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By

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Abstract

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Urban forests have aesthetic, environmental, human health, and economic benefits that motivate tree planting programs. Realizing these benefits depends on tree survival. Cost-benefit analyses for urban forest ecosystem services are sensitive to mortality rate assumptions and associated population projections. However, long-term mortality data is needed to assess the accuracy of these assumptions. Analytical tools from demography, such as life tables, mortality curves, and survival analysis, can improve our understanding of urban tree mortality. Demographic approaches have been widely used in forest ecology to quantify population dynamics and project future changes in wildland systems. However, to apply demographic techniques to urban forests, longitudinal data is needed, with repeated mortality observations on individual trees. In this dissertation, I analyzed five years of longitudinal data from two Northern California studies: street trees in Oakland and yard trees in Sacramento. These field projects are complemented by a conceptual overview of demographic approaches to urban tree mortality (Chapter 1), and an investigation of practitioner-based tree monitoring programs.

For the Oakland study (Chapter 2), I documented tree mortality and planting rates, net population growth, and assessed selected risk factors for survival. I monitored the entire street tree population in a small plot for five years after an initial inventory (2006). I adapted the classic demographic balancing equation to quantify annual inputs and outputs to the system, tracking pools of live and standing dead trees. There was a 17.2% net increase in live tree counts during the study period, with 3.7% overall annual mortality. However, population growth was constrained by high mortality of small/young trees. Size-based mortality rates followed a Type III curve, with highest mortality for small trees, and lower for mid-size and large trees. I used multivariate logistic regression to evaluate the relationship between 2011 survival outcomes and inventory data from 2006. Significant associations were found for size class, foliage condition, planting location, and a multiplicative interaction term for size and foliage condition.

For the Sacramento study (Chapter 3), I assessed tree losses during the establishment phase for a residential tree give-away program. A cohort of young trees distributed in 2007 was monitored for five years. I used Random Forests to identify the most important risk factors at different life

history stages, and survival analysis to evaluate post-planting survivorship. Analysis included socioeconomic, biophysical, and maintenance characteristics. In addition to field observations of tree planting status, survival, and maintenance, I also collected property ownership information (renter vs. owner-occupancy, homeowner change, and foreclosure) through the Multiple Listing Service and neighborhood socioeconomic characteristics from the U.S. Census. I found that 84.9% of trees were planted, with 70.9% survivorship at five years post-planting. Planting rates were higher in neighborhoods with higher educational attainment, and on owner-occupied properties with stable residential ownership. Five-year survival was also higher for properties with stable homeownership, as well as for tree species with low water use demand. When I incorporated maintenance characteristics from the first year of field observations, factors related to tree care were important to survival. Many residents did not adhere to recommended maintenance practices. These results illustrate the critical role of stewardship and consistent homeownership to young tree mortality on residential properties, and suggest that survival assumptions in urban forest cost-benefit models may be overly optimistic.

To learn more about practitioner-driven monitoring efforts, I surveyed 32 local urban forestry organizations across the United States about the goals, challenges, methods, and uses of their monitoring programs (Chapter 4). Non-profit organizations, municipal agencies, state agencies, and utilities participated. Common goals for monitoring included evaluating the success of tree planting and management, taking a proactive approach towards tree care, and engaging communities. Challenges included limited staff and funding, difficulties with data management and technology, and field crew training. Programs used monitoring results to inform tree planting and maintenance practices, provide feedback to individuals responsible for tree care, and manage hazard trees. Participants emphasized the importance of planning ahead: carefully considering what data to collect, setting clear goals, developing an appropriate database, and planning for funding and staff time. Urban tree monitoring partnerships between researchers and local organizations should be developed, with standardized protocols and clear research questions. Such partnerships would provide urban forestry professionals with improved mortality information to evaluate the success of planting programs, while expanding the data sets available to researchers. The Oakland and Sacramento studies (Chapters 2 and 3) offer examples of demographic approaches to urban tree mortality that can be replicated and expanded as more longitudinal data becomes available from both researchers and practitioners.

Dedication

To Lucy Ann Hafer, for lighting my day up with her smile,
and tolerating field work in utero

and to the late Fred Scatena, for helping me
find my footing as a researcher and urban ecologist

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Preface

My dissertation focused on a morbid topic: urban tree mortality. This is the central common theme through each of my chapters. My fascination with tree death has even earned me the nickname “Dr. Death” by my colleague Greg McPherson at the USDA Forest Service. While dwelling on the rates and causes of tree death may seem morose, my interest in monitoring tree mortality comes from a passion for cities, and a desire to enhance the urban forests in which we live. The data presented in this dissertation offers new insights into population dynamics of urban forest ecosystems, with results that have practical application for managers. The ultimate goal of my tree mortality research is to provide practitioners with quantitative analysis to guide the sustainable management of urban forests, ensuring that planting campaigns large and small make a lasting impact in the landscape.

This preface is a reflection on the process of implementing urban tree mortality field studies in collaboration with non-academic local partners. The two field studies summarized in my dissertation represent five years of field data collection: an annual census of all street trees in a small West Oakland plot (Chapter 2), and tracking a cohort of newly planted yard trees in suburban Sacramento County (Chapter 3). For both projects, I worked closely with non-profit organizations, and collected all the field data myself – with the assistance of wonderful UC Berkeley undergraduate students and local volunteers. The ideas below represent lessons learned from those experiences, based on countless hours meeting with project partners, collecting field data, and analyzing results.

Research partnerships are built on personal relationships

While research partnerships are often referred to as collaborations between organizations, individual people are at the heart of community-university partnerships. In the course of completing my dissertation, I have helped at numerous planting events in Oakland with Urban Releaf, and stayed as an overnight guest with staff from the Sacramento Tree Foundation, often bringing along homemade baked goods to share with my colleagues in both locations. Building friendly relationships was a critical first step towards developing meaningful research partnerships. The different perspectives and life experiences that each of us brought to the table often resulted in different opinions about how to proceed with the research projects, but we were consistently able to move forward with the research, discussing our different ideas from a place of friendship and trust. Building that camaraderie also involved recognizing our different day-to-day job needs: my need as a graduate student to pursue funding opportunities, complete my thesis on time, and present results in my manuscripts with dense, dry science writing, and my partners’ needs to showcase their research partnerships in the community, and get results translated into actionable conclusions.

Common vocabulary is essential for clear communication

Even the most basic vocabulary – such as “tree” and “dead” – can carry different meanings among both academics and practitioners. When I give presentations about the Sacramento shade tree study, I explain the different interpretations of survival and survivorship (using common definitions from demography) and survivability (the term used by my local partners). These terms are not synonymous. When I began discussing mortality with staff at the

Sacramento Tree Foundation, I quickly realized that our different ways of interpreting these survival-related words was leading to some misunderstandings and confusion. I found that even in the urban forestry literature, “mortality” and “lifespan” are not clearly defined. To advance our understanding of tree death, it is essential to have common vocabulary. Whenever possible, urban foresters should borrow terms from ecology and demography to describe studies on tree mortality and life expectancy. At the same time, it is my duty as a researcher to make sure that I clarify all my scientific terms and jargon when discussing tree mortality with my non-academic partners.

Accurate location information is the cornerstone of long-term tree monitoring

While this lesson may seem obvious to any ecologists engaged in long-term research, locational accuracy has not been a hallmark of urban forest inventories. While the street and yard trees in my research had addresses, this information alone was not often sufficient to find the trees. In both Sacramento and Oakland, I inherited data from my local partners as the baseline for beginning my own field work. In Sacramento, the baseline was the Sacramento Tree Foundation’s list of free shade trees distributed, and in Oakland, it was an inventory of street trees conducted by the USDA Forest Service with youth staff from Urban Releaf. Neither of these data sets were originally intended for monitoring. While most trees were easily located with the original records, approximately 5% of trees had locations that were not sufficiently described. Issues with reliably locating trees and plots were also raised in the survey responses from urban forestry organizations across the US (Chapter 4). One straightforward solution to tree location is labeling every tree in a study with a unique identification tag, yet city officials sometimes resist tags for aesthetic reasons, and have concerns about tag vandalism and girdling tags. Because tags were not a viable option in my study sites, I used a combination of reference objects records (e.g., tree distance and azimuth to a building corner), tree pictures, and an ordering system for street trees (detailed in Chapter 2, Appendix 1). Taking GPS coordinates for every tree is another obvious solution, but many practitioners lack GPS equipment, and in dense urban centers, tall buildings can interfere with readings. In conversations with other researchers and practitioners collecting long-term tree monitoring data, I have learned about a variety of different solutions to recording tree locations, and I am in the process of developing robust protocols on this issue with those colleagues.

Persistence, politeness, and appreciation lead to high participation rates

My field study in Sacramento and my questionnaire project of organizations across the US both depended upon participant recruitment and cooperation for adequate data collection. In both cases, I achieved high participation rates, largely due to simple persistence. Repeated phone outreach attempts – whether by phone, email, or in person – lead to excellent participation when conducted in a kind and friendly manner. In Sacramento, with the frequent changes in residents due to home sales and foreclosures, new residents were often surprised by my appearance at their door. These individuals may never have opened the letter sent to their address with background about our tree survival study. In most cases, a friendly greeting gained me access to the backyard. Some residents who saw me year after year looked forward to their annual visit from the “tree lady”, and asked about my progress in school. Some residents who were reluctant to let me in their yards the first time we met happily gave me permission to enter their back yards in later years, even if they were not home. My polite and accommodating attitude paid off with the continued participation of new residents, individuals who never intended to participate in the Sacramento Shade Tree Program. For the questionnaire study, I was asking municipal arborists and non-profit program managers to take time from their already over-worked schedules to complete a lengthy questionnaire. For that project, I began with a friendly phone call, listening to the triumphs and challenges of tree monitoring in each participant's city. For both of these projects, showing gratitude and establishing a respectful rapport was critical to securing high participation.

From initial inspirations to next steps: Building a national network for urban tree monitoring

My interest in urban tree mortality was originally sparked by discussions with staff at the Philadelphia Green program of the Pennsylvania Horticultural Society (PHS), several years before I came to UC Berkeley. Colleagues at PHS were wondering about the mortality rates of the trees they planted. They casually mentioned that street trees are said to have a seven-year average lifespan, which turned out to be more myth than reality. Estimating typical street tree mortality rates and life expectancy was the focus of my masters thesis at the University of Pennsylvania. Little did I know that this conversation with PHS staff would inspire my doctoral research, and launch me into a national urban tree monitoring project. The questionnaire study (Chapter 4) was conducted with colleagues from the Urban Tree Growth & Longevity Working Group, and it has lead us to form a national network of researchers and practitioners developing standardized protocols for urban tree monitoring. Our work builds on the collective experience and wisdom of our members, who include fellow graduate students, university and Forest Service researchers, municipal arborists, non-profit staff, and private arboriculture consultants. Our goal is to produce standards that have practical utility for local managers while also generating research-grade data for the study of urban tree populations. Continuing this work will be a part of my new job with the Forest Service; I have accepted a position with the new Philadelphia Field Station after my graduation from UC Berkeley. Not only is Philadelphia my hometown, but the Field Station is also located in the PHS office. I have the opportunity to work with the same colleagues whose observations and questions motivated my first urban forest project years ago. I intend to continue listening to and learning from local partners, gaining inspiration for research projects from their insights, and applying my Berkeley training to address our shared curiosity about urban tree death.

Chapter 1

Urban tree mortality rates: Applying concepts from demography and forest ecology

Abstract

Realizing the benefits of tree planting programs depends on tree survival. Cost-benefit analyses for urban forest ecosystem services are sensitive to mortality rate assumptions and associated population projections. However, long-term mortality data is needed to assess the accuracy of these assumptions. With more accurate population projections, urban forest managers can also plan for cycles of tree planting, death, removal and replacement to achieve stable canopy cover. Analytical tools from demography, such as life tables and mortality curves, should be used to improve our understanding of urban tree mortality rates and lifespans. Demographic approaches have been widely used in forest ecology to quantify population dynamics and project future changes in wildland systems. These methods can be adapted for urban trees. However, to build demographic models, we need longitudinal data, with repeated mortality and growth observations on individual trees. Urban tree monitoring partnerships between researchers and local organizations should be developed, with standardized protocols and clear research questions. Such partnerships would provide urban forestry professionals with improved mortality information to evaluate the success of planting programs, while expanding the data sets available to researchers.

