil water dynamics (Kristoffersen 2007), and soil compaction (Randrup et al. 2001).

When breaking the data down into planting size typologies across species (pit, planting strip, amount of non-limited soil), it is apparent that there is a reduction in growth when there is less than 20 m^2 of soil surface, as is typical of a linear strip of 2 meters width, planted at 10 meter spacing (Table 9), and an extreme reduction in canopy size, with a tree pit of 2 m by 3m (Table 10). For planting strips (linear strips or shared pits), small spatial changes of those strips can yield noticeable differences, 20 years later, consistent with findings in a terminal size study of tree species in central and northern New Jersey (Sanders et al. 2012).

The data are limited to central and northern New Jersey, but the conclusions can probably be expanded to the mid-Atlantic United States. This study provided parallel results from a similar study in Florida; however, more data from various locations across the US are needed in order to determine a comprehensive growth expectation model for a wide variety of locations for each species. Our best chance for planting success lies in providing adequate planting space. The levels of growth achievable with the current legislated planting requirements are limited. This study suggests that better planting design will better meet the intent for successful tree establishment. By providing a wider soil zone around trees, we can increase canopy coverage. There is a dramatic increase in canopy size when trees are planted in linear strips of at least $40m^2$ as opposed to $6m^2$ planting pits.

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Table 6: Site project age and species present for canopy analysis in central and northern New Jersey parking lots

Site	Town	Age (years)	ACRU[7]	PRSE[8]	PYCA[9]	QUPA[10]	ZESE[11]
1	Bridgewater	18	26	30	---	---	32
2	Princeton	19	---	49	70	67	---
3	Princeton	20	68	16	32	---	---
4	Freehold	22	---	62	12	---	42
5	Elizabeth	20	12	---	---	---	72
6	North Brunswick	23	---	41	37	21	---
7	North Brunswick	21	35	13	---	17	10
8	Woodbridge	22	---	30	52	9	13
9	Paramus	23	---	60	---	16	27
10	Paramus	19	18	---	67	---	17
11	Hackensack	20	---	12	40	31	---
12	Princeton	23	30	---	25	---	9
13	Piscataway	22	7	---	34	12	---
14	Edison	19	---	---	41	23	56
15	Edison	21	37	---	17	13	76
Total:			**233**	**313**	**427**	**209**	**354**

[7] ACRU is *Acer rubrum*
[8] PRSE is *Prunus serrulata*
[9] PYCA is *Pyrus callaryana*
[10] QUPA is *Quercus palustris*
[11] ZESE is *Zelkova serrata*

Table 7: Percent reduction of canopy area calculation in central and northern New Jersey Trees

Percentage of canopy radius	Example radius	Area of example	Percentage reduction canopy area
100	50	7854	0
90	45	6362	19
80	40	5027	36
70	35	3849	51
60	30	2827	64
50	25	1964	75
40	15	707	91
20	10	314	96
10	5	79	99

Table 8: Comparison of Canopy Area Reduction for trees in 20 m^2 soil in Parking lots from New Jersey and Florida.

Species	State	Tree Count	Parking Lot	Space relative to unlimited trees on parking edge.	Canopy reduction at 20 m^2
Acer rubrum	NJ	233	8	42.2% of edge	80% reduction
Prunus serrulata	NJ	313	9	71.6% of edge	49% reduction
Pyrus calleryana	NJ	427	11	42.1% of edge	80% reduction
Quercus palustris	NJ	209	9	66.2% of edge	56% reduction
Zelkova serrata	NJ	354	10	59.8% of edge	64% reduction
Platanus occidentalis	FL	78	3	71.8% of edge	49% reduction
Ulmus parvifolia	FL	287	4	55.2% of edge	64% reduction
Quercus shumardii	FL	43	2	71.4% of edge	49% reduction
Quercus laurifolia	FL	41	1	89.9% of edge	19% reduction
Quercus virginiana	FL	241	6	-------------------	--------------------

Table 9: Comparison of canopy area reduction for trees in 6 m^2 soil in Parking lots from New Jersey and Florida.

Species	State	Tree Count	Parking Lot	Space relative to unlimited trees on parking edge.	Canopy reduction at 6 m^2
Acer rubrum	NJ	233	8	35.7% of edge	85% reduction
Prunus serrulata	NJ	313	9	70.4% of edge	51% reduction
Pyrus calleryana	NJ	427	11	36.0% of edge	88% reduction
Quercus palustris	NJ	209	9	59.3% of edge	64% reduction
Zelkova serrata	NJ	354	10	56.3% of edge	67% reduction
Platanus occidentalis	FL	78	3	57.6% of edge	67% reduction
Ulmus parvifolia	FL	287	4	52.9% of edge	73% reduction
Quercus shumardii	FL	43	2	68.6% of edge	50% reduction
Quercus laurifolia	FL	41	1	87.6% of edge	21% reduction
Quercus virginiana	FL	241	6	-------------------	--------------------

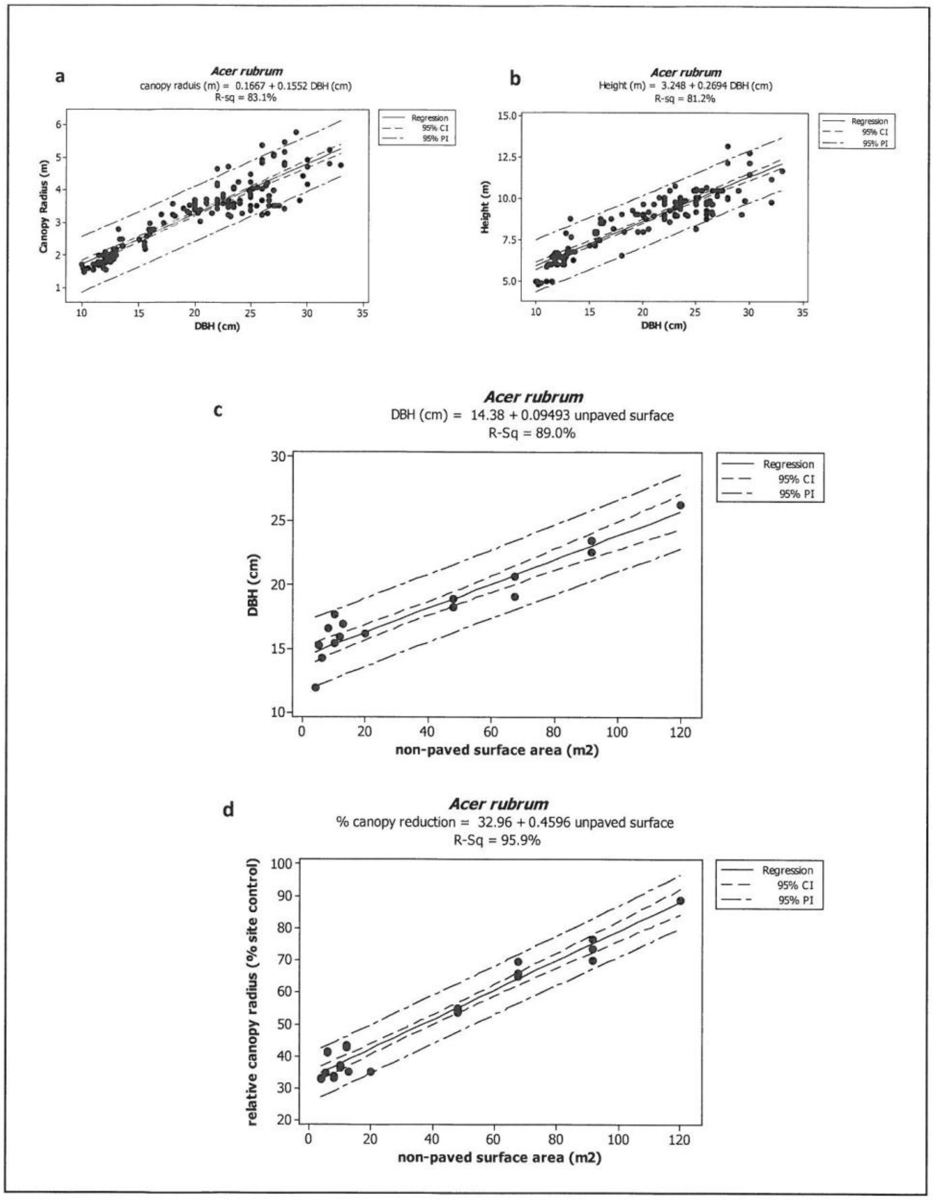

Figure 3: Canopy size relationships of *Acer rubrum* in northern and central New Jersey parking lots.
CI = confidence interval for the regression line from the existing date set at a= 0.05
PI = prediction interval for the new observations at a =0.05 for trees observed beyond the data set subject to the same criteria of treatment, species, age, region, and analysis.
3a. Simple linear regression relationship canopy radius to DBH.
3b. Simple linear regression of tree height to DBH.
3c. Simple linear relationship of DBH to open soil space in design detail. DBH data represents mean dbh within pavement size opening groups, normalized to mean canopy DBH for on-site control groups to fit multiple sites into analysis
3d. Simple linear relationship of canopy radius to open soil space in design detail. Canopy radius data represents mean radius data points within pavement size opening groups, normalized to mean canopy radius for on-site control groups to fit multiple sites into analysis

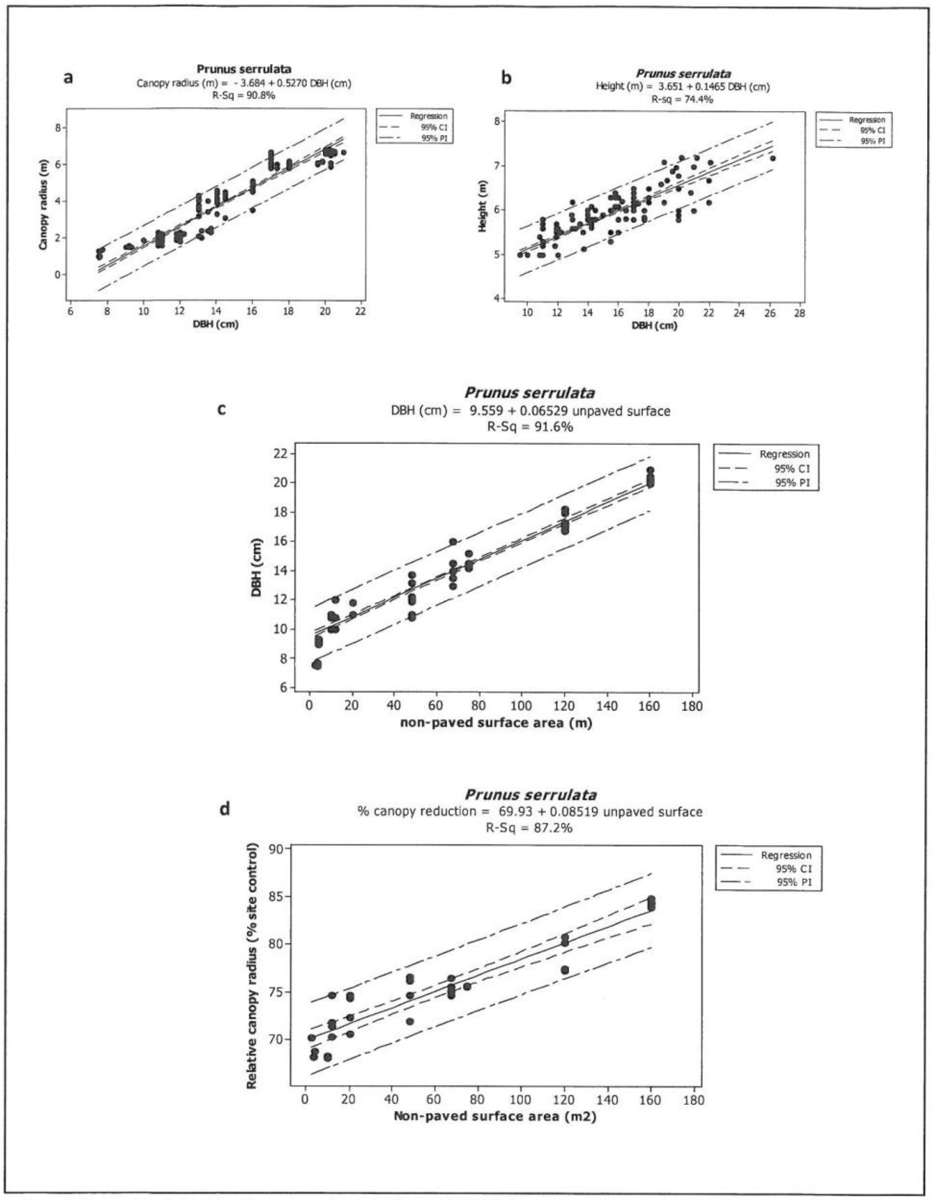

Figure 4: Canopy size relationships of *Prunus serrulata* in northern and central New Jersey parking lots.
CI = confidence interval for the regression line from the existing date set at a= 0.05
PI = prediction interval for the new observations at a =0.05 for trees observed beyond the data set subject to the same criteria of treatment, species, age, region, and analysis.
4a. Simple linear regression relationship canopy radius to DBH.
4b. Simple linear regression of tree height to DBH.
4c. Simple linear relationship of DBH to open soil space in design detail. DBH data represents mean dbh within pavement size opening groups, normalized to mean canopy DBH for on-site control groups to fit multiple sites into analysis
4d. Simple linear relationship of canopy radius to open soil space in design detail. Canopy radius data represents mean radius data points within pavement size opening groups, normalized to mean canopy radius for on-site control groups to fit multiple sites into analysis

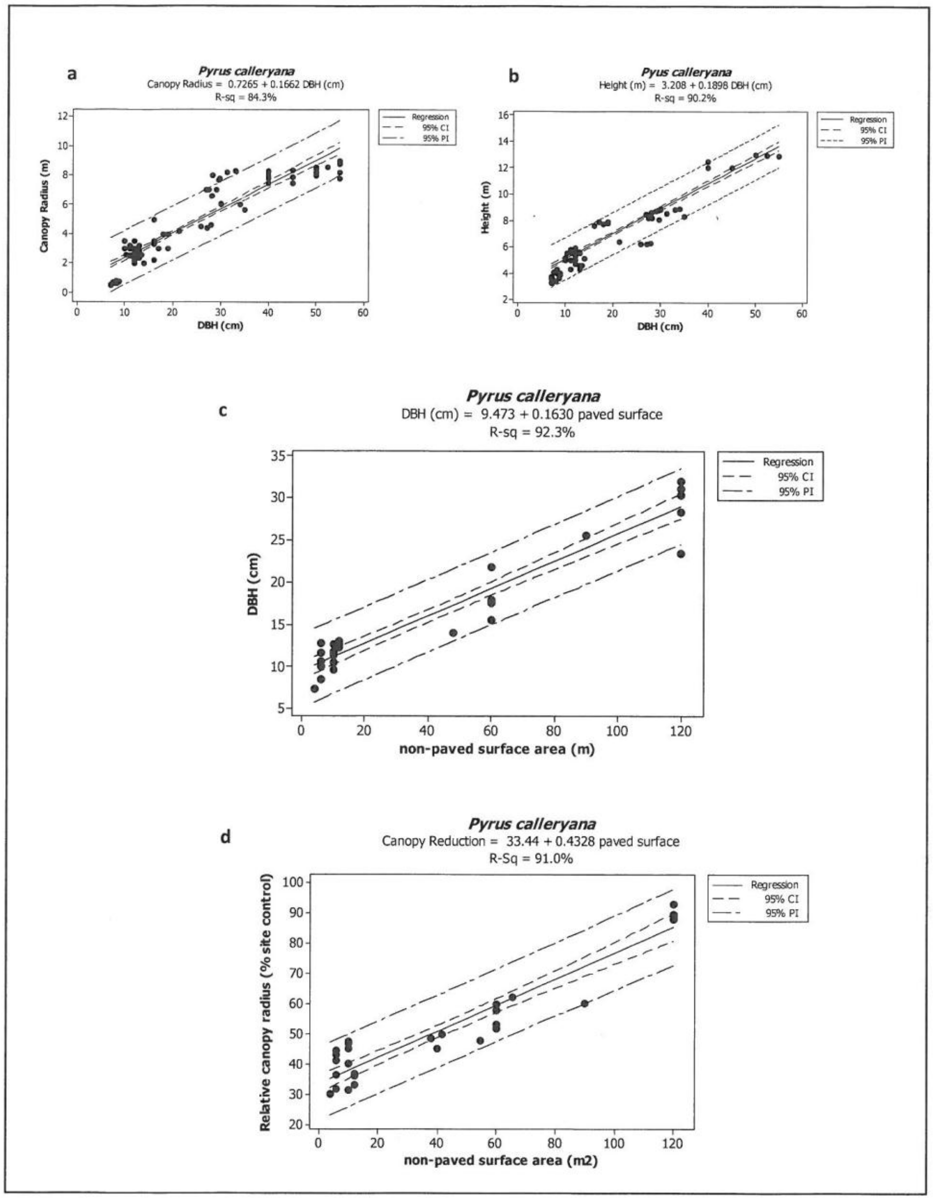

Figure 5: Canopy size relationships of *Pyrus calleryana* in northern and central New Jersey parking lots.

CI = confidence interval for the regression line from the existing date set at a= 0.05

PI = prediction interval for the new observations at a =0.05 for trees observed beyond the data set subject to the same criteria of treatment, species, age, region, and analysis.

5a. Simple linear regression relationship canopy radius to DBH.

5b. Simple linear regression of tree height to DBH.

5c. Simple linear relationship of DBH to open soil space in design detail. DBH data represents mean dbh within pavement size opening groups, normalized to mean canopy DBH for on-site control groups to fit multiple sites into analysis

5d. Simple linear relationship of canopy radius to open soil space in design detail. Canopy radius data represents mean radius data points within pavement size opening groups, normalized to mean canopy radius for on-site control groups to fit multiple sites into analysis