Keywords

life cycle, long-term monitoring, matrix projection model, mean life expectancy, population half-life, survivorship

Introduction

Urban forests provide environmental, socioeconomic, and human health benefits (Dwyer et al. 1992; Nowak and Dwyer 2007; Tzoulas et al. 2007). Large-scale tree planting initiatives are underway in cities across the United States and around the world, aiming to provide these benefits to urban environments. Campaigns to plant a million trees in Los Angeles, CA; New York City (NYC), NY; Philadelphia, PA; and other cities have captivated the public's attention. These and other urban forestry initiatives are justified, in part, by models that quantify and monetize tree benefits, such as the i-Tree software suite (McPherson et al. 1999; Nowak and Crane 2000; itreetools.org; Silvera Seamens 2013).

However, realizing the public value of urban forest programs depends on tree survival. Cost-benefit assessments for urban forests are sensitive to assumed mortality rates (Hildebrandt and Sarkovich 1998; McPherson et al. 1998; McPherson and Simpson 2001; McPherson and Simpson 2003; McPherson et al. 2008; Morani et al. 2011). But are these assumed rates accurate? What proportion of trees will survive decades after planting, when the anticipated benefits are greatest? What are the implications of future tree death for managing the urban canopy, in terms of tree removal and replacement? These questions concern the rates and processes of urban tree death. Understanding urban tree mortality is critical to accurately modeling tree population changes over time and quantifying ecosystem services.

In this paper, we review urban tree mortality in relation to wildland tree mortality and demographic concepts. Specifically, we focus on the rates of urban tree death. We discuss analytical approaches common to plant demography and forest ecology that can advance our understanding of urban forest dynamics. We then outline an approach to studying urban tree mortality that relies on long-term monitoring to produce the data necessary for demographic analyses.

Why study urban tree mortality rates?

Altering the mortality rate assumptions in urban forest cost-benefit models drastically affects projected tree values (Hildebrandt and Sarkovich 1998; McPherson et al. 2008; Morani et al. 2011). Cost-benefit analyses have used a variety of mortality assumptions (Table 1). Urban tree mortality rates vary by size class (Nowak 1986; Nowak et al. 2004) and age (Richards 1979; Miller and Miller 1991). Some cost-benefit models accounted for such differences (McPherson 1994; McPherson et al. 1998; McPherson and Simpson 2001; McPherson et al. 2008; Nowak et al. 2002; Morani et al. 2011); others stated mortality assumptions without differentiation by tree size or age (McPherson et al. 1999). A few studies (Nowak et al. 2002; Morani et al. 2011) explicitly used mortality rates from field data (Nowak 1986), but many cost-benefit studies did not reference specific field data to support mortality assumptions. The field-based mortality data used in these cases (Nowak et al. 2002; Morani et al. 2011), and for others using the i-Tree Eco (formerly UFORE) model, originated with a single study of *Acer* spp. street trees in Syracuse, NY (Nowak 1986).

The mortality rates used in cost-benefit studies are essentially demographic population projections, although they are not generally labeled as such. In their simplest form, these predictions use a particular annual mortality rate to estimate survivorship several decades after planting (e.g., McPherson et al. 2008). In a more complex example, Morani et al. (2011) used mortality and growth rates to estimate population counts and tree sizes over 100 years for the MillionTreesNYC program. Mortality data can be used to construct life tables, survivorship curves, and other demographic tools. The application of these concepts to urban forests is the focus of our paper.

Before proceeding further, we must note that while urban forests are broadly defined to include all trees and vegetation in cities and urbanized areas (Konijnendijk et al. 2006), our review centers on intentionally planted trees in street and lawn settings. Urban tree mortality rates differ by planting location and land use (Nowak 1986; Nowak et al. 1990; Miller and Miller 1991; Nowak et al. 2004; Lawrence et al. 2012). City trees in remnant forests of native and naturalized species, or vacant lots, are probably closer to wildland forests in terms of demographic characteristics. In contrast, we focus on urban trees whose planting and removal are driven by human intervention. Our conclusions will be most relevant to heavily managed portions of the urban forest.

Throughout this paper, as we draw on demographic concepts used in natural, wildland forests, we must bear in mind essential differences in the life cycle for wildland trees and urban trees in heavily maintained landscapes. Wildland forests have natural processes of seed dispersal and germination, with large amounts of seeds produced, followed by competition among seedlings and saplings for light and other resources. Tree mortality in wildland forests is often a long process in which stressed individuals exhibit slow growth (Waring 1987; Pedersen 1998; Das et al. 2007), eventually succumbing to death through contributing factors such as wind, insects, or pathogens (Harcombe and Marks 1983; Franklin et al. 1987). Tree death as a cumulative process resulting from multiple factors was conceptualized by Manion (1981) as the decline disease spiral, later adapted by Franklin et al. (1987) as the mortality spiral.

In contrast to wildland forests, urban street and yard trees are typically produced by nurseries and planted as saplings in a sidewalk soil pit, planting strip, or manicured lawn. While urban environments pose many hazards to trees, such as compacted and contaminated soil (Craul 1999), construction (Hauer et al. 1994) and vandalism (Nowak et al. 1990), urban trees may also have advantages not present for wildland trees, including fertilizer, irrigation, and pest control (Harris et al. 2004), and reduced competition for light. Arborists aim to remove large unhealthy street and lawn trees before they die, in order to prevent property damage from falling limbs or infrastructure conflicts (Harris et al. 2004). Alternatively, some healthy city trees are removed due to human preferences or land use changes. Tree mortality and removal are thus central elements of urban forest management. Cycles of tree planting, death, removal, and replacement shape the structure and function of our urban forests, and affect the amount of canopy available to provide ecosystem services. To apply demographic concepts used in wildland forest ecosystems, we must make adaptations to suit the circumstances of heavily managed urban trees.

Demographic concepts

In discussing tree mortality, urban foresters should use concepts and terms established in population biology and demography: annual mortality rates, annual survival rates, life tables, life cycle diagrams, survivorship curves, mean life expectancy, and population half-life. These concepts are used by human demographers, actuaries, conservation biologists, and forest ecologists to quantify and project populations.

Mortality and survival information for urban trees can be broken down by age classes or size classes. Although forest ecologists typically use size classes (Harcombe 1987), we explore both approaches here, as both are relevant to urban forestry.

Age-based life tables, survivorship curves, and mortality curves

An age-based life table organizes survival information by age classes. In human demography, these life tables are broken down by sex, race, and other factors to assess mortality across different groups and calculate life expectancy. For urban trees, separate age-based life tables could be constructed for each species, land use type, or management regime, to quantify how these factors affect mortality trends.

As an example of an age-based life table for urban trees, we present survival data from trees in Sacramento County, CA (Chapter 3, Table 3a). This data is from a study that monitored a cohort of shade trees that were distributed by the Sacramento Tree Foundation and the Sacramento Municipal Utility District (SMUD). These trees were planted mostly in residential lawns. While the “clock” in an age-based life table typically begins at birth, in the context of urban trees, the “clock” begins at time of planting. Notation and formulae for the age-based life table, as applied to an urban tree planting cohort, are summarized in Table 2. The time interval between age classes in this example, and throughout the rest of our discussion of age-based life tables, is set to one year, meaning that p_x and q_x are interpreted as annual survival and mortality at age x, respectively. However, age-based life tables may also be constructed with different age intervals, or varying time period intervals, which requires slight changes to the equations in Table 2 (Carey 1993).

Our Sacramento example (Table 3a) tracks a cohort for the first few years following planting, and illustrates the key components of an age-based life table. Note that the first three columns (K_x, D_x, W_x in Table 3a) contain raw data; the other columns are demographic terms calculated from the data. Survivorship l_x is cumulative from the time of planting to x, while annual survival rate p_x and annual mortality rate q_x are defined by a particular time interval x to x +1. A graph of $\ln(l_x)$ vs. x is called a survivorship curve (Figure 1, where $\ln(l_x)$ is the natural logarithm of survivorship), and the shape of this curve depends on how q_x changes over time. The graph of q_x vs. x is called the mortality rate curve. In the Type I survivorship curve, annual mortality is highest for old individuals, giving survivorship a convex shape. In Type II, annual mortality is constant, and $\ln(l_x)$ vs. x is a straight line with negative slope. In Type III, annual mortality is highest for young individuals, leading to a concave survivorship curve. Another possible shape is the rotated sigmoid survivorship curve, and corresponding U-shaped (or bathtub-shaped) mortality curve (Figure 1, after Harcombe 1987). These represent high mortality for both old and young individuals, with low mortality rates in between. Survivorship curves are conventionally ln transformed in order to depict more clearly changes in the proportion of individuals surviving over time.

A complication in our Sacramento example, and in demographic studies in general, is that the survival fate of all individuals could not be determined. We have only partial survival data for a few of the Sacramento lawn trees, in situations where we could not secure permission to access the back yard every year. These incomplete observations are referred to as censored data. Specifically, these trees have been lost to follow-up: we do not know their survival status after the properties became inaccessible. This situation is called right censoring. There are many different approaches to censored data in survival analysis (Klein and Moeschberger 1997). We employed a simple method to compensate for censoring (Y_x in Table 2) that depended on two assumptions: the causes of mortality and censoring were independent, and censoring times were uniformly distributed over the interval x to x +1. In urban forestry studies like ours with private properties and incomplete observations, the analytical methods must ensure that censored data do not bias calculated vital rates (Appendix 1).

When annual mortality is constant (Type II), it can be calculated from survivorship using the following equation:

$$q_{annual} = 1 - \left(K_x / K_0 \right)^{1/x}$$

(eqn. 10, after Sheil et al. 1995, eqn. 6)

where K_x and K_0 are the population sizes at the beginning and end of time interval x. The fraction K_x / K_0 is simply cumulative survivorship from the time of planting (Table 2, eqn. 5). Stated equivalently, when mortality is constant, annual survival and survivorship are related by $p_{annual} = (l_x)^{1/x}$. Note that this relationship does not compensate for censoring. With the constant mortality assumption, typical lifespan can be easily quantified. The mean life expectancy e_0 from the time of planting, $x = 0$, is:

$$e_0 = -1 / \ln\left(p_{annual} \right)$$

(eqn. 11, after Seber 1982, eqn. 1.3)

The population half-life $t_{0.5}$, or the time at which half the planting cohort has died (i.e., survivorship is 50%), is:

$$t_{0.5} = \ln\left(0.5\right) / \ln\left(p_{annual} \right)$$

(eqn. 12, after Sheil et al. 1995, eqn. 10)

Estimating the mean life expectancy and population half-life for urban trees would help managers anticipate future tree losses. While the "average lifespan" has been frequently discussed in urban forestry (Moll 1989; Skiera and Moll 1992), the term is not clearly defined, and rarely based on field data. Nowak et al. (2004) estimated a 15-year average lifespan using observed mortality rates for different tree size classes in Baltimore, MD, yet the exact formula used to calculate lifespan was not specified. For street trees, a recent meta-analysis (Roman and Scatena 2011) estimated annual survival rates and demographic lifespan metrics (Table 4). Survivorship data from 11 previous studies was pooled in a regression analysis, with the assumption of constant mortality, to estimate annual survival. A similar approach – pooling survivorship data from cohorts in different years – was also applied to street trees in Philadelphia, PA (Roman and Scatena 2011; Table 4).

When mortality is not constant over time, as is probably the case with urban trees (Richards 1979; Miller and Miller 1991), mean life expectancy can still be calculated, but the cohort life table must be completed to the oldest age classes. To determine life expectancy, several additional columns are added: L_x, T_x, e_x (Table 2). For an individual that has reached age x, the average age of death is the current age plus the expectation of death for that age class, $x + e_x$. Classic examples calculating e_x are given in Carey (1993) and Seber (1982), which focus on animal populations. The expectation of life when $x = 0$ is considered the life expectancy at birth (or in the case of urban trees, time of planting), similar to the mean life expectancy defined above using the constant mortality assumption (eqn. 11). Calculating e_0 using eqn. 9 (Table 2) when annual mortality is not constant requires having life table data for the cohort until the last possible year of life.