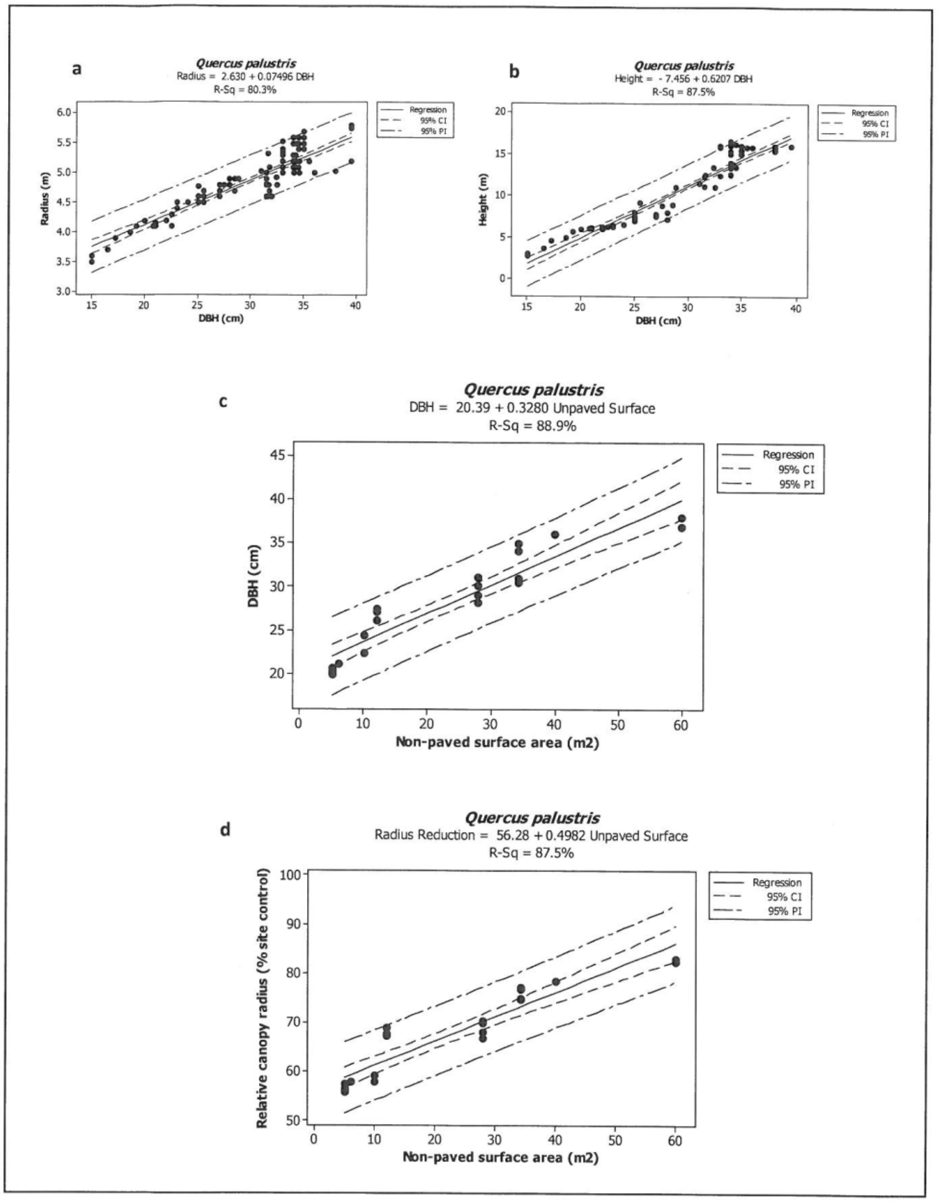

Figure 6: Canopy size relationships of *Quercus palustris* in northern and central New Jersey parking lots.
CI = confidence interval for the regression line from the existing date set at a= 0.05
PI = prediction interval for the new observations at a =0.05 for trees observed beyond the data set subject to the same criteria of treatment, species, age, region, and analysis.
6a. Simple linear regression relationship canopy radius to DBH.
6b. Simple linear regression of tree height to DBH.
6c. Simple linear relationship of DBH to open soil space in design detail. DBH data represents mean dbh within pavement size opening groups, normalized to mean canopy DBH for on-site control groups to fit multiple sites into analysis
6d. Simple linear relationship of canopy radius to open soil space in design detail. Canopy radius data represents mean radius data points within pavement size opening groups, normalized to mean canopy radius for on-site control groups to fit multiple sites into analysis

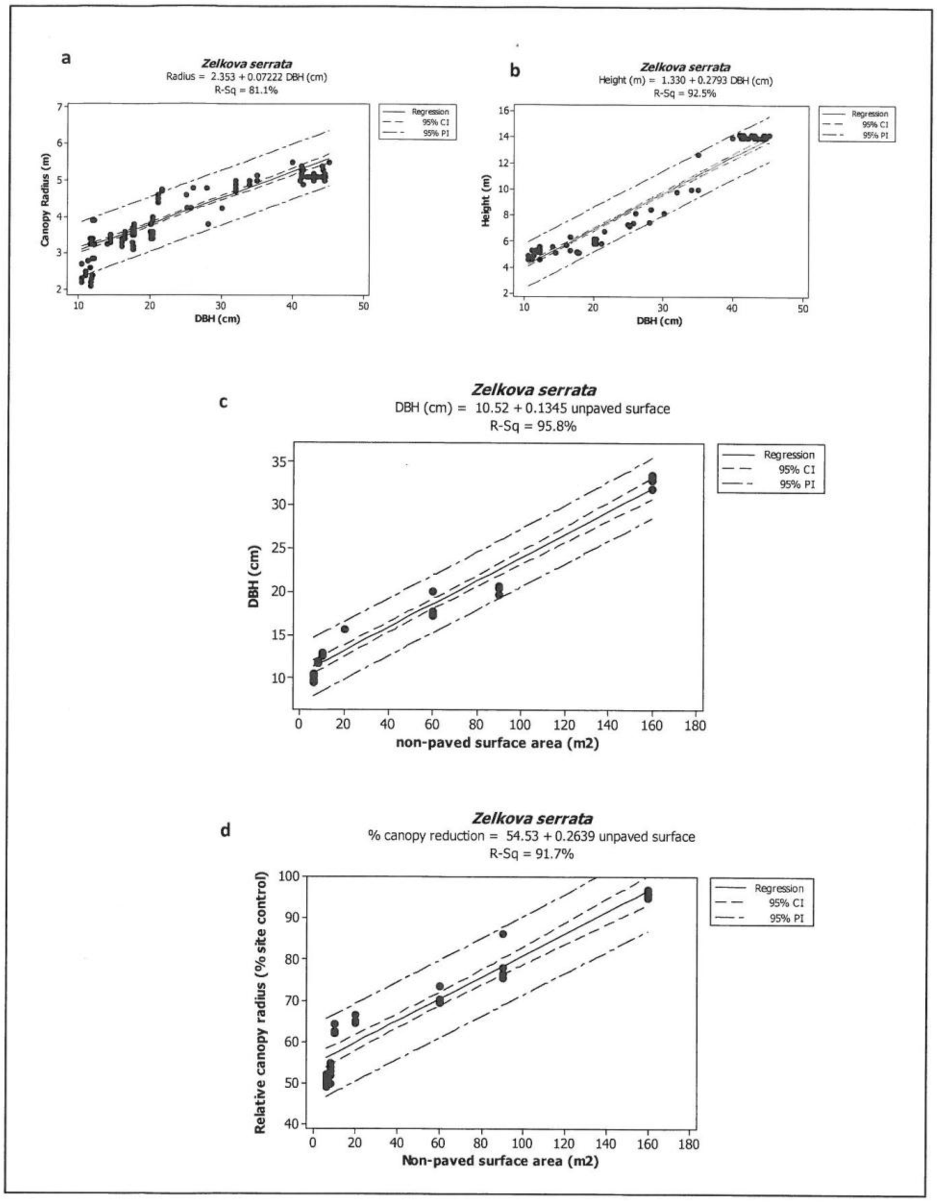

Figure 7: Canopy size relationships of *Zelkova serrata* in northern and central New Jersey parking lots.
CI = confidence interval for the regression line from the existing date set at a= 0.05
PI = prediction interval for the new observations at a =0.05 for trees observed beyond the data set subject to the same criteria of treatment, species, age, region, and analysis.
7a. Simple linear regression relationship canopy radius to DBH.
7b. Simple linear regression of tree height to DBH.
7c. Simple linear relationship of DBH to open soil space in design detail. DBH data represents mean dbh within pavement size opening groups, normalized to mean canopy DBH for on-site control groups to fit multiple sites into analysis
7d. Simple linear relationship of canopy radius to open soil space in design detail. Canopy radius data represents mean radius data points within pavement size opening groups, normalized to mean canopy radius for on-site control groups to fit multiple sites into analysis

Chapter 4

Establishing Maximum Size Expectations for Urban Trees with Regard to Designed Space

ABSTRACT

A major issue confronting application of forest management principles to urban tree management is a dearth of adequate data relating planting site, tree size, and tree age. Our object here is to determine the extent to which terminal size (stem diameter) can be linked to site type, allowing informed management and design decisions. Data from eleven communities in northern New Jersey were considered. Diameter Breast Height (DBH) distributions have already led to regionalized service life expectancies for commonly planted species, grown on different types of sites. Given those expectations, our goal here was to develop a method to identify trees approaching senescence. Three common urban landscape site types were evaluated: tree pits, planting strips, and unlimited soil sites. Thirty-one (31) taxa were present in large enough numbers to use in species-specific analysis. These species were classified into small, medium, and large size categories, based on published growth expectations. Our study developed DBH occurrence percentiles, and those larger than the 95th were described as having entered the maximum size range. Maximum attainable sizes differed among the three planting site types, and irrespective of size class, reduced planting space resulted in reduced maximum attainable size.

INTRODUCTION

Management of the urban forest inventory requires the balancing of public benefits against the public risks. Careful planning requires a detailed tree inventory that allows for literally 10s of thousands of trees in the urban context. Such an inventory typically provides information on the species composition and total numbers for the municipality, as well as tree location, tree age, and tree condition for individual trees, allowing point-by-point (tree-by-tee) management decisions, where that is appropriate. Too often, however, inventories are maintained with insufficient detail to enable either broad management or point-by-point intervention, where that is needed. We are also hampered by the lack of established methods for assessing age structure and sustainability within a mixed-species urban tree population.

Varied urban planting sites cause difficulties in the plant selection process and although short-term performance and survival is tracked, often long term performance is not evaluated. Moreover, urban areas are heterogeneous, and environmental stresses vary considerably, even among adjacent planting sites. Berrang (1985) examined over 80 variables for 375 trees planted near the Consolidated Edison facilities in New York City, and found that excess soil moisture, mounding of soil on roots, soil salts, and overall root system size were the most important factors affecting a tree's overall health. Chacalo (1994) surveyed 1261 street trees in Mexico City, collecting data for seven different variables, and concluded that problems with overall tree health could be attributed to planting in inappropriate locations, overall species choice, and lack of adequate planning and maintenance. Both of these studies indicated that planting site selection is a site-specific challenge, largely determining both short and long term success (survival and growth).

General wisdom suggests that urban design should provide a visual metric for evaluation to assist management decisions, because there is wide general understanding of soil biotic capacity and carrying capacity in of naturally forested systems, but our understanding of biotic and carrying capacity for urban context lags far behind that for naturally forested areas. Explicit evaluations of tree performance, in response to soil opening size (Sanders and Grabosky 2012; Grabosky and Gilman 2004) suggest merit for an explicit urban evaluation of biotic capacity within urban context. As the sustainability of some major projects (Los Angeles 2006; PlaNYC 2010; Million Trees Philadelphia 2010; Million Trees Miami 2010) are explicitly evaluated, it is becoming increasingly clear that the ability to predict plant performance and longevity, with relation to design choices is crucial for an appropriate program analysis. Planting spaces in urban design are usually reflective of three different site types: tree pits, planting strips, and non-limited soil sites. 'Non-limited' includes trees where the area underneath the drip-line of remains unpaved; an example 'non-limited' tree would be planted in a park or lawn setting. 'Planting strip' had less available soil per tree, best described as strip of ground where the available soil is bordered on two sides by structures or pavement, thus limiting the tree root zone. 'Tree pit' includes a cut-out, tree-well, or raised planter, where available soil was extremely limited, meaning the available soil was does not include the entire drip line of the tree.

Trees in the urban context are often planted in different sites that can vary for a multitude of variables. Although complete site analysis could be performed on every tree, this is not efficient, nor does it provide a general model for assessing trees. Using available soil surface as a 'minimalist' visual site type characteristic should facilitate better planting choices. Interpretation of tree size within a site can vary depending on its precise location. As a hypothetical example, a 51 cm DBH *Quercus rubra* is common and considered very large in a sidewalk zone in our study location, but it is very rare to see a 61 cm DBH in this zone. The same tree would be considered a mid-range size for the species, if found as a park or yard tree within visual distance of the sidewalk zone. *Cornus florida* or *Tilia tomentosa* would have differing profiles. Choices of plant placement in design require an understanding of maturity and longevity expectations, species by species.

Apparent available soil is certainly not the only variable which affects or informs the expectation of a tree's overall health, longevity, and maximum size, but it is an easily recordable and cataloged characteristic that can help predict what other variables may have an effect. The association of apparent available soil to tree size provides a prediction of tree growth behavior, reflective of the earlier observations of Grabosky and Gilman (2004), not a causation of the phenomenon. In application, site type could later be combined with work suggested by Bond (2010; 2012) to examine the condition of the tree in context with its expected maximum size.

Urban trees are often planted with an expectation of nearly zero attrition, in spite of ample experience to the contrary, skewing major portions of the urban canopy towards a single age class, and thus minimizing sustainability of the urban forest canopy (Clark et al. 1997). Urban forestry has historically emphasized tree planting and survival, with little attention directed to what constitutes mature and over-mature trees in the urban forest, or how common design responses affect ultimate tree size or service. Managing the urban forest as though it were a natural forest, we are unable to address stand longevity or a useful harvest interval.

Jim (1994) states that the many facets of managing tree replacement can be a daunting challenge to urban tree managers, and Hitchmough (1994) concedes that complexity and political nuances of the task have often been given as excuses for inactivity. Managing urban tree replacement is not a new problem (Solotaroff 1911). Pescott (1968), when addressing the planned replacement of trees, indicates that as early as 1954 in England, a government committee of relevant experts was formed to advise the Ministry of Works on the felling and planting of trees in all types of locations. Four decades later, Hitchmough (1994; p. 269) claimed, “many landscape management organizations are not adequately prepared to cope with problems of this intensity and magnitude.” By developing a repeatable method to define and explain removal and replacement a management plan can be implemented. In order to develop a management planning for urban tree harvest and replacement a useful endpoint to urban tree service life needs to be established.

Sinclair and Hudler (1988: p. 29) define the term “decline” of trees, as opposed to natural senescence, as “a premature progressive loss of health, distinguished from the normal occurrence of senescence by premature debilitation.” However, there is no current definition of “premature” death, because there are no established definition for normal rates of attrition and life expectancies for trees in urban settings. Sinclair and Hudler (1988) further developed four major aspects of tree decline: three describing disease, pests, and environmental stresses; the fourth describing synchronous cohort senescence. Trees of similar age growing in groups tend to display group behavior, including shared patterns and timing of senescence. Our intention herd is to develop a better understanding of senescence timing, acknowledging that planting site types we have suggested help us to define an evaluation of and management expectation for expected senescence time, one that is associated with environmental stresses the trees encounter.

This study focuses on the harvest interval by deriving a methodology to define over-mature trees, and providing a context for developing urban tree size expectations, which will in turn help to define a proper harvest interval. Our specific goals were to: (1) develop a better understanding of senescence in urban tree populations, and (2) determine how different planting design choices might influence mature size expectations.

MATERIALS AND METHODS

Our study included community inventories collected throughout northern and central New Jersey by a consultancy firm (Paul Cowie and Associates) hired to develop the inventories, as part of a state-wide program for the development of community forest management plans (NJDEP Division of Parks and Forestry 2011). Eleven communities were inventoried between 1995 and 2010, contributing to a database of approximately 45,500 trees and over 280 taxa. Data collected included tree diameter at breast height (DBH), planting site type, planting site area, tree species, tree genus, and maintenance recommendations. ANOVA was used to evaluate whether the 11 communities could be grouped, and based on the collective data, trunk DBH percentiles were developed.

We defined three planting site types, based on available soil: non-limited, planting strip, and pit. To avoid biases created by any single town's maintenance and care, only the taxa that were planted in all three-site types for at least three municipal inventories were included in this study. In addition to groupings by available soil, trees were also grouped according to maximum height in a park-like setting (Table 11). Small species were considered to be those whose maximum height at maturity was less than 9.1 meters, medium tree species were achieved greater than 9.1 meters but less than 15 meters, and large tree species were those that achieved greater than 15 meters (Gerhold et al. 1993; Hightshoe 1998; Bassuk et al. 2009).

Since this study focused on maximum size, the trees in each species were ranked into DBH percentiles as an aggregated species grouping across all communities within each site type. For each species - site cohort, those within the ninety-fifth percentile were described as the maximum size range (Table 12). This is a standard procedure for ranking populations in epidemiology and educational tests, and works well for the purposes of our study (Cannell 1988; Fraenkel and Wallen 1992; Schoonjans et al. 2011). The logic is that trees no longer occurring above a certain size class were removed because of excessive size for the planting site, removed due to declining vigor or condition, or died before exceeding the observed terminal limit size. Of the 280 taxa analyzed, we found 31 taxa present in large enough populations to tabulate within the size classification. The smallest number of trees considered acceptable for inclusion within a particular site type had to be greater than seven, which was required for at least one tree for inclusion into the 95th percentile rating. For 95th DBH percentile size-specific analyses, 32,898 trees were examined.

In this context, we assign the term over-mature to draw attention to the lack of trees in a larger DBH size category, which infers loss of the species from the inventory at some level. Of those 31 species (32,898 trees), eleven species (27,989 trees) qualified for a formal species-specific analysis on whether there is a difference in site type for maximum observed stem diameter size. Data were analyzed for those eleven species occurring on all three types of sites, using a null hypothesis of no difference in terminal size within species across the three soil types. A one-way ANOVA was used to determine whether there were differences between the within-species size classes for the three site types.

DBH values within each species were then normalized against their associated species maximum ([species max – species min]/SD) for inclusion in the general linear model. This allowed cross-species DBH size comparisons, despite differences in their species-specific size distributions and expectations, allowing multiple species to be grouped together into small, medium, and large tree types. As a relative scale, the species were combined for a general linear model on tree size (Small, Medium, Large), site type (pit, strip, non-limited), and an interaction term. There were a total of 32,898 trees examined for analysis in the general linear model, providing general trends across species, based on maximum plant size versus site type. A Bonferroni analysis was done on the interaction data means for DBH as a conservative method to control error-rate (Table 14). All analyses were conducted in MiniTab 14.1, after the data met all assumptions for statistical analysis; alpha was set at 0.05.

RESULTS

The 11 species used in the species-specific analysis represented 27,989 trees, or 61.6% of the general database population (Table 10). In all 11 species that met the requirements for inclusion in the analysis, site type had a meaningful impact on maximum observed size. The average maximum DBH size in a medium sized tree, *Pyrus calleryana*, in a pit was 18.1 cm, which was greater than its size in the planting strip (17.4 cm) but less than in the non-limited soil (21.5 cm). Within these 11 trees, Trees that had more available soil grew larger than trees that had small amounts of soil.

Each of the 31 species included in Table 10 had large enough sample numbers to permit definition of an over-mature rating for north-central New Jersey communities. The 31 species used in the general linear model employed 92% of the total inventory population. Maximum DBH in small, medium, and large trees varied, based on planting species. The general linear model showed that there was a statistically significant difference ($p < 0.001$, SD=18.1) between maximum DBH in small, medium and large trees in the tree planting typologies (non-limited, planting strip, and tree pit) (Table 12). All small trees DBH (13.8 cm – 26.0 cm) regardless of planting site type were less than their medium (33.7 cm – 57.8 cm) and large (71.4 cm – 94.2 cm) counterpart trees. There was also a statistically significant interaction between site type and species size ($p = 0.024$). Tree growth varied based on site type. In small trees, the average DBH of tree pit trees (13.8 cm) was less than planting strip (19.5 cm) and non-limited soil trees (26.0 cm). In medium sized trees, the smallest average DBH in the 95 percentile was found in planting strips (33.7 cm), followed by a greater DBH in tree pits (37.8 cm) and non-limited (57.8 cm). Mean separation by Bonferroni on preplanned comparisons of site-type showed there was a difference in growth based upon apparent available soil. Tree size was different by definition (Table 13). The general linear model exhibited an interaction of site type and tree DBH. The small, medium, and large trees have statistically larger DBH in the planting strips and non-limited soil than in the tree pit (Table 11). It should also be noted that small trees were observed in small sites, but not in large sites, this is common for the urban forest. Common wisdom dictates that large trees will provide more shade, which is why small trees tend not to be planted in such sites.