Unfortunately for urban foresters and wildland forest ecologists, constructing complete cohort life tables for trees is generally not feasible. Trees are such long-lived organisms that following a cohort until the last possible year of life is beyond a researcher's own lifespan. Forest ecologists typically use size classes, rather than chronological age, to build life tables and mortality curves. However, for urban trees, age-based life tables and survivorship curves still hold relevance. Built into the mortality assumptions of many urban forestry cost-benefit analyses (Table 1) are estimates of annual mortality rates and resulting survivorship. As an example, we used the assumed high and low mortality scenarios in the Million Trees Los Angeles program (McPherson et al. 2008) to construct age-based life tables (Table 5) and survivorship curves (Figure 2). The benefits for the Los Angeles example were evaluated 35 years after planting begins. Most tree losses were assumed to occur in the establishment phase (Richards 1979), the first 5 years after planting (McPherson et al. 2008). The low mortality scenario was projected to yield \$1.95 billion in benefits, while the high mortality would reduce benefits by 32%, to \$1.33 billion (McPherson et al. 2008). These mortality assumptions could be checked by monitoring a cohort after planting, which would also yield essential information about mortality trends during the establishment phase across different species, land uses, and other risk factors. Even if the age-based life tables and survivorship curves generated by field data only extend to the first five to ten years after planting, they would still provide a useful framework for urban forest managers to evaluate program success and make management decisions.

Urban forest managers could plan ahead for replacement planting projects at the population half-life. Using the million tree campaigns as examples, we could ask: when will half the trees be dead? The population half-life for the Million Trees Los Angeles benefits projections (McPherson et al. 2008) can be calculated using the assumed annual survival rates (Table 5). For example, in the high mortality scenario, determine the half-life by solving for *x*:

$$l_x = 0.5 = (0.95)^5 (0.98)^{x-5}$$

With the high mortality scenario, half-life would be 27 years, and with the low mortality scenario, half-life would be 133 years (Figure 2). In contrast, the half-life in the street tree meta-analysis was 13-20 years (Roman and Scatena 2011; Table 4). Although the Los Angles program will include a range of planting locations (with presumably better survival than street trees), a half-life of 133 years seems overly optimistic. In the Sacramento Shade Tree Program – which primarily involves yard trees maintained by residents – SMUD has observed that 54% of trees distributed remained alive at 5 years, and 43% at 10 years (Lindeleaf 2007; Appendix 1). Determining whether the population half-life for large-scale planting programs is 10, 20, or 100 years has significant implications for urban forest management. For managers to anticipate the planting levels required for consistent canopy cover, it is essential to have accurate mortality rates.

<u>Stage-based life tables and life cycle diagrams</u>

In contrast to cohort life tables, stage-based life tables organize mortality information by life stages that are biologically meaningful to the study system. For trees, the stages are often defined by diameter at breast height (DBH) size classes (Harcombe 1987). As time moves forward, trees in a stage-based life table can advance to the next size class (G_x), remain in the same size class (R_x), or die (Tables 3b, 3c). This is represented graphically in a stage-based life cycle diagram for urban trees (Figure 3b) and wildland trees (Figure 3c), contrasted with an age-structured life cycle diagram (Figure 3a). The survival rate for a particular size class is $G_x + R_x$. Mortality M_x is defined as the proportion of individuals dying in size class *x* during the time interval:

$$M_x = 1 - G_x - R_x$$

(eqn. 13, after Harcombe 1987)

The time interval length for a stage-based life table is specific to each study. Thus the interpretation of M_x might be annual mortality rate, or some other interval specified by a particular study. In an example for a wildland forest species (Table 3c), *Abies concolor* in the Sierra Nevada, California, a five-year interval was used (van Mantgem and Stephenson 2005).

Stage-based life tables and life cycle diagrams for wildland tree populations include a fecundity rate or recruitment rate F_x, which is the number of offspring produced in the time interval by size class x. In the *A. concolor* example, F_4 is the number of new trees observed in the smallest size class at the end of the five-year period. Note that only trees in the largest size class were assumed to contribute to recruitment in this study (van Mantgem and Stephenson 2005; Table 3c, Figure 3c). For our urban tree age- and stage-structured models (Tables 3a, 3b, Figures 3a, 3b), there is no natural recruitment. Instead, we included a planting rate $Plant_x$, representing trees added into the system in the smallest age or size class.

Matrix models

Matrix projection models incorporate the information contained in the life cycle diagram and life table to analyze population characteristics and predict future changes (Caswell 2001). The general form of these models is:

$$\mathbf{n}(t+1) = \mathbf{A}\mathbf{n}(t)$$

(eqn. 14, after Caswell 2001, eqn. 2.3)

In the case of the stage-structured system for trees, the vector **n** (t) is the abundances of trees in different size classes at time t, and **A** is the transition matrix that describes the probability that each size class will contribute at the next time step (Harcombe 1987). The population growth rate λ is the dominant eigenvalue of the transition matrix **A** (Caswell 2001). For example, the stage-based life table for *A. concolor* has all the elements for the transition matrix (van Mantgem and Stephenson 2005). To use stage-based matrix models, both mortality rates and growth rates are required. Matrix models can also be applied to age-based life tables, which do not require growth rates.

For forest ecosystems, matrix models have been used to predict future size distributions and population trends (van Mantgem and Stephenson 2005), evaluate sustainable harvesting (Olmsted and Alvarez-Buylla 1995; López et al. 2007), analyze stand dynamics in relation to environmental stochasticity (Lytle and Merritt 2004), evaluate changes in population growth rate with different vital rates (Enright and Watson 1991; Zuidema and Franco 2001), determine the effects of pollution on successional patterns (McBride and Laven 1999), and assess extinction risk and population trends for species of conservation concern (Schwartz et al. 2000; Kohira & Ninomiya 2003; Kwit et al. 2004; Chien et al. 2008). While the stage-based life cycle for wildland forests (Figure 3c, eqn. 14) might be appropriate for urban trees in remnant parks, where natural seedling establishment occurs, heavily managed urban sites require a different approach. Instead of a fecundity rate, our urban tree life cycles (Figures 3a, 3b) have a planting rate $Plant_x$. Newly planted trees are input from outside the system, and there is no natural recruitment. The classic matrix model (eqn. 14) represents a closed system, while our urban tree life cycle (Figures 3a, 3b) represents an open system. Thus, the application of matrix modeling to urban trees will require changes in both conceptualization and calculation. Nevertheless, determining the appropriate age and size distribution for urban forest stability is a central management issue (Richards 1979; Richards 1983), and these models provide an analytical framework to assess the influence of planting and mortality rates on urban forest structure.

Size-based mortality curves

Forest ecologists have observed Type III or U-shaped size-based tree mortality rate curves (Figure 1), with different results from various forest systems and species (Buchman 1983; Harcombe and Marks 1983; Buchman and Lentz 1984; Buchman 1985; Harcombe 1987;

Monserud and Sterba 1999; Lorimer et al. 2001; Umeki 2002; Coomes and Allen 2007; Metcalf et al. 2009). In a recent large study in forests across the eastern United States with over 430,000 tree records (Lines et al. 2010), all 21 species examined were found to have U-shaped mortality curves. Regardless of mortality curve shape, forest ecologists have generally reported very low rates of annual mortality for canopy trees, typically 1-3% or even less (e.g., Harcombe and Marks 1983; Franklin et al. 1987; Condit et al. 1995; Lorimer et al. 2001), absent catastrophic disturbances (Lugo and Scatena 1996).

Urban trees may follow a U-shaped mortality curve by size class, similar to most tree species in wildland forests. We adapted the annual mortality rates reported for Baltimore, MD by Nowak et al. (2004) into a mortality curve (Figure 4). Annual mortality was highest for small trees 0-9cm DBH (9.0%), lowest for trees 30.6-45.7cm DBH (0.5%), then rose again for larger trees, with some fluctuations (1.8-3.3%). While the general U-shape is similar to trees in wildland environments, with both having ~1% mortality for mid-sized trees (i.e., the bottom of the U-shape), there are essential differences in the mortality process that affect rates of death. Large urban trees are sometimes cut down and removed before they fully succumb to pathogens and stresses. These hazard removals may increase mortality rates for large DBH urban trees, and cause a more distinct up-swing at the tail of the U-shape, in comparison to wildland trees. On the other end of the mortality curve, very small trees in the urban landscape are typically newly planted, with higher mortality rates during the establishment phase (Richards 1979; Miller and Miller 1991). In contrast, some urban trees at risk for death recover from damage and disease through human intervention, which may lead to lower mortality rates compared to wildland trees. However, given the scarcity of data on urban tree mortality rates categorized by size class, these comments on the shape of the urban tree mortality curve remain speculative.

When comparing the mortality rate curves of urban and wildland forests, we must also bear in mind that studies vary in their delineation of size classes. Some wildland tree demography studies have a lower cut-off that would exclude newly planted urban trees (e.g., only trees >2cm were used in Lorimer et al. 2001), while others include young seedlings (e.g., 3 year old seedlings in Cleavitt et al. 2011) that would be in the nursery production stage for urban forests. DBH classes in wildland tree demography are often set to represent canopy position, but these positions (e.g., sapling, understory, co-dominant, dominant) do not carry the same meaning in urban street and lawn environments. Size-based mortality curves and life tables will aid our understanding urban tree mortality trends, but as with other aspects of forest demography, we must adapt these tools to urban systems.

The need for urban tree monitoring and longitudinal data

The specific type of monitoring we discuss in this paper concerns longitudinal data: repeated observations on the same individual trees over time. Longitudinal urban tree studies would provide mortality and growth rates to build life tables, survivorship and mortality curves, and matrix models. Other types of long-term monitoring data are also useful to understand changes in the urban landscape over time, such as canopy cover, land use, and program operations. However, only data tracking the fate of individual trees is suitable for the demographic analyses discussed here.

While urban forest inventory systems have been developed to provide managers with quantitative data on the composition and structure of the urban canopy (Nowak and Crane 2000; McPherson and Simpson 2002; McPherson et al. 2005), monitoring involves more than a one-time inventory (Baker 1993). When repeated observations are intended, researchers encounter issues that extend beyond the needs of a single inventory, such as reliably locating plots and individual trees during subsequent visits (Chapter 4), and determining the appropriate observation intervals. Collecting data over many years also requires considerable advance planning for staffing and funding (Caughlan and Oakley 2001). Although urban forest researchers have recognized the need for long-term data (Baker 1993; McPherson 1993), we do not yet have coordinated programs to conduct longitudinal studies.

Lessons from ecological monitoring in other systems

Ecologists have developed strategies and tools for effective monitoring, and the lessons learned from these projects provide guidance for monitoring efforts in urban forestry. Lindenmayer and Likens (2010) argued that monitoring programs should be driven by conceptual models of the study system with clear research questions and rigorous study design. Other attributes of effective monitoring are dedicated leadership, strong partnerships among scientists, resource managers, and policy-makers, frequent use of the collected data, and an adaptive monitoring framework that responds to new technologies and research questions (Lindenmayer and Likens 2009; Lindenmayer and Likens 2010). Urban forest practitioners who collect monitoring data also stressed the importance of clear objectives and uses of the data (Chapter 4).

There have been several long-term monitoring programs in forest ecosystems in the United States, including the Forest Inventory and Analysis (FIA) program of the USDA Forest Service and Long-Term Ecological Research (LTER) sites sponsored by the National Science Foundation. Globally, the Center for Tropical Forest Science is a network of dozens of tropical and temperate plots, all following the same methods to re-census trees every five years (Condit 1995; www.ctfs.si.edu). The FIA program serves as a census for forest ecosystems in the United States (Smith 2002; fia.fs.fed.us), with recent integration of the Forest Health Monitoring program (fhm.fs.fed.us) and annual field measurements (McRoberts et al. 2005) to generate longitudinal data. Many studies of tree mortality have used FIA data (e.g., Woodall et al. 2005; Lines et al. 2010). Although these programs focus primarily on non-urban forests, the methods and analytical tools can be adapted to urban systems. This is already happening with FIA urban pilot programs (Cumming et al. 2001; Cumming et al. 2007). The LTER sites (which are not exclusive to forest systems) were developed with a recognition that many ecological phenomena operate over decades, and longer, requiring long-term investment in data collection (LTER Network 2011). There are two LTER sites in urban environments: Baltimore, MD (beslter.org) and Phoenix, AZ (caplter.asu.edu). In Phoenix, annual tree surveys are already underway (caplter.asu.edu). To gather comprehensive longitudinal data on urban trees, it is essential that urban foresters and urban ecologists coordinate our efforts, with clear research questions and strong partnerships, learning from the experiences of forest ecologists working in long-term monitoring programs.