DISCUSSION

Trees in the urban forest face many environmental stresses that are exaggerated in the developed municipal landscape when compared to their natural system counterparts. In order to discuss stand management, an understanding of species senescence and mortality is needed to define a service life endpoint. Budgetarily, it is hard to manage an urban forest if you are not aware of the endpoint of the varied species in the urban forest. In traditional forest management, harvesting or harvest intervals are in part determined by the maximum size a particular species can reach in a forest, or how long it takes to reach a targeted merchantable size based on its maximum potential. Although, there are many factors that could possibly affect a tree's maximum size in the urban forest, this study focused on tree selection on a site-specific level with the criterion of apparent available soil. Although age is relevant to the urban forest, a more practical criterion for evaluation of maturity is the expected maximum size, and this can break into rapidly defined site types in which the tree species grows as a proxy variable. Most inventories already supply the metrics used in this study and offer the ability to apply the approach to other tree species and regions.

Trees in the 95th percentile reflect when the tree can become a liability to the urban forest as seen in the comments by the arborist who consulted on the project and collected the inventories. For example (Table 11), *Acer platanoides*, *Acer saccharum*, and *Quercus palustris,* trees that were included in the 95th percentile (the maximum size trees) in these species were recommended for immediate removal due to poor condition. This suggests that their useful service life had already been exceeded. The data also illustrated pulses of planting activity (data not shown) as clusters of size, in which a large numbers of trees at a specific DBH where observed. This suggests large planting initiatives that planted certain species at the same time at a common purchased size. These trees will all reach a relative maturity at the same time within a site type, potentially causing an entire portion of the urban forest to be removed for risk at the same time. The urban forest needs to be managed for the future to avoid an even-aged population in which all trees would be dying at the same time. In traditional silvicultural forest management even age stands can be useful as some species regenerate better this way. As the urban forest is typically manually planted, it may be wiser to move towards un-even age stands for a given species to protect against loss of a certain proportion of the canopy cover due to when the species reaches a maximum DBH. Knowing the maximum DBH for a given species would allow urban foresters to estimate how rapidly they are approaching this potential end point for a species and begin selective harvesting and replanting over time with the goal of moving towards a mixed aged stand. As such, the urban forest should be managed on a schedule in which trees are planted in multiple years to enable a cropping rotation to determine an uneven-aged stand management. For a street renovation planning sequence, there is a possible benefit in planning harvest intervals on a small street-level scale rather than over the entire management zone.

Care in the interpretation of results is warranted. While some of our species (*Pyrus calleryana* and *Zelkova serrata*) have only been actively planted over the past 30 years, their full life expectancy has probably been reached. For such species, the maximum size criterion will probably require adjustment in the future, as the trees have not yet achieved a maximum DBH. As the area of unpaved surface in the planting zone becomes larger, from pavement opening, to linear planting strip to park/lawn situations, the terminal DBH increases. Clearly, at performance expectations must be species-specific, we have enough information to provide some planting design guidance. This could provide a context for developing urban site index or urban size expectation to help determine harvest interval, the time when trees begin to accrue costs at a greater rate than returning ecosystem services. This study also used 11 cities, whereas a manager would typically use this method in a single municipality. Although there were differences in soils, compaction, and other site-specific criteria from location to location, the relationships described in all 11 communities are consistent and provide for a start of an urban tree size expectation. Care is needed in the interpretation of these results, but the methods provide a robust and repeatable benchmarking approach for long-term evaluation by managers and researchers.

Treated as an aggregated general population in the general linear model, DBH is assumed to be associated with canopy size. Natural form suggests that tree canopy volume is proportional to natural height and DBH. Regardless of the size class of the tree species, reduced planting space resulted in reduced maximum DBH. As trunk size is related to canopy volume, it stands to reason that reduced planting spaces result in reduced canopy volume. For design, problem solving, or planning in the management of the urban canopy and trying to determine service life of trees, plotting DBH versus tree height or canopy can provide urban forest managers a reasonable estimation of size expectation.

CONCLUSION

The traditional goals of urban forest management have been to minimize premature tree losses and to manage trees to the point at which they begin to accrue costs that are greater than the ecosystem services they provide. Our goal was to develop a construct for cohort senescence, which is developed and replicated from existing data in regional inventories. Thus, the manager could use these maximum size determinations with existing community tree inventories. This study seeks to improve management by using consistent site types to use in conjunction with current inventory practices. With these data, maximum size and a reasonable service end point have been determined for several tree species. DBH is a viable surrogate for age. Additionally, design plans need to accommodate a reasonable design size expectation to then provide a reasonable idea of services for the associated investment if some service is associated with canopy size/volume. Within such planning and evaluation, this rapid assessment is very useful.

This study shows a DBH size for when urban trees reach a size in which they would be considered at their service endpoint. There are exceptions to every rule, and we allow for the possibility that some trees may grow larger in a site typology. However, on a whole, these numbers serve as a general recommendation for service endpoints. Preventative maintenance is crucial in the urban forest and allows for a stable canopy across time.

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Table 10: Trees Observed in complete urban municipal tree census inventories from 11 communities across North/Central New Jersey. A listing of the most common species occurring in each size class by planting zone typology, their occurrence in the data set, and the maximum range observed DBH. Those ***bold italic***, were used within species specific analysis, given their occurrence in adequate numbers across all planting typologies.
*Cultivar popularized and gained rapid wide adaptation

SPECIES		n	
	Pit	strip	nonlimited
SMALL			
Acer palmatum	0	7	84
Cornus florida	0	16	304
Prunus serrulata	1	87	770
Syringa reticulata	8	107	34
MEDIUM			
Acer campestre	0	45	61
Carpinus betulus	1	263	44
Cladrastis kentukea	0	18	62
Phellodendron amursense	1	14	60
Picea pungens	0	6	178
Prunus yedoensis	0	3	165
Pyrus calleryana*	***136***	***714***	***687***
Sophora japonica	4	24	76
LARGE			
Acer platanoides	***94***	***4052***	***3304***
Acer pseudoplatanus	0	14	69
Acer rubrum	***30***	***1404***	***2408***

Acer sacchariunum	***8***	***530***	***392***
Acer saccharum	***30***	***694***	***1280***
Aesculus hippocastanum	9	49	89
Gleditsia triacanthos	***60***	***225***	***507***
Platanus x acerifolia	***43***	***1411***	***883***
Prunus serotina	0	13	618
Quercus alba	3	9	226
Quercus bicolor	5	69	140
Quercus coccinea	3	39	262
Quercus palustris	***66***	***1973***	***3358***
Quercus phellos	8	36	172
Quercus rubra	***32***	***337***	***1494***
Quercus velutina	1	20	205
Tilia cordata	***32***	***332***	***551***
Ulmus americana	11	60	339
Zelkova serrata	***47***	***529***	***343***

TABLE 11: Trees in Central and Northern New Jersey. 95th percentile (Maximum observed) DBH values including the average DBH (cm) in 95th percentile and the maximum DBH (cm) observed.
Subscript within row were deemed statistically significant at alpha 0.05
* all trees in category observed to be recommended for removal on condition appraisal

Species	Pit		Planting Strip		Non-limited	
	Max	Average	Max	Average	Max	Average
Acer platanoides	42	27.34_{a}	27.5	27.3_{a}	43.5	30.7_{b}
Acer rubrum	27.5	11.9_{a}	41	31_{b}	46	32.3_{b}
Acer sacchariunum	33.5	33.5_{a}	49	40.6_{b}	56	45.7_{c}
Acer saccharum	36*	32.1_{ab}	34	29.4_{a}	45	34.9_{b}
Gleditsia triacanthos	15.5	15.5_{a}	27.5	19.9_{a}	45	32.1_{b}
Platanus x acerifolia	42	39.3_{a}	49	38.4_{a}	61	41.6_{a}
Pyrus calleryana	21.5	18.1_{ab}	24	17.4_{a}	25	21.5_{b}
Quercus palustris	45.5*	33.7_{a}	44	36.0_{b}	80	41.0_{c}
Quercus rubra	34	33.5_{a}	50	38.4_{b}	71	48.2_{c}
Tilia cordata	27.5	24.5_{a}	39.5	31.4_{b}	45	36.9_{c}
Zelkova serrata	9.5	9.5_{a}	22	16.9_{b}	37.3	28.7_{c}

TABLE 12: Central and Northern New Jersey Maximum size expectations. General Linear Model, DBH (cm) vs. Planting site type and planting tree size and comparisons of means.

Factor	Type	Levels	Values
Site Typology	fixed	3	nonlimited, planting strip, tree pit
Species Size	fixed	3	large, medium, small

Source	DF	F	P
Site Typology	2	26.63	<0.001
Species Size	2	187.00	<0.001
Interaction	4	2.81	0.024

Table 13: Comparison of DBH Means (cm) in species size versus site typology with bonferroni protection in pre-planned contrasts. Subscript in column denote statistical significant at alpha 0.05

Site Typology	**Species Size**		
	Small	**Medium**	**Large**
Nonlimited	26.0 (b)	57.8 (b)	94.2 (b)
Planting Strip	19.5 (a)	33.7 (a)	79.5 (a)
Tree Pit	13.8 (a)	37.8 (a)	71.4 (a)

Chapter 5

Conclusions and future directions

In the process of conducting these studies, the overwhelming conclusion is: site and species matter in the urban context. In evaluating design choices and growth, we can examine pits, strips, and parks differently. While there are clearly species-to-species differences and exceptions to the rules, there are clear breakout points within general species size classes (small, medium, large) and in specific species as well. There is a clear difference in the terminal size of trees based on the tree sites. Current management techniques should also be changed for better species selection by planting smaller growth potential trees in small planting spaces and allowing larger planting spaces for larger trees.

While there is research into micronutrients in urban soils (Scharenbroch and Lloyd 2006; Pouyat et al. 2007; Scharenbroch and Catania 2012), especially focused on zinc levels (Bartens el al. 2012), this is not a reliable predictor of success in the urban forest, nor is it an easily measurable item for managers to evaluate and provide remediation. What really matters on a gross scale is the access to soil capacity rather than the details within the volume of soil. Based on these studies, it is reasonable to organize a concept for assigning an urban site index to predict or define plant performance and/or growth expectations in a manner useful to urban tree managers. In the development of a site indexing system the easily measured parameter with a large explanatory power is apparent available soil. Apparent available soil has the ability to describe the size of the tree and the potential reduction in canopy when compared to a non-limiting soil amount. It also has a greater impact on trees of a larger stature (medium and large trees, chapter 4). This makes logical sense when considering the amount of soil that is necessary to provide ample root structure in a large tree.