Looking ahead: opportunities for future research

To develop a network of longitudinal urban forest studies, researchers should collaborate with municipal foresters and non-profit organizations. Some local urban forestry organizations in the United States already gather mortality data on tree planting programs (Chapter 4). These studies rarely result in peer-reviewed journal articles (e.g., Lu et al. 2010), but sometimes lead to internal program reports (e.g., Sullivan 2004; Gates and Lubar 2007; Lindeleaf 2007). To promote data sharing among professionals and researchers, and to advance monitoring efforts already underway, standardized protocols for urban tree monitoring should be developed (Leibowitz 2012). Standardization would enable comparisons across and within cities, and avoid duplicated efforts to develop monitoring methods. Such partnerships would provide urban forestry professionals with improved mortality information to evaluate the success of planting and management programs, while expanding the data sets available to researchers.

However, there are challenges and limitations to long-term urban tree monitoring and demographic approaches. Gathering longitudinal data over many years requires adequate funding, continuity in leadership, and robust study design for statistical analysis. To facilitate permanent plots, the initial inventory or tree planting information should include detailed site maps, geospatial coordinates, and/or tagged tree identification numbers. Additionally, matrix population models and other demographic tools are traditionally applied to a single species, in systems with natural reproduction. Our urban street and lawn tree examples lump many species together, and are open systems with human-driven planting, not natural reproduction. Although we can adapt demographic tools, we must be cognizant of the different assumptions and interpretations for urban trees. Nevertheless, by borrowing concepts from other disciplines, urban forestry gains clearly defined terms and well-established methods for quantifying mortality.

As we gather more long-term urban tree monitoring data, we will be better equipped to answer key questions about mortality using demographic approaches: (1) How do observed annual mortality rates, growth rates, and long-term survivorship compare to assumed rates in cost-benefit models? (2) Based on observed mortality and growth rates, what level of annual planting is required to balance typical mortality, and maintain a stable canopy cover? (3) What is the shape of the mortality curve for urban trees, and how does it vary across species, land uses, planting locations, and management regimes? The following dissertation chapters begin to address the questions raised in this review with primary field studies in two California locations: West Oakland (Chapter 2) and Sacramento County (Chapter 3).

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Table 1. Urban tree mortality rate assumptions used in cost-benefit analyses. Most studies listed determined long-term cumulative mortality to time *x* based on differential mortality rates by age classes. Some also used size and condition classes. These studies included urban trees across a variety of land use types and planting locations. Most studies projected future benefits for newly planted trees, but one example (Nowak et al. 2002) assessed benefits based on an inventory of current tree stocks.

Location	Time interval *x* (yrs)	Cumulative mortality to time x (cumulative survivorship l_x)	Notes	Citation
Los Angeles, CA			Trees planted over 5 years;	McPherson et al. (2008)
high mortality	35	56% (44%)	High mortality scenario: 5% annual mortality years 1-5, and 2% annual mortality thereafter;	
low mortality	35	17% (83%)	Low mortality scenario: 1% annual mortality years 1-5, and 0.5% annual mortality thereafter	
Modesto, CA			1.4% annual mortality averaged across all age classes	[1] McPherson et al. (1999)
Sacramento, CA			Shade trees	[2] Hildebrandt & Sarkovich (1998)
high mortality	30	42% (58%)		
low mortality	30	30% (70%)		
Sacramento, CA	30	67% (33%)	For 100 trees, 21 died by year 5, and 1 additional tree died annually thereafter	McPherson et al. (1998)
CA urban forests	15	25% (75%)	3% annual mortality years 1-5; 1% annual mortality thereafter	McPherson and Simpson (2001)
Chicago, IL	30	35% (65%)	Variable mortality for different tree locations	McPherson (1994)
Brooklyn, NY			Annual mortality for different condition classes: dead 100%; dying 50%; critical13.08%; poor 8.86%; fair 3.32%; good-excellent 1.92% for 0-7.6 cm. DBH, 1.46% for >7.6 cm DBH	[3,5] Nowak et al. (2002)
New York City, NY			Trees planted over 10 years;	[4,5] Morani et al. (2011)
4% avg. ann. mort.	100	95.4% (4.6%)	Annual mortality for different DBH size classes proportional to: 2.9% for 0-7 cm; 2.2% for 8-15 cm; 2.1% for 16-46 cm; 2.9% for 47-61 cm; 3.0% for 62-76 cm; 5.4% for >77 cm	
6% avg. ann. mort.	100	99.1% (0.9%)		
8% avg. ann. mort.	100	99.8% (0.2%)		

[1] McPherson et al. (1999) gave a single annual mortality rate across age classes, and explained that Modesto, CA mortality rates were “supplied by the city”.
[2] Hildebrandt and Sarkovich (1998) illustrated graphically that young trees have higher mortality rates than established trees, but annual mortality rate values for different age classes were not provided.
[3] Nowak et al. (2002) used varying mortality rates based on size and condition class, and used these rates to calculate carbon release from dead trees over one year.
[4] Morani et al. (2011) used varying mortality rates for different size classes to project population size, in order to simulate air pollution removal throughout a 100-year time horizon. Mortality rates for individual trees changed as time passed and trees grew into a different size class. Mortality rates were set to average at 4%, 6%, or 8%, with size-specific mortality rates proportional to those reported in the table.
[5] Morani et al. (2011), Nowak et al. (2002), and others using the i-Tree Eco (formerly UFORE) model referenced Nowak (1986) observed street tree mortality in Syracuse, NY, with rates differentiated by size and health condition classes.

Table 2. Notation and formulae for the age-based urban tree life table. Notation mostly follows Carey (1993), with censoring terms from Klein and Moeschberger (1997).

Term	Definition
x	age, in this case measured in years, starting from time of planting ($x = 0$)
K_x	number of individuals alive at beginning of interval x to $x+1$
W_x	number of individuals censored (lost to follow-up) during the interval x to $x+1$
Y_x	number of individuals at risk of death during the interval x to $x+1$, assuming that censoring times are uniformly distributed during the interval $Y_x = K_x - (W_x/2)$ (eqn. 1, after Klein and Moeschberger 1997, p 138)
D_x	number of deaths in the interval x to $x+1$
q_x	proportion dying from x to $x+1$, annual mortality rate when there is no censoring: $q_x = D_x/K_x$ (eqn. 2) to compensate for censoring: $q_x = D_x/Y_x$ (eqn. 3)
p_x	proportion surviving from x to $x+1$, annual survival rate $p_x = 1 - q_x$ (eqn. 4)
l_x	proportion of the cohort surviving from planting to age x, commonly called survivorship to age x; $l_0 = 1$ by definition when there is no censoring: $l_x = K_x/K_0$ (eqn. 5) to compensate for censoring: $l_x = (l_{x-1})(p_{x-1})$ $= \prod_{i=1}^{x} \left(1 - (D_{i-1}/Y_{i-1})\right)$ (eqn. 6, after Klein and Moeschberger 1997, eqn. 5.4.1)
L_x	number of years lived by the average individual in the cohort from x to $x+1$, called the cohort person-years in human demography $L_x = (l_x + l_{x+1})/2$ (eqn. 7, after Carey 1993 eqn. 2-3a)

T_x total number of years remaining for the average individual from age x to the last possibly year of life w

$$T_x = L_x + L_{x+1} + L_{x+2} + \cdots + L_w$$

$$= \sum_{i-x}^{w} (L_i)$$

(eqn. 8,after Carey 1993 eqn. 2-4)

e_x expectation of life at age x

$$e_x = T_x / l_x$$

(eqn. 9, after Carey 1993 eqn. 2-5)

Table 3. Life table format. (a) An age-based life table from a cohort of lawn trees in Sacramento County, CA (Chapter 3; Appendix 1). The time of planting is x=0. See Table 2 for notation and definitions. (b) An idealized stage-structured life table for urban trees with tree diameter size classes as stages. Time interval not specified. G_x = proportion in size class x growing into the next size class; R_x = proportion remaining in the same size class; $Plant_x$ = number of trees planted in a size class. (c) A stage-structured life table for *Abies concolor* in the Sierra Nevada (Hodgedon Meadows), adapted from van Mantgem & Stephenson (2005). Time interval was five years. F_x = rate of new trees <5cm DBH observed at the end of the time interval; only the largest size class contributes to recruitment. Notation in stage-structured life tables follows Harcombe (1987). See Figure 3 for associated life cycle diagrams.

(a)

Year	**Age, x**	**# alive at beginning of interval, K_x**	**# deaths in interval, D_x**	**# censored during interval, W_x**	**# at risk during interval, Y_x**	**Annual mortality rate, q_x**	**Annual survival rate, p_x**	**Survivorship to age x, l_x**
2007	0	409	49	0	409	0.120	0.880	1.000
2008	1	360	20	2	359	0.056	0.944	0.880
2009	2	338	17	8	334	0.051	0.949	0.831
2010	3	313	13	3	311.5	0.042	0.958	0.789
2011	4	297	10	6	294	0.034	0.966	0.756
2012	5	281	n/a	n/a	n/a	n/a	n/a	0.730

(b)

Size class x	**Proportion growing to the next size class,** G_x	**Proportion remaining in same size class,** R_x	**Number of trees planted,** $Plant_x$
0	G_0	R_0	$Plant_0$
1	G_1	R_1	
2	G_2	R_2	
3	G_3	R_3	
4		R_4	

(c)

Size class x	**DBH range (cm)**	**Proportion growing to the next size class over 5 years,** G_x	**Proportion remaining in same size class over 5 years,** R_x	**Fecundity,** F_x
0	<5.0	0.082	0.855	
1	5.1-10.0	0.123	0.828	
2	10.1-20.0	0.107	0.866	
3	20.1-40.0	0.072	0.913	
4	>40.0		0.973	4.081

Table 4. Estimated annual street tree survival rates, mean life expectancy, and population half-life. Adapted from Roman & Scatena (2011). Ranges in values are due to differences in estimation with and without weighting by sample size. Constant mortality was assumed. The terms annual mortality rate (with constant mortality assumption), mean life expectancy, and population half-life are defined in the text (eqns. 10, 11, 12).

	Annual mortality rate, q_{annual} (p_{annual})	**Mean life expectancy, e_0 (years)**	**Population half-life, $t_{0.5}$ (years)**
Meta-analysis of 11 previous studies, survivorship data from 1-66 years after planting in different cities	0.035-0.051 (0.949-0.965)	19-28	13-20
Philadelphia field survey, survivorship 2-10 years after planting	0.034-0.045 (0.955-0.966)	22-29	15-20

Table 5. Age-based life table for annual mortality rate assumptions used in cost-benefit analysis for the Million Trees Los Angeles program. Adapted from McPherson et al. (2008). The terms survivorship, annual survival rate, and annual mortality rate are defined in the Table 2 (eqns. 2, 4, 5). For years 5 and beyond, annual mortality remains constant at either 2% (high mortality scenario) or 0.5% (low mortality scenario).

	High mortality scenario			Low mortality scenario		
Age, *x*	**Survivorship to age *x*, l_x**	**Annual survival rate, p_x**	**Annual mortality rate, q_x**	**Survivorship to age *x*, l_x**	**Annual survival rate, p_x**	**Annual mortality rate, q_x**
0	1.000	0.950	0.050	1.00	0.990	0.010
1	0.950	0.950	0.050	0.990	0.990	0.010
2	0.903	0.950	0.050	0.980	0.990	0.010
3	0.857	0.950	0.050	0.970	0.990	0.010
4	0.815	0.950	0.050	0.961	0.990	0.010
5+	0.774	0.980	0.020	0.951	0.995	0.005

Figure 1. Survivorship and mortality curve shapes. The classic Type I, II, and III survivorship curves and associated mortality rate curves. For age-based cohort mortality, survivorship l_x is the proportion surviving to age x, and mortality rate q_x is the proportion of individuals dying in the interval x to $x+1$. For size-based mortality, M_x is the proportion of individuals dying in size class x (after Harcombe 1987), and the survivorship curve is not relevant.