What has astounded me throughout the research is the prevalence of a common but clearly misguided practice by designers, planters and managers. Trees are consistently planted in small holes with the hope that these trees will reach their full potential despite ample evidence and to the contrary. Ignored is the fact that in a 6 m^2 pit it is very hard for a tree to reach maximum potential without causing problems to the sidewalk, curb, or underground pipes. Instead, they plant smaller sized trees (potential growth height under 25 ft.) in park scenarios, with access to unlimited soil. These smaller trees would be better suited in the tree pits. There is less of a problem for small trees with adequate soil amounts and even if it reaches its horticultural prescribed size, it is still under the majority of power lines if it were planted in a street tree scenario.

Another idea that has been illustrated in Paramus, New Jersey is a lien on properties, where the town owns five feet in from the sidewalk (Kuser 2000). Instead of trees planted in the small strips between curbs and sidewalks, they are planted on the house side of the property line, which allows for an increase in available soil and assures that the trees are far enough away from the power lines so that they do not need to have large sections removed for safety clearance. Although this is not practical for all communities, it can be a good template for future practice when possible. The more realistic option is to plant trees as others fail, with better species-for-space selection. Urban foresters have over-simplistically coined the term “right tree, right place” and though, there is not necessarily a right tree for every place, there is often a better choice or option for each planting scenario. If a goal is to increase urban canopy, then we should plant trees where they have the opportunity to achieve the largest potential of their natural canopy as possible, since urban canopy is measured on a canopy volume aspect (Bond 2010).

Another point of emphasis should is: natural history informs future service life (i.e. species matters). The urban forest is much more diverse than a typical forest. This is not only a standard practice, but also a necessary one, in order to combat cosmopolitan pests that can decimate species, thus leaving the urban forest barren as illustrated by Michigan after Emerald Ash Borer, which caused the removal of 353,000 Ash trees (Kovacs et al. 2010). Landuse is a useful predictor of transplant survival: there are some trees that are proven performers, able to withstand the harshest environments. My suggestion is that we take these proven performers and use them in the industrial corridors (the harshest land use) but then utilize the rest of the city to plant a more widely diverse range of trees. Although this is not the most ideal of situations, it is better than the alternative of no trees or dead trees in the industrial corridors.

There has been a wealth of knowledge that I have learned from while conducting my research, not only my own studies, but others as well. However, the field of urban forestry lacks a set of organizational constructs that would allow this knowledge to be delivered to other scientists to further the field, as well as common citizens. These citizens need this valuable information just as much as scientists do, in order to raise awareness and promote the proper use of the majority of the urban forest which does not reside on public grounds.

Organization of Knowledge

Plant growth has long been evaluated using a logistic/sigmoid growth curve (Figure 8). I have also used the logistic growth curve, not only as a way of describing growth, but to help define necessary stages for the urban forest. During the initial stage (lag phase) of transplanting and establishment in the urban forest, the rate of growth is slow. That rate increases rapidly during the exponential phase, but as the supply of nutrients becomes limited generally, the growth slows to the stationary phase. My three studies look at each one of these aspects and seek to define individual aspects of them.

Chapter 2 described an establishment phase for urban trees, suggesting that early transplant mortality and establishment mortality could be limited to provide a greater continued success for the urban forest.

Chapter 3 entails the maturity portion of Figure 9. In order to better manage for the future, we need to have clear size expectations based on research, not just ideals alone. Examining 20 year old trees in ground in three different scenarios (tree pit, planting strip [linear strip] and non-limited soil), we can see reductions in canopy volume and diameter breast height (DBH) based on apparent available soil. This is important for the future if we continue to monetize the services of the urban forest (e.g. i-Tree, Sustainable Sites). There is a large reduction in total canopy, which transfers to a decrease carbon sequestration capacity. There is a decrease in amenity value for large trees that are not in adequate places to ensure maximum growth.). Chapter 3 also starts to define the height of the curve and shows what the potential is 20 years post-planting.

In Chapter 4 (Establishing Maximum Size Expectations for Urban Trees with Regard to Designed Space), maximum DBH serves as a surrogate for age in order to determine an end point for a harvest interval for removal of urban trees prior to senescence. Harvest interval is dependent not only on the grouping of species based on maximum height, but also on site type (tree pit, planting strip, or non-limited soil). This chapter helps to define the height of the curve and how long it takes to get there. Although age was not directly studied, data in the future can be extrapolated to determine time period as a function of DBH. If DBH can serve as a proxy for canopy volume, we can create a service life endpoint using this study.

Urban citizens are provided services by the ecosystems in which they habituate. The quality of life for urban citizens is improved by locally generated services in the urban forest (Bernatzky 1983;Bolund and Hunhammar 1999). If we take the sigmoid growth curve and add ecosystem/environmental component to it we can start to evaluate these perceived ecosystem services (Figure 9). If we use this curve to define a single ecosystem service, for example maintenance of air quality, we can start to measure the CO_2 sequestration by trees, and the air pollutant removal by trees. This can show us the benefits we are receiving based on this environmental service; we can also add costs to this tree. How much did it cost to plant and maintain this tree. This curve can be used not only to measure individual ecosystem services (i.e. storm water management), but can also be used to evaluate all ecosystem services by urban trees. Various work has already established methods for assessing urban forest structure and ecosystem services (Konijnendijk et al. 2005; Nowak et al. 2008; Albetri 2009; Dobbs et al. 2011) to greater enhance the social and environmental services provided by current canopies.

Managing tree replacement involves the planning and scheduling of trees to be replaced with the aim being to sustain or increase the benefits from these trees and the continuation of a steady landscape. The task of long-term planning in the urban forest is difficult, however. Currently, urban foresters are marketing urban tree services based on canopy volume (i-Tree streets, i-Tree eco/UFORE). My studies show that the x and y axis on Figure 2 vary by species selection (small, medium, large) and also by planting site type (pit, strip, non-limited).

With the addition of these compounding factors, we do not have just one curve for trees, but nine different curves each with a different trajectory. With the addition of designed soil (CU-soil and Silva cells) and planting scenarios, there could be at least an additional site type, which would give a trajectory of 12 different curves to more accurately describe the ecosystem services of the urban canopy, based on site and tree size.

Currently, all trees in i-Tree are modeled from one species in one site type. This does not take into account the truly diverse urban forest. A simple cost/benefit curve for the urban forest can be used to demonstrate the necessity of quantifying a harvest interval for urban trees (Figure 9). This research promotes a proactive management with planned removals, before trees become hazardous. The cost-benefit model or harvest interval may also be useful to managers in indicating when benefits of a tree begin to decline. On the other hand, while these models are not currently used in management, they are used and found very useful in demonstrating the value of a tree to politicians and members of the forest service (i-Tree).

When we examine trees over a long time period, we see a pattern of deferred replacement or disturbance in the urban forest. This can be attributed to the most common method of urban management, where mature and over-mature trees are maintained, propped-up, and sustained for as long as possible until they are no longer safe or attractive. Only then are they removed and replaced with young specimens. Keeping large, mature specimens in the landscape is certainly a benefit to this approach. Arboricultural techniques aimed at prolonging the life of trees will improve vigor and slow the rate of senescence by providing an ideal growth environment, however these practices cannot halt or reverse decline indefinitely. A major drawback to this approach is the time lag between the aesthetic contribution of the large tree before decline and the new tree returning to the same aesthetic level.

If you further dissect the logistic growth curve (Figure 8) and think of it from an epidemiological or medicinal background, a stable dosing or trajectory occurs. In the area of pain management, when a person has pain, an analgesic is taken at the start of the pain and then to relieve suffering at a prescribed interval rather than at the continued onset of pain. This constant dosing is done in order to retain the maximum benefits while keeping a consistent amount of medicine in the body. If we imagine managing the urban forest in the same way, we can think of deriving a harvest interval to maintain a constant urban canopy cover.

The idea of site index has long been used in forestry, but the concept of establishing an urban site index is a new concept. I believe that an urban site index is a necessary step in understanding how to better improve the growth potential in the urban forest, as well as helping to provide larger trees in each growing situation. Although the studies do not define the entire site index in the urban environment or all the aspects that could influence site conditions, they do explain a large portion, and offer a launching pad for future research to continue to describe a site index.

How to Use it

One purpose of cost-benefit models is to provide a tool to assist managers in making a decision about when to harvest and replace trees. This research is promoting proactive management, with planned removals before trees become hazardous. In this case, the model may be particularly useful to managers to indicate when the benefits of a tree are beginning to decline.

We can quickly see the consequences of this design in: survival; 20 year growth trajectory; and in terminal size. By informing managers of the trajectories of species, we can greatly increase not only the efficiency in their job, but also in the performance of urban trees. Calculating DBH growth for the four species that occurred in both Chapters 3 and 4, we can see an accurate depiction of a reduction of size based on terminal capacity (Table 14). A manager can now use these tables and formulas to realize the potential of where their tree is now versus where it has the potential to be. No longer are we held to a standard set by ideal conditions.

Traditionally, there has been a cycle of replacement in the urban forest where replacement of a tree only occurred after a removal, creating an unstable canopy (Figure 11). If we treat urban trees as crops, a "crop cycling" could be implemented to reduce the considerable lag time between the aesthetic contribution of the old tree before decline and the new young tree returning to the same aesthetic level and the ability to provide a stable and sustained set of ecosystem services over time (Figure 12). Additionally, the constant trajectory and an idea of how large trees will get in different site types, allows managers to budget for when care and maintenance will be needed as well removal and replanting. It allows funds to be tracked for planting programs over multiple years and budgeted for when they will be needed, rather than all at once.