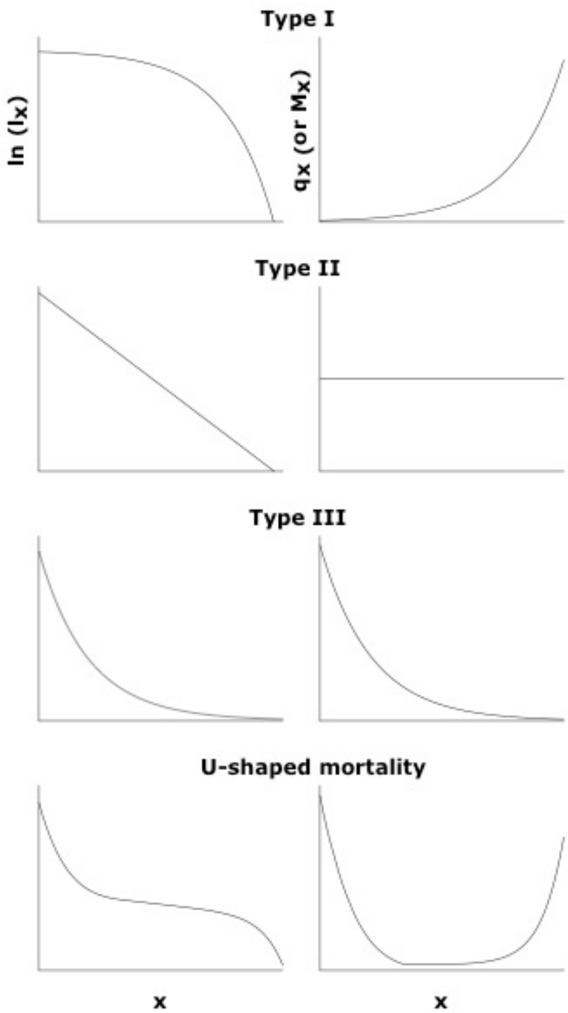


Figure 2. Assumed survivorship curves used in cost-benefit analysis for the Million Trees Los Angeles program. Adapted from McPherson et al. (2008). See Table 4 for annual survival rates.

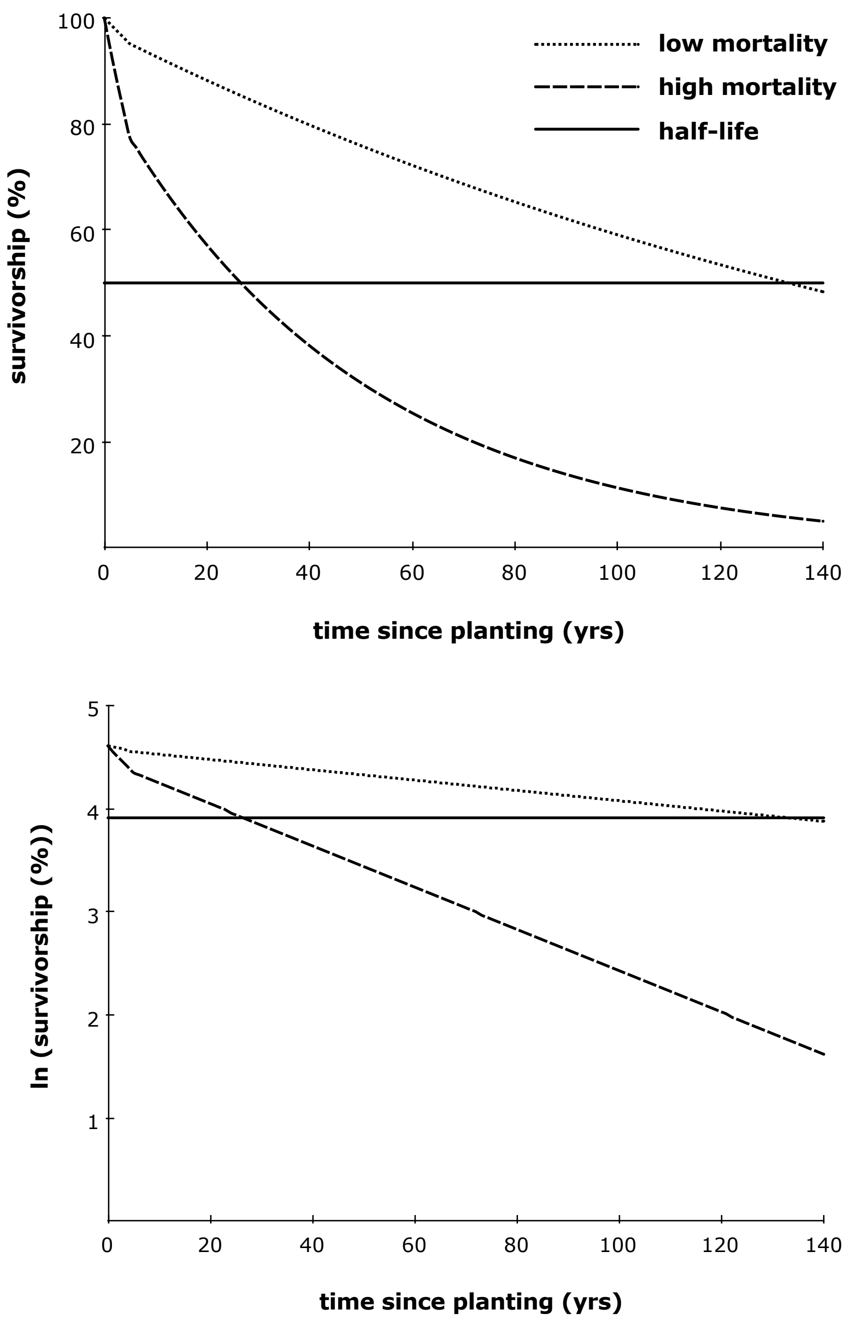
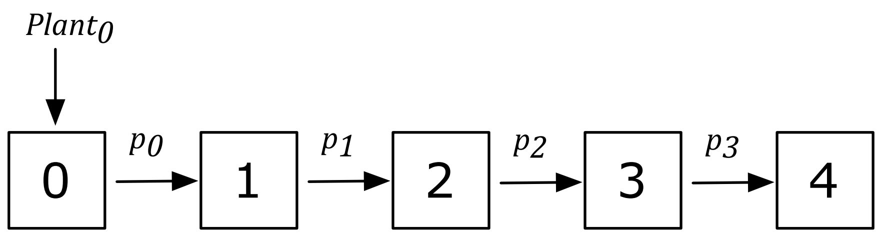
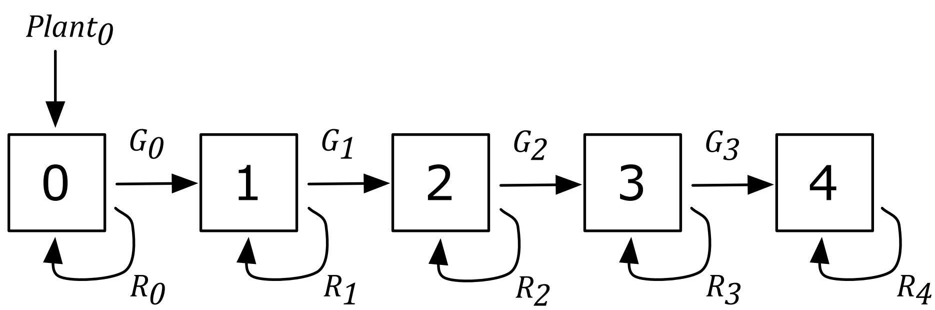


Figure 3. Life cycle diagrams for (a) age-structured urban trees, (b) stage-structured urban trees, and (c) stage-structured wildland trees. Boxes in the age-structured model represent age classes; boxes in stage-structured models represent tree size classes. The age-structured model (a) is simplified with only 5 age classes, assuming that the oldest age class is the maximum lifespan. $Plant_0$ is the planting rate for both urban trees models (a and b), assuming that trees are only planted in the smallest age or size classes. Note that our cohort life table example (Table 3a) does not include a planting rate, only the survival rates. In the age-structured model (a), p_x is the annual survival rate for each age class x. Both stage-structured models (b and c) assume that individuals in a given size class can advance to the next size class, or remain in the same class, but can neither go back to a smaller size class nor skip ahead two size classes. See Table 3 for notation for stage-structured models. The wildland stage-structured model (c) assumes that only trees in the largest size class contribute to recruitment (F_4).

(a)



(b)



(c)

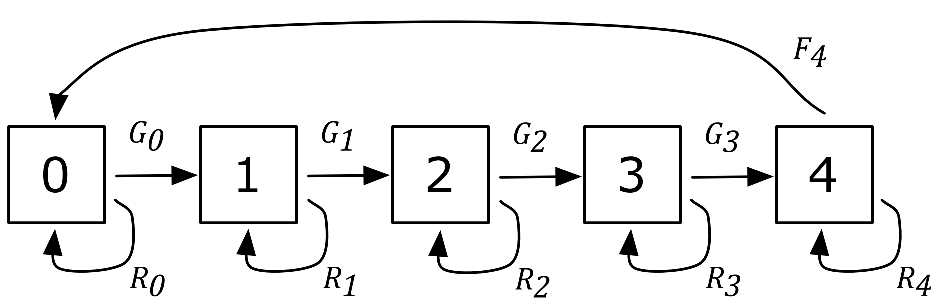
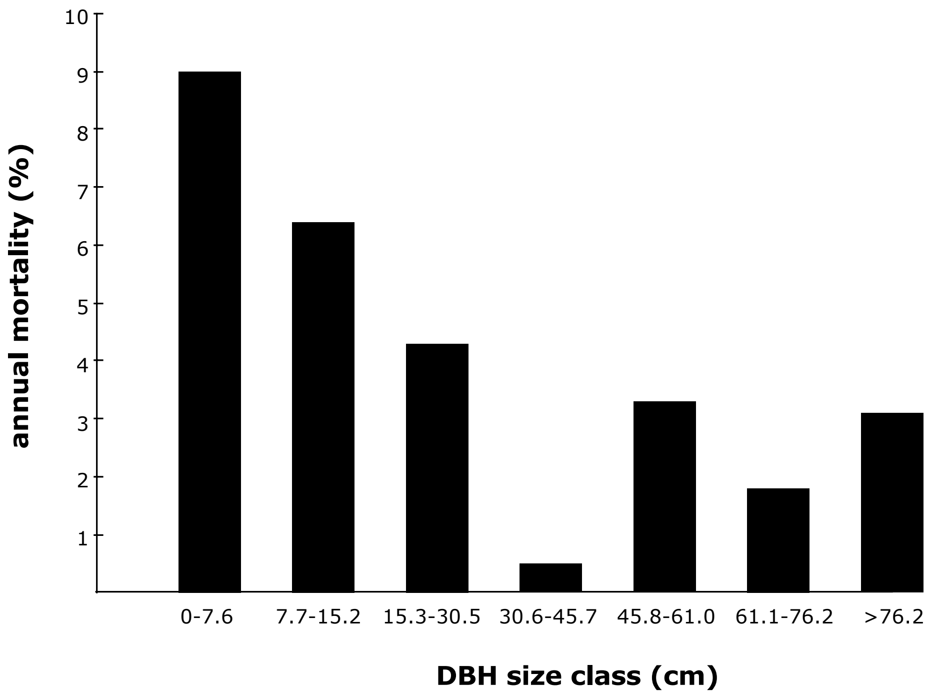


Figure 4. Urban tree mortality rate curve for Baltimore, MD. Adapted from Nowak et al. (2004). Trees were observed over a 2-year study period, and included many species, land uses, and planting locations.



Appendix 1: Supplemental details about Sacramento tree survival calculations.

See Chapter 3 (Appendix 2) for additional details on how SMUD calculated results for the Lindeleaf (2007) study.

For our Sacramento life table (Table 3a), trees were distributed in the Sacramento Shade Tree Program between Jan.-Dec. 2007, with annual mortality observations and field work each summer 2008-2012. The life table presented here only considers trees that were planted, omitting trees that were distributed to residents and never planted. This life table includes trees on all land uses, whereas our other manuscript was limited to single-family residential properties (Chapter 3). Interval censoring was also present in our Sacramento shade tree study. The exact date of death was unknown; rather, the trees died between two known field dates, typically between two subsequent summers. Cohort life tables can accommodate interval censoring as long as the death event occurs within the pre-defined age classes. However, interval censoring becomes problematic for the life table when the interval length was longer than one year (e.g., tree seen alive in 2009, unknown 2010 and 2011, observed removed 2012). Our cohort life table in this chapter (Table 3a) excludes three such cases. A different approach to mortality data, the Turnbull (1976) estimator to Kaplan-Meier survival curves, can accommodate both right and interval censoring. For more precise survivorship estimates, we used the 'interval' package in R (Fay and Shaw 2010), with specific planting dates and field dates included (Chapter 3). Nonetheless, the cohort life table is a simple approach that both researchers and practitioners in urban forestry may find useful to summarize mortality data.