Future Directions

The current software used to model the ecological and environmental benefits of the urban forest is i-Tree. I-Tree was developed to promote effective urban forest management and sound arboricultural practices; however, as a model there are details that need updating. Only recently has i-Tree started to include information on specific species, recognizing that not all species will reach the same height or have the same canopy spread with similar DBH. However, the species specifications that it has started to include come from Horticopia, whose growth trajectory is based on ideal conditions. By incorporating urban sites into i-Tree, a more accurate idea of ecosystem services could be produced based on a reduction of growth seen in different sites. The pit, strip, and lawn (non-limited soil) are predictors of longevity and level of service. I acknowledge that this is merely a start, and by no stretch is this the only constraint on urban trees, however it is one that is easily measurable and describes a lot of the differences in tree sizes within the same species. The idea of "right tree, right place" has long been a mantra discussed in the urban forestry community: the thought that there is a right tree for every place in the urban forest, may be ill-informed but I do think there are trees that better fit sites.

An area that needs future research is time scale. Currently we have an idea of how site effects survival, growth in 20 years, and terminal size, but we do not know how long the process takes to reach a terminal size. Since we have been using size as a surrogate for age, further examinations need to be taken to develop the time-scale axis (y-axis) in the cost-benefit curve (Figure 9). Although this could be researched by coring urban trees, I feel that in a cosmopolitan pest environment, an additional introduction just to determine age does not seem worth the life of the tree. Another practical method, though time consuming, to attain the age of trees would be to look back into municipal planting records.

What is lacking from the current research is how site can impact the height and length of service in the curve. This is an idea for future research, one in which I intend to further in my post-doctoral research. With the introduction of a time-scale and age to our knowledge base, we can start longitudinal studies in order to create an urban site index nationally. Although regions may vary based on plant species and climatic inputs, there is the possibility for an urban site index to better inform planting practices in urban environments. These longitudinal studies will also look at the blurry idea of site transition zones, and will attempt to answer the questions "When does a pit become a linear strip?", and "When does a linear strip become non-limiting soil?"

This dissertation has sought out to enhance the understanding of defining and measuring the urban forest. Though this process remains a complex issue, I have provided an organization construct for researchers to help improve the urban forest for years to come.

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Table 14: Tree DBH (cm) trajectory based on 20 year growth and terminal size

Species	**Tree Pit**			**Planting Strip**			**Non-limited Soil**		
	20 years	**terminal AVG.**	**MAX**	**20 years**	**terminal AVG.**	**MAX**	**20 years**	**terminal AVG.**	**MAX**
Acer rubrum	14.6	11.9	27.5	16.3	31	41	25.8	32.3	46
Pyrus calleryana	10.5	18.1	21.5	12.7	17.4	24	29.0	21.5	25
Quercus palustris	22.4	33.7	45.5	27.0	36.0	44.0	60.0	41.0	80
Zelkova serrata	11.3	9.5	9.5	13.2	16.9	22.0	26.7	28.7	37.3

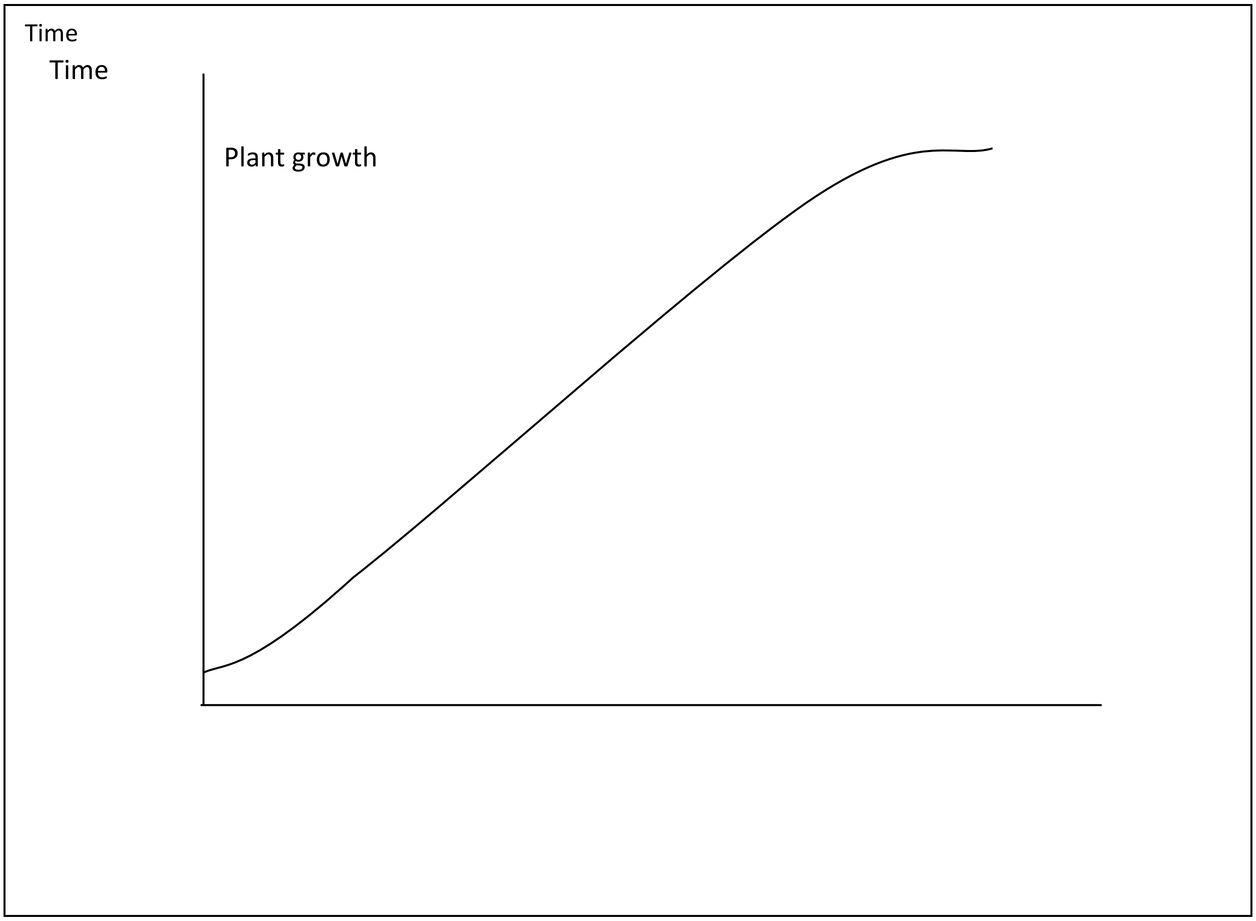

Figure 8: Sigmoid Growth Curve for Plants

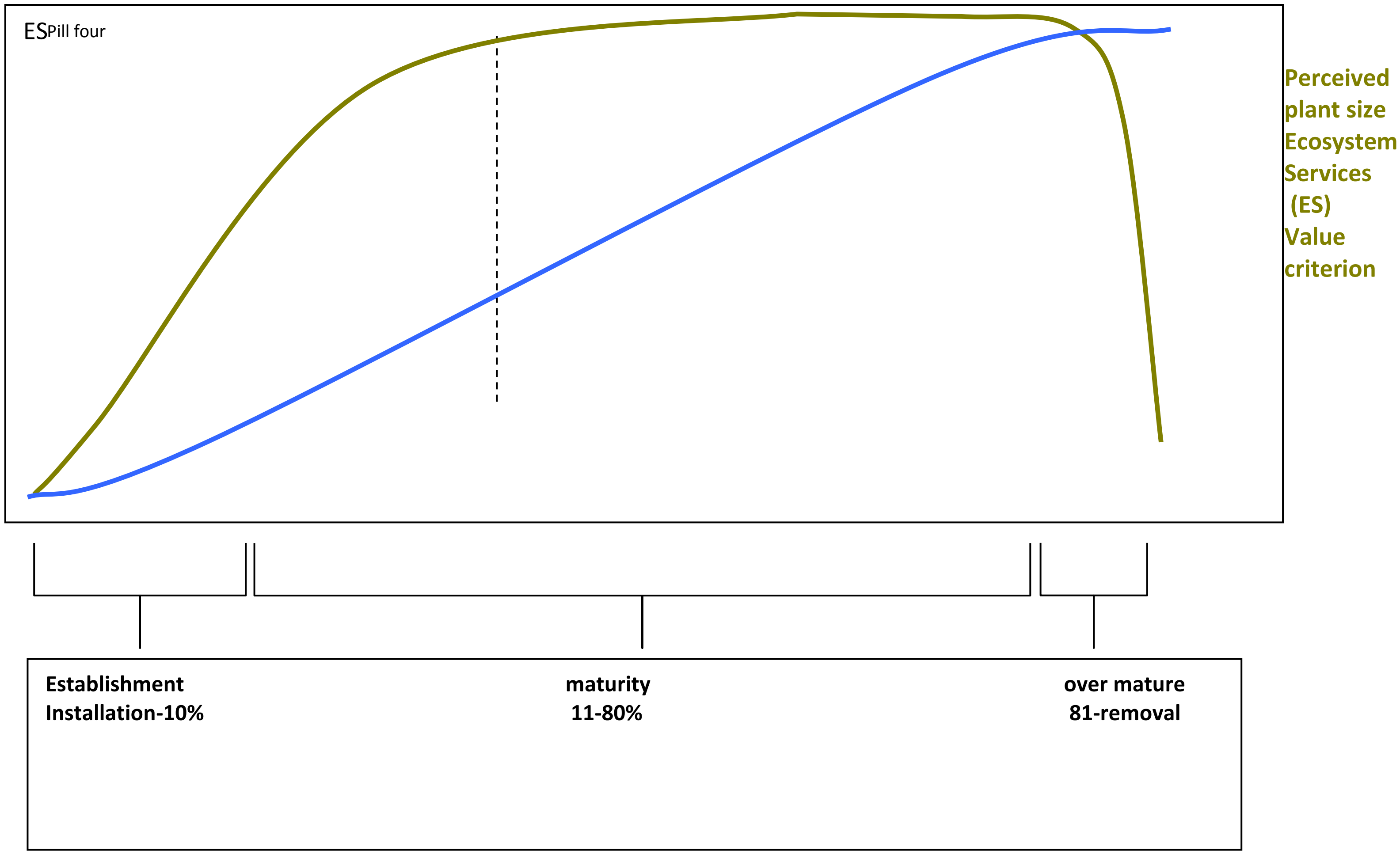

Figure 9:Cost-Benefit Curve for Urban Trees

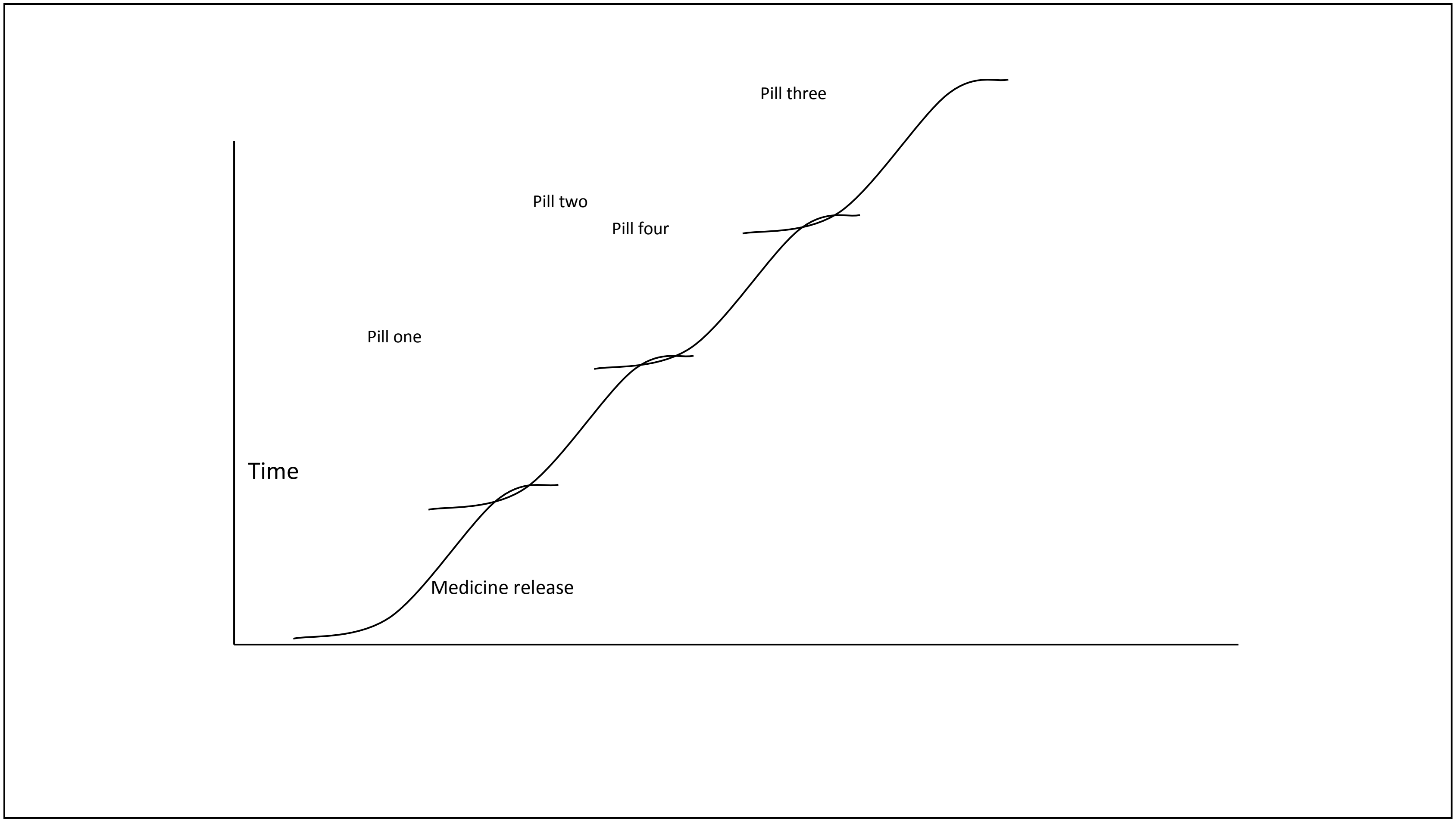

Figure 10:
Stable release of medicine

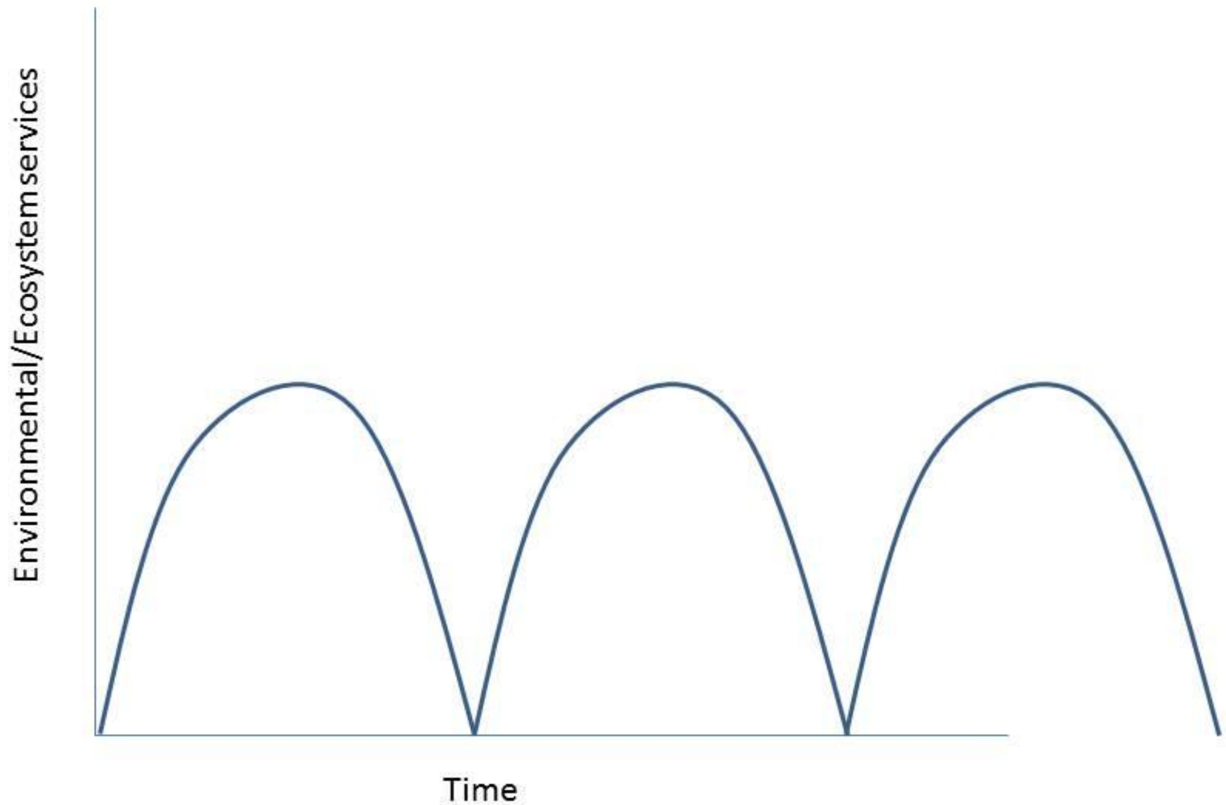

Figure 11: Traditional Harvest Interval for urban trees

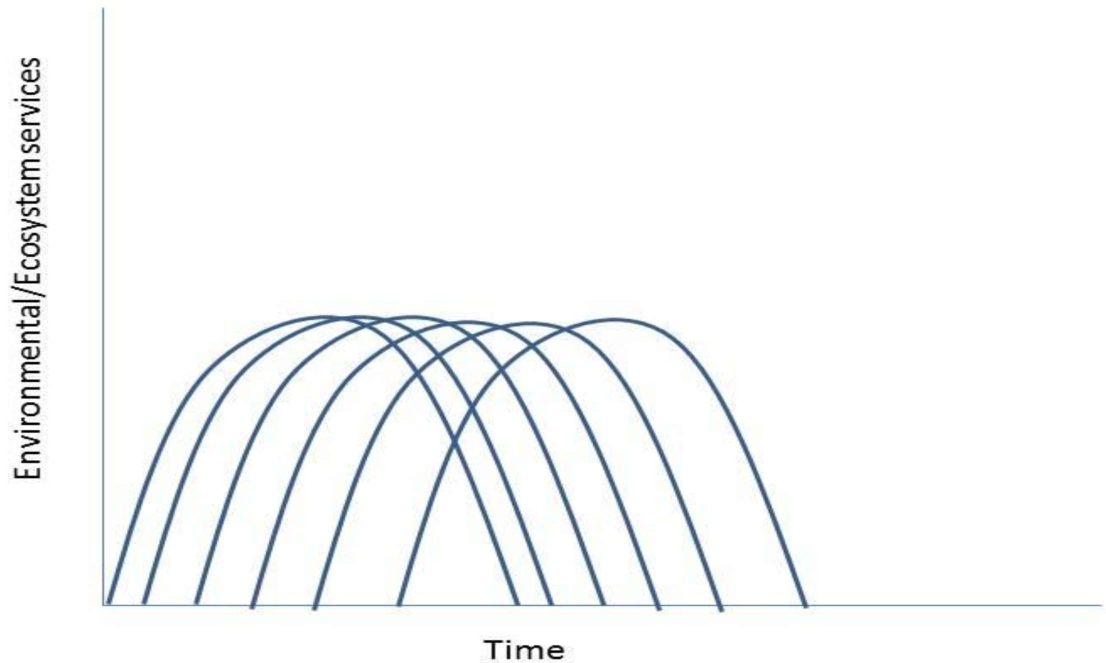

Figure 12: Urban harvest interval for stable environmental/ecosystem services.