Chapter 2

The balance of planting and mortality in a street tree population, West Oakland, CA

Abstract

Street trees have aesthetic, environmental, human health, and economic benefits in urban ecosystems. Street tree populations are constructed by cycles of planting, growth, death, removal and replacement. The goals of this study were to understand how tree mortality and planting rates affect net population growth, evaluate the shape of the mortality curve, and assess selected risk factors for survival. We monitored a street tree population in West Oakland, CA for five years after an initial inventory (2006). We adapted the classic demographic balancing equation to quantify annual inputs and outputs to the system, tracking pools of live and standing dead trees. There was a 17.2% net increase in live tree counts during the study period (995 in 2006, 1166 in 2011), with population growth observed each year. Of the live trees in 2006, 822 survived to 2011, for an annual mortality rate of 3.7%. However, population growth was constrained by high mortality of young/small trees. Annual mortality was highest for small trees, and lower for mid-size and large trees; this represents a Type III mortality curve. We used multivariate logistic regression to evaluate the relationship between 2011 survival outcomes and inventory data from 2006. In the final model, significant associations were found for size class, foliage condition, planting location, and a multiplicative interaction term for size and foliage condition. Street tree populations are complex cultivated systems whose dynamics can be understood by a combination of longitudinal data and demographic analysis. Urban forest monitoring is important to understand the impact of tree planting programs.

Keywords

demography, monitoring, mortality curve, Oakland, survivorship, urban forest

Introduction

Street trees are essential to the green infrastructure of cities. These trees – located in sidewalk cut-outs, street-side planting strips, and medians – have aesthetic, environmental, public health, and economic benefits (Dwyer et al. 1992; McPherson and Simpson 2002; Nowak and Dwyer 2007). Many great streets and boulevards are characterized by their trees (Lawrence 1988; Jacobs 1995). Street trees improve air quality, reduce stormwater runoff, sequester carbon dioxide, shade buildings to reduce energy use (McPherson and Simpson 2002; McPherson 2003; McPherson et al. 2005), increase property values (Laverne and Winson-Geideman 2003; Donovan and Butry 2010; Donovan and Butry 2011), and promote consumer behavior in business districts (Wolf 2003; Wolf 2004). Urban areas with more street trees have been associated with lower prevalence of childhood asthma (Lovasi et al. 2008). Street trees also contribute to urban design aesthetics and walkable, livable neighborhoods (Appleyard 1981 Southworth 2003; Southworth 2005; Tilt et al. 2007; Merse et al. 2008). The planting and maintenance of street trees are central components of urban forestry programs around the world. To maximize the value of urban tree planting initiatives, these trees must survive to maturity, when canopy cover and associated benefits are greatest.

Street tree populations are constructed by human-driven cycles of planting, growth, death, removal, and replacement. To increase the overall number of street trees in a given city or neighborhood, the number of newly planted trees must exceed losses from death and removal. Tree size and age class distribution are important to street tree population stability, with an adequate proportion of recently planted young, small trees needed to offset early mortality (Richards 1983; McPherson and Rowntree 1989; Maco and McPherson 2002). Richards (1979) suggested that young street tree death – as opposed to older tree mortality – was the primary determinant of the replacement rate needed to maintain the street tree community in Syracuse, NY. Projecting future changes and replacement planting needs (Richards 1979; Bartsch et al. 1985; Brack 2006) in urban tree populations requires information on tree mortality and planting rates (Nowak et al. 2004).

Demographic concepts, such as survivorship and mortality curves, are useful to analyze urban tree mortality rates (Roman and Scatena 2011). Size-based mortality curves, which illustrate how death rates vary by size class, are widely discussed in forest ecology (Buchman 1983; Harcombe and Marks 1983; Buchman and Lentz 1984; Buchman 1985; Harcombe 1987; Monserud and Sterba 1999; Lorimer et al. 2001; Umeki 2002; Coomes and Allen 2007; Metcalf et al. 2009; Lines et al. 2010). Trees in wildland (i.e., non-urban) forests generally follow U-shaped mortality curves with respect to trunk diameter size class (Harcombe 1987; Lines et al. 2010), in which annual mortality is relatively high for small understory trees, low and steady for mature overstory trees, and rises again for very large trees. Street trees may follow a similar U-shaped mortality curve, albeit with different causal mechanisms. The first several years after planting, referred to as the establishment period, may have the highest annual mortality rates (Richards 1979; Miller and Miller 1991). Street tree death rates may stabilize for mid-size trees, then rise again in the larger size classes with senescence-related death (Richards 1979), and removal of large trees that are hazardous to infrastructure or property (Harris et al. 2004; Smiley et al. 2007). For example, size-based mortality rate data from street trees in Syracuse, NY (Nowak 1986) followed a U-shaped mortality curve, as did mortality rates for trees across the urban landscape in Baltimore, MD (Nowak et al. 2004). Another mortality curve shape observed in wildland forests is the Type III curve, in which mortality rates are highest for small trees, and low for mature and very large trees (Harcombe 1987; Lorimer et al. 2001). Identifying the shape of the street tree mortality curve would be useful for urban forest management by improving our understanding of tree death and removal rates, and subsequent replacement needs. Accurate mortality curves would also be useful in cost-benefit analyses of urban forest ecosystem services, which are sensitive to assumed mortality rates (Hildebrandt and Sarkovich 1998; McPherson et al. 1998; McPherson and Simpson 2003; McPherson et al. 2008; Morani et al. 2011).

Previous studies of urban tree mortality have identified numerous causes of tree death and removal, including biophysical and social factors. Urban tree mortality has been associated with species, size, and health condition of the tree, as well as planting location and land use at the site (Nowak et al. 1990; Nowak et al. 2004; Lu et al. 2010; Lawrence et al. 2011). Socioeconomic status of the neighborhood, vandalism, and community involvement have also been connected to mortality (Sklar and Ames 1985; Nowak et al. 1990; Pauleit et al. 2002; Boyce 2010; Lawrence et al. 2011). Other factors contributing to urban tree mortality include compacted and contaminated soils (Grabosky and Bassuk 1995; Craul 1999; Scharenbroch et al. 2005), water stress (Whitlow et al. 1992; Nielsen et al. 2007), construction damage (Hauer et al. 1994), nursery production and transplanting technique (Ferrini et al. 2000), extreme weather events (Hauer et al. 1993; Duryea et al. 1996; Duryea et al. 2007; Staudhammar et al. 2011), and invasive pests and pathogens (Dreistadt et al. 1990; Poland and McCullough 2006; Lacan and McBride 2008). However, previous urban tree mortality studies commonly investigated risk factors for tree death with univariate analysis (Nowak et al. 1990; Nowak et al. 2004; Lu et al. 2010), assessing each factor individually without accounting for confounding or interactions among factors. To understand the causes of tree death in complex urban environments, researchers should assess the strength of individual factors in multivariate models (e.g., Lawrence et al. 2011; Staudhammer et al. 2011), which are widely applied in mortality research in forest ecology (e.g., Das et al. 2007; Lines et al. 2010) and public health (Hosmer and Lemeshow 2000; Jewell 2004).

In this study, we used five years of street tree monitoring data from the neighborhood of West Oakland, CA to investigate mortality rates and risk factors. Our research objectives were to 1) determine how the street tree population size changed over the study period, in relation to annual planting and mortality rates; 2) assess the shape of the street tree mortality curve; and 3) analyze the association between selected risk factors and survival with multivariate logistic regression.

Methods

Study system

This study took place in Oakland, CA, a Mediterranean climate city whose tree cover has increased with human settlements due to current and historic community-driven tree planting initiatives (Cole 1979; Nowak 1993). The research site (Figure 1) is located in the West Oakland neighborhood and encompasses approximately 12 by 12 city blocks (bounded by 35th St., Martin Luther King, Jr. Way, West Grand Ave., and Peralta St.). The USDA Forest Service and Urban Releaf, a local non-profit organization, completed a street tree census in 2006 as a baseline to model hydrologic effects of increased street tree population and canopy cover (USDA Forest Service 2006; Xiao and McPherson 2011).

West Oakland is a predominantly African-American and low-income community (Costa et al. 2002; Gonzales et al. 2011). The neighborhood has a concentration of pollution sources from highways and industry, including the Port of Oakland and trucking businesses (Costa et al. 2002; Fisher et al. 2006; Gonzales et al. 2011), and high rates of childhood asthma and lead poisoning (Costa et al. 2002). In response to these environmental justice concerns, West Oakland is the focus of street tree planting efforts by Urban Releaf and the City of Oakland. The research site has a residential, commercial, industrial, and institutional land uses, often mixed within a city block.

Field data collection

The initial 2006 street tree inventory followed i-Tree Streets (formerly STRATUM) protocols (www.itreetools.com). Core information measured included tree size, health, location type, and adjacent land use (Table 1). To assess the impact of current planting initiatives on the street tree population in West Oakland, we monitored all street trees in the study plot annually from 2007-2011. Field work took place in Jun.-Oct. each year. During the monitoring years, we recorded newly planted trees and status of previously observed trees. Tree status was recorded as removed, standing dead, or alive. Trees marked alive or standing dead were retained in the dataset for monitoring checks the following year. Standing dead status was defined by the absence of any green leaves and live buds. Additional details about field methods, including quality assurance / quality control and logistical concerns, are found in Appendix 1.

For this study, we used a restrictive definition of street trees: only trees in sidewalk cut-outs and planting strips, plus trees in medians, were included for monitoring. Only planting strip locations along the street side of the sidewalk were included. Some additional trees in lawns within the right-of-way or planting strips adjacent to buildings were in the 2006 inventory, but inconsistencies regarding whether those trees were included in 2006 prevented the inclusion of those planting location types in the monitoring study.

Data analysis

Demographic equations, mortality rates, and population growth

Annual tree counts and mortality observations were used to calculate the elements of the street tree demographic balancing equations, and to determine annual mortality rate and population growth. The classic balancing equation (Preston et al. 2001) demonstrates how population size changes over time with the addition of individuals through birth and in-migration, and the subtraction of individuals through death and out-migration (Table 2a, eqn. 1).

For street trees, applying the balancing equation requires modifications in both calculation and conceptualization. While the classic balancing equation (Table 2a, eqn. 1) is traditionally applied to a population of the same species, the street tree balancing equations (Table 2b, eqn. 2 and eqn. 3) include the entire community of trees, with multiple species. Other authors have used the term “street tree population” to describe all street trees in a given area (McPherson & Rowntree 1989; McPherson & Simpson 2002; McPherson 2003). We follow that convention while acknowledging that street tree populations are anthropogenically-constructed systems with multiple species.

The street tree population in West Oakland is an open system: trees enter through planting and leave through removal (Figure 2). In this study system, we observed no natural recruitment of new seedlings. The pool of street trees at any particular census T included both living trees, $N_A(T)$, and standing dead trees, $N_D(T)$ (Figure 2). Consider the pool of live trees observed at year T. At the next monitoring check, $T+1$, those trees are either still alive

($Survived[T, T+1]$), standing dead ($Died[T, T+1]$), or removed / missing ($Removed_A[T, T+1]$). Newly planted trees are added in through $Plant_A[T, T+1]$; this specifically refers to newly planted trees that are observed alive at time $T+1$. These changes in the pool of live trees are encapsulated in the modified balancing equation (Table 2, eqn. 2).

The annual mortality rate, AMR, from T to $T+1$ is:

$$AMR[T, T+1] = \frac{Died[T, T+1] + Removal_A[T, T+1]}{N_A(T)} = 1 - \frac{Survived[T, T+1]}{N_A(T)}$$

(eqn. 4, after Sheil et al. 1995 eqn. 6)

Previous urban forest studies have similarly combined dead and removed (i.e., “missing”) trees in the definition of mortality rate (Nowak et al. 2004; Lu et al. 2010; Roman & Scatena 2011). To calculate annual mortality rate, only the observed status at each census was relevant. It is unknown whether trees represented by $Removed_A[T, T+1]$ were removed while still alive, or removed after dying. Additionally, the annual mortality rate, as defined here, includes only live trees from time T in the denominator; Nowak et al. (2004) calculated mortality rates of re-censused urban forest plots in Baltimore, MD in the same manner.

Next, consider the pool of standing dead trees observed at year T. Some of these trees were removed by the next census ($Removed_D[T, T+1]$), and the rest remained in the landscape as standing dead trees ($StillDead[T, T+1]$). Newly planted trees observed at time $T+1$ were added to the dead tree pool if they were standing dead during summer field work ($Plant_D[T, T+1]$). Presumably all new trees were alive when they were put in the ground, but by the summer monitoring check, a few had already died.

The change in live street tree counts is referred to as the population growth rate. As with the demographic balancing equation, methods are rooted in population biology of natural systems. The intrinsic population growth rate λ and the annual population growth rate λ_T (Table 3, eqn. 5) are central to demographic models (Silvertown et al. 1993; Morris & Doak 2002). In count-based population viability analysis, the arithmetic mean of the log population growth rate, μ, is used to assess population trends and predict extinction risk (Morris & Doak 2002). Population trajectories will tend to grow when $\mu > 0$ and $\lambda > 1$, while trajectories will tend to decline when $\mu < 0$ and $\lambda < 1$. The variance of the log population growth rate is given by σ^2, a measure of the year-to-year variability in population counts (Morris & Doak 2002, eqn. 3.9). We calculated the estimates of μ (Table 3, eqn. 6) and σ^2 using annual counts of live street trees. In this study, the population count-based approach was strictly used to describe observed trends in the street tree population, and not to project future changes in population size.

We also calculated three other informative metrics from the annual tree censuses (Table 4, eqns. 7-9). These metrics – proportion standing dead, proportion standing dead removed, and proportion of newly planted live trees among total live trees – complement the classically-defined mortality rate and population growth rate, and they help to summarize observations of tree death, removal, and planting in the population.

Mortality and survivorship curves

To assess the shape of the street tree mortality curve, we used size-based mortality rates for the five-year (2006-2011) observation period. Diameter at breast height (DBH) size class bins were organized similar to Nowak et al. (2004) (Table 1). The value $Survived[2006, 2011]$ represents the number of trees that were alive in 2006 which survived to census 2011. The annual mortality rate based on census data from 2006 and 2011 is:

$$AMR[2006, 2011] = 1 - \left(\frac{Survived[2006, 2011]}{N_A(2006)}\right)^{(1/5)}$$

(eqn. 10, after Sheil et al. 1995 eqn. 6)

Mortality rates were calculated separately for each DBH size class to create the mortality curve. Note that this formula is simply an extension of eqn. 4, which was only applicable to one-year time intervals. Previous forest ecology studies reporting mortality curves have used a wide range of interval periods (e.g., 1-21 years in Lines et al. 2010).

A subset of the initial 2006 inventory was used to create this size-based mortality curve. Palm trees were not relevant to this mortality curve because their DBH size class is not meaningfully related to health or age. *Cupressus sempervirens* was also excluded because of inaccessible DBH due to tree growth form. Trees lacking DBH information in the 2006 database and trees omitted by field crews from the 2006 inventory were also excluded from the mortality curve (Appendix 1). Multi-stem trees were included in the mortality curve, with the geometric mean of recorded stems used for size class categorization (sensu Nowak et al. 2004).

We also calculated age-based survivorship for newly planted trees observed during the monitoring years 2007-2011 to quantify tree survival during the establishment period. All new street trees observed during census T were treated as an even-aged cohort. Although the trees were not planted at precisely the same time, complete planting records were unavailable, and for simplicity we lumped them into cohorts according to the year of first observation.

Association between five-year survival and selected risk factors

To analyze the association between of several potential risk factors and tree survival, we constructed logistic regression models. The outcome of interest was five-year tree survival (2006-2011), and the potential explanatory variables were DBH size class, foliage health condition, wood health condition, planting location site, and land use recorded in 2006 (Table 1). These risk factors were selected because they are commonly recorded items in most street tree inventories, and they have been previously connected to mortality (Nowak et al. 2004; Lu et al. 2010; Lawrence et al. 2011). Species was not included in regression models because of the wide assortment of different species included (Appendix 2), and clustering of certain species in different size classes, making it difficult to include species meaningfully in the models. We used a subset of the original inventory for the regression model; only trees with complete data for all risk factors were considered. In addition to the exclusion reasons listed above for the size-based mortality curve, a few trees that lacked 2006 health condition were also excluded.

Multivariate logistic regression models for mortality or survival enable interpretation across a range of risk levels, and for the incorporation of interactive effects (Jewell 2004). Logistic regression is commonly used to study binary outcomes in epidemiology for human populations, such as death and disease occurrence (Hosmer and Lemeshow 2000; Jewell 2004), and tree mortality in wildland forests (e.g., Das et al. 2007; Lines et al. 2010). We built models using the logit function in Stata 11 (StataCorp 2009). The general form of a multivariate logistic regression model, expressed as the logit function, is:

$$\log\left(\frac{p_{x,y}}{1 - p_{x,y}}\right) = a + bx + cx$$

(eqn. 10, after Jewell 2004 eqn. 14.2)

where X and Y represent independent risk factors, and $p_{x,y}$ is the probability of survival given that those risk factors take on particular values. For ordinal variables (DBH size class and health condition rating), the coefficient b is interpreted as the log odds ratio (OR) of a unit increase in x, holding c fixed. The odds ratio is a measure of effect size, describing the strength of association between the explanatory variable and the outcome (see Jewell 2004). For nominal variables (location site and land use), indicator ("dummy") variables were used (Hosmer and Lemeshow 2000; Jewell 2004), with one category selected as baseline and compared against the other categories. In these cases, the coefficient is interpreted as the log odds comparing a given category to the baseline (baselines for the final model are provided in Table 7a). Survival was used as the outcome of interest, as opposed to mortality, for ease of interpreting odds ratio results.

For model building, we used an iterative process to compare nested models with likelihood ratio tests; the final model had the highest likelihood, corrected for degrees of freedom (Hosmer and Lemeshow 2000; Jewell 2004). We also used likelihood ratio tests to evaluate the use of indicator variables for DBH size class and health condition. Indicator variables may be appropriate if mortality risk does not change linearly as size class or health condition increases (Jewell 2004). We considered multiplicative interaction between health condition and size class, because small trees are more susceptible to stress and injury (Richards 1979; Miller and Miller 1991). This specific interaction was included based on field observations and plausible mechanisms for interaction; interactions between other explanatory variables are possible but were not considered.

The fit of the final model was evaluated with two diagnostics: the Hosmer-Lemeshow goodness-of-fit test and the receiving operator characteristic (ROC) curve. The Hosmer-Lemeshow test divides the sampled individuals into categories of predicted risk, using a Pearson X^2 to compare predicted and observed risk (Hosmer and Lemeshow 2000; Jewell 2004). For this test, a small p-value indicates lack of fit (Hosmer and Lemeshow 2000; Jewell 2004). The area under the ROC curve was used to assess model discrimination, where 0.5 indicates no discrimination, 0.7-0.8 indicates acceptable discrimination, 0.8-0.9 indicates excellent discrimination, and >0.9 indicates outstanding discrimination (Hosmer and Lemeshow 2000).

Results

Demographic equations, mortality rates, and population growth

The total number of live street trees in the plot increased from 995 in 2006 to 1166 in 2011 (Table 5a); this is a net increase of 171 trees, or 17.2%. Live tree counts from 2006 included 31 trees that were assumed to have been omitted from the initial inventory records (Appendix 1). Of the 995 live trees in 2006, 822 survived to 2011, for an annual mortality rate of 3.7% (eqn. 10).

The annual population growth rate was positive each year during the study period, with low variance ($\hat{\mu} = 0.0317$, $\hat{\sigma}^2 = 0.0004$, Table 5a). A total of 401 new live trees were recorded from 2007-2011, with an average of 80 new live trees per year (Table 5a). Based on the modified balancing equations, the annual mortality rate during the study period ranged from 2.3-10.3% (Table 5a, eqn. 4). The average annual proportion standing dead was 1.7%. Of the standing dead trees observed during census T, an average of 56.7% were removed by the next census (Table 5b). Illustrations of live and standing dead street trees are shown in Figure 3.

Mortality curves and young tree survivorship

A subset of 940 live trees from 2006 was used to construct the size-based mortality curves (94% of the total live trees in 2006). Excluded trees were 12 palms, 2 *Cupressus sempervirens*, and 41 trees missing the 2006 DBH measurement. For this subset of trees, the annual mortality based on the 2006-2011 observation interval was 3.8% (eqn. 10). The street trees in this neighborhood generally followed a Type III mortality curve, with 5.6% annual mortality for the smallest size class, 0.8-1.6% for mid-size trees, and 0% for the largest size class (Figure 4). The smallest size class also constituted a majority (61%) of the trees in the mortality curve (Figure 4).

Survivorship data for the newly planted trees observed in monitoring years 2007-2011 (Table 6) shows high mortality in the first few years after planting. Averaging across the cohorts, typically 99% of new trees were observed alive during their first census, 91% survived 1 year after they were first observed, 83% survived 2 years, and 75% survived 3 years.

<u>Association between five-year survival and selected risk factors</u>

The final logistic regression model (n=924, 93% of the total live trees in 2006) included explanatory variables DBH size class, foliage health condition, planting location site, and an interaction term for DBH size class and foliage condition (Table 7a). Diagnostic evaluations indicated that the final model had acceptable discrimination (area under ROC curve = 0.7648) and no evidence of lack of fit (Hosmer-Lemeshow goodness-of-fit p-value = 0.2112).

Larger trees and those with better foliage health ratings had higher survival over five years. Trees in sidewalk cut-outs had higher survival compared to planting strips. The three largest DBH size class categories (Table 1) were combined due to the absence of deaths in the largest size classes; zero cells in contingency tables are commonly collapsed in logistic regression due to challenges in estimating odds ratios (Hosmer and Lemeshow 2000).

The multiplicative interaction term allowed for assessment of survival outcomes at varying levels of size class and foliage condition (Table 7b). Foliage condition was strongly associated with survival for the smallest size class ($\widehat{OR} = 2.298$, $p<0.001$). However, for mid-size and larger size classes, foliage condition was not significantly related to survival. DBH size class was significantly associated with survival across all foliage conditions, but the relationship was stronger (higher $\widehat{OR}$) for trees in the dying and poor health condition ratings.

Discussion

The West Oakland street tree population grew during the study period, with additions from new plantings exceeding losses from removals and deaths. However, the rate of growth was constrained by high morality of young and small trees. Many new young trees died or were removed during the first few years after planting (Table 6). This observation is complemented by the relatively high mortality rate for trees in the smallest size class (Table 2). The size-based mortality curve for West Oakland street trees has a Type III shape (Figure 4; Harcombe 1987), unlike the U-shaped mortality trend seen in Syracuse (Nowak 1986; street trees only) and Baltimore (Nowak et al. 2004; street, yard and park trees). It is possible that different cities and segments of the urban forest have different mortality curve shapes. However, compared to the Baltimore and Syracuse studies, the West Oakland plot also had very few large trees. If only one of the 11 large trees in our Oakland study had died over the five-year study period, the mortality curve would have been U-shaped. Additional long-term data is needed to assess the conditions under which urban trees exhibit U-shaped and Type III mortality curves. Determining the shape of the urban tree mortality curve is important for population projections and monetization of ecosystem services (McPherson et al. 2008; Morani et al. 2011).

Our analysis of changing population size over time was rooted in the classic demographic balancing equation (Preston et al. 2001). The street tree balancing equations (Table 2, Figure 2) provided a conceptual framework to summarize transitions in the population, separating the pools of living and standing dead trees. The live street tree population in West Oakland was in a continual state of flux during the study period. Large inputs of new young trees every year were necessary to out-pace mortality losses. These findings provide quantitative support for Richards' (1979) assertion that young tree death drives urban tree population cycles. Researchers have previously suggested that an adequate proportion of young/small trees is needed for population stability (Richards 1983; McPherson and Rowntree 1989; Maco and McPherson 2002). In this neighborhood, the large proportion of small trees (61% of trees in smallest size class 2006), coupled with very high mortality of young (Table 6) and small (Figure 4) trees, suggests vulnerability to population crashes if planting efforts slow down. New live trees accounted for an average of 7.4% of the total live tree population every year (Table 5a). As Clark et al. (1997) explained, "sustainable urban forests require human intervention"; this is especially true for street tree populations, as they are constructed by human-driven cycles of planting and removal.

A thorough evaluation of site conditions and maintenance problems was beyond the scope of this research, but such data might offer more direct evidence for causal mechanisms of tree death. The persistence of standing dead trees in the landscape (Table 5b) may indicate slow follow-up to remove and replace dead trees. It is possible that financial resources would be better spent planting fewer trees, and investing more heavily in site modifications and tree care during the establishment phase (Richards 1979; Miller and Miller 1991), to prevent high mortality of young trees. The net increase in population counts and anticipated ecosystem services would be enhanced by lowering young tree mortality rates.

To assess the association between selected risk factors and five-year survival outcomes in West Oakland, we used multivariate logistic regression models (Jewell 2004; Hosmer and Lemeshow 2000). During model building, variables that were not significant (land use, wood health condition) were discarded. Significant explanatory variables in the final model were DBH size class, foliage health condition, planting location, and a multiplicative interaction term between size class and foliage condition (Table 7a). Both DBH size class and foliage condition were treated linearly in the final model, without indicator variables. Without the interaction term, a linear trend in log odds risk for foliage condition did not describe the pattern effectively, and indicators should be used. However, with the interaction term, the simpler model without indicator variables for foliage condition was adequate (results not shown).

Trees that were small and had poor foliage condition in 2006 were less likely to survive to 2011. These results are consistent with previous findings for urban trees in Baltimore, MD (Nowak et al. 2004), although health condition was not separated by foliage and wood in that study. The interaction term allowed a closer inspection of the relationship between size class and foliage (Table 7b). Increasing DBH size class was significantly associated with increased survival across all foliage ratings, with the largest odds ratio for trees with foliage categorized as dying. However, the association between health condition and survival was only significant for the smallest size class. In other words, for mid-size and large trees, there was no significant relationship between foliage condition and survival. There are two possible explanations for this observation. First, relative to large trees, small trees are more susceptible to stress and injury (Nowak et al. 2004), including inadequate maintenance, accidents, and vandalism. Richards

(1979) suggested that establishment-related losses are unique to young and small trees, before they have grown sufficiently to withstand minor injuries. Second, while large tree removal requires trained personnel and equipment (Harris et al. 2004; Smiley et al. 2007), small tree removal is relatively easy. Small trees could have been removed by neighbors due to concerns for tree health or dissatisfaction with tree appearance. Note that in our study and in other urban forest research (Nowak et al. 2004; Lu et al. 2010; Roman & Scatena 2011), mortality is a combination of trees observed standing dead and those observed missing or removed.

In terms of planting location, trees located in sidewalk cut-outs were more likely to survive than those located in planting strips, with no significant difference for median trees (Table 7a). For newly planted street trees in New York City, trees in lawns had higher survival than trees in sidewalks, but soil pit area for sidewalk trees did not have a significant effect on mortality (Lu et al. 2010). The explanation for higher survival of sidewalk cut-out trees in West Oakland is unclear. In this neighborhood, both cut-outs and planting strips provide little space for growing trees (common width 0.6-0.9 m). It is possible that the effect of planting location was confounded by risk factors that were not included in the model.

The small study plot used in this case research may limit the ability to make generalizations to other street tree populations. Different mortality patterns may be observed in neighborhoods with different socioeconomic classes (Nowak et al. 1990), planting programs, maintenance regimes, species composition, and baseline proportions of small trees. However, the annual mortality for West Oakland trees (3.7%) is within the range of typical annual street tree mortality (3.5-5.1%) from a meta-analysis of other studies (Roman & Scatena 2011), which indicates that overall mortality rates were not unusual. Other limitations to our study include potential bias from trees with incomplete information for inclusion in the size-based mortality curves and logistic regression models, and from our method of incorporating trees that were assumed omitted in the 2006 inventory (Appendix 1). Additionally, annual censuses may have missed "ghost mortalities" (sensu Sheil 1995; van Mantgem & Stephenson 2005) – trees that were planted and removed between observations. Lastly, the effect size for responses with large confidence intervals should be treated with caution (Table 7). For some strata in our analysis, small sample sizes within strata of categorical explanatory variables may have contributed to uncertainty reported in the odds ratio (Greenland et al. 2000).

This case study provides a conceptual and methodological framework for future urban tree mortality research. The street tree balancing equations, metrics of population transitions, mortality curves, and multivariate models can be replicated in other cities and neighborhoods, and adapted to other segments of the urban forest. To the best of our knowledge, there is only one previously published study with multi-year street tree monitoring data across all size classes (Boyce 2010), conducted by a neighborhood association. Collaboration and data sharing between urban forest researchers and local practitioners should be enhanced to improve our collective understanding of urban tree population dynamics. Long-term tree monitoring (Baker 1993; McPherson 1993; Pauleit et al. 2002; Brack 2006; Cumming et al. 2008) and longitudinal data are needed to assess the impact of urban forest planting programs in the context of on-going mortality. To gather comprehensive monitoring and mortality data on urban trees, it is essential that urban foresters and urban ecologists coordinate our efforts, partnering with local practitioners and learning from the experiences of forest ecologists working in long-term monitoring programs (Condit 1995; Sheil 1995; Smith 2002; McRoberts et al. 2005; Lindenmayer and Likens 2009; Lindenmayer and Likens 2010).

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Table 1. Street tree inventory data used in the monitoring study. Category definitions for health condition, land use, and location site generally followed i-Tree Streets (formerly STRATUM) (www.itreetools.com).

Variable	Description
Diameter at breast height (DBH)	Stem diameter (cm) at 1.37m from ground; for multi-stem trees, the quadratic mean of observed stems was used
DBH size class	0.1-7.6, 7.7-15.2, 15.3-30.5, 30.6-45.7, 45.8-61.0, >61.0 cm [1]
Health condition rating	Numeric code for the health of the tree, with separate ratings for wood (structural health) and leaves (functional health): dead or dying (extreme problems), poor (major problems), fair (minor problems), good (no apparent problems)
Land use	Land use of buildings adjacent to the tree: single-family residential, multi-family residential, industrial / large commercial, park/vacant/other, small commercial
Location site	Type of planting site where the tree is located: planting strip, sidewalk cut-out, or median[2]

[1] DBH size classes generally followed Nowak et al. (2004); however, the largest size classes defined in that study were combined here due to small sample sizes. In logistic regression models, the largest size classes were further collapsed, with >30.5 cm as the combined largest size class.

[2] Other location site categories were included in i-Tree Streets but excluded from this study (e.g., lawns/yards).

Table 2. (a) Classic demographic balancing equation and associated terms. (b) Demographic balancing equation adapted for a street tree population; eqn. 2 balances the live trees and eqn. 3 balances the standing dead trees. T is time (years) for both (a) and (b).

(a)

$$N(T+1) = N(T) + B[T,T+1] - D[T,T+1] + I[T,T+1] - O[T,T+1]$$

(eqn. 1, after Preston et al. 2001, eqn. 1.1)

Term	Definition
$N(T)$	Number of individuals alive at time T
$B[T,T+1]$	Number of births between T and $T+1$
$D[T,T+1]$	Number of deaths between T and $T+1$
$I[T,T+1]$	Number of in-migrations between T and $T+1$
$O[T,T+1]$	Number of out-migrations between T and $T+1$

(b)

$$N_A(T+1) = N_A(T) + Plant_A[T,T+1] - Died[T,T+1] - Removed_A[T,T+1]$$

(eqn. 2)

$$N_D(T+1) = N_D(T) + Plant_D[T,T+1] + Died[T,T+1] - Removed_D[T,T+1]$$

(eqn. 3)

Term	Definition
$N_A(T)$	Number of trees alive at time T
$N_D(T)$	Number of trees standing dead at time T
$Plant_A[T,T+1]$	Number of new planted trees between T and $T+1$ that are observed alive at $T+1$
$Plant_D[T,T+1]$	Number of new planted trees between T and $T+1$ that are observed standing dead at $T+1$
$Died[T,T+1]$	Number of trees alive at time T that are observed standing dead at $T+1$
$Removed_A[T,T+1]$	Number of trees alive at time T that are observed removed / missing at $T+1$
$Removed_D[T,T+1]$	Number of trees standing dead at time T that are observed removed / missing at $T+1$
$Survived[T,T+1]$*	Number of trees alive at time T that are observed alive at $T+1$
$StillDead[T,T+1]$*	Number of trees standing dead at time T that are observed standing dead at $T+1$

*Although $Survived[T,T+1]$ and $StillDead[T,T+1]$ are not used in eqn. 2 and eqn. 3, they help to illustrate the balancing equations in Fig. 1.

Table 3. Formulae and terms for the population growth rate. The relationships here are used in density-independent count-based models of population viability (Morris & Doak 2002). In the context of this urban forestry study, λ_T is interpreted as the annual street tree population growth rate. To calculate $\hat{\mu}$ and $\hat{\sigma}^2$, the number of live trees at time i, N_{Ai}, was used for all places where simply N_i is used here. The total number of census counts is $q + 1$.

Term	Definition
λ_T	Annual population growth rate $$N(T+1) = \lambda_T N(T)$$ (eqn. 5, after Morris & Doak eqn. 2.1)
$\hat{\mu}$	Estimated value of μ, the arithmetic mean of the log population growth rate $$\hat{\mu} = \frac{1}{q}\sum_{i=0}^{q-1} \ln(N_{i+1}/N_i)$$ (eqn. 6, after Morris & Doak 2002, eqn. 3.9)

Table 4. Supplemental metrics of population change for street trees. These metrics summarize observations about tree deaths, removals, and plantings.

Term	Definition
Proportion standing dead $[T]$	$$\frac{N_D(T)}{N_A(T) + N_D(T)}$$ (eqn. 7)
Proportion standing dead removed $[T, T+1]$	$$\frac{Removed_D[T, T+1]}{N_D(T)}$$ (eqn. 8)
Proportion of newly planted live trees among the total number of live trees $[T]$	$$\frac{Plant_A[T, T+1]}{N_A(T)}$$ (eqn. 9)

Table 5. Annual street tree counts in West Oakland: a) live tree counts, annual mortality rates, and population growth rates, and b) standing dead tree counts, proportion standing dead during each census, and proportion of standing dead trees removed each interval. The demographic balancing equation terms are defined in Table 2, and $\hat{\mu}$ is defined in Table 3 (eqn. 5). The annual mortality rate (eqn. 4) is defined in the text. The proportion standing dead (eqn. 7) and proportion standing dead removed (eqn. 8) are defined in the Table 4.

(a)

year, T	live trees, $N_A(T)$	$Died$ $[T,T+1]$	$Removed_A[T,T+1]$	$Plant_A[T,T+1]$	$\frac{Plant_A[T,T+1]}{N_A(T)}$	annual mortality rate, $AMR[T,T+1]$	$\ln\left(\frac{N_A(T+1)}{N_A(T)}\right)$
2006	995	18	84	139	0.1347	0.1025	0.0365
2007	1032	17	28	48	0.0464	0.0436	0.0029
2008	1035	8	21	54	0.0509	0.0280	0.0239
2009	1060	7	17	78	0.0700	0.0226	0.0497
2010	1114	4	26	82	0.0703	0.0269	0.0456
2011	1166	n/a	n/a	n/a	n/a	n/a	n/a

$\hat{\mu} = 0.0317$
$\hat{\sigma}^2 = 0.0004$

(b)

year, T	standing dead trees, $N_D(T)$	proportion standing dead $[T]$	$Died[T,T+1]$	$Removed_D[T,T+1]$	$Plant_D[T,T+1]$	proportion standing dead removed $[T,T+1]$
2006	5	0.0050	18	3	5	0.6000
2007	25	0.0237	17	15	0	0.6000
2008	27	0.0254	8	15	0	0.5556
2009	20	0.0185	7	9	1	0.4500
2010	19	0.0168	4	12	0	0.6316
2011	11	0.0093	n/a	n/a	n/a	n/a

Table 6. Survival fate of new planted street trees, observed during annual monitoring 2007-2011. All new street trees observed during census T were treated as an even-aged cohort. Although the trees were not planted at precisely the same time, for simplicity we lumped them into cohorts according to the year of first observation. Under each year T are the numbers of new trees from that cohort observed alive during subsequent censuses. In parentheses is the proportion surviving out of the total number planted in that cohort.

	Year, T					
	2007	2008	2009	2010	2011	average
total # new trees in year T	144	48	54	79	82	
# live new trees in year T	139 (0.9653)	48 (1.0000)	54 (1.0000)	78 (0.9873)	82 (1.0000)	0.9905
1 yr. later	128 (0.8889)	41 (0.8542)	54 (1.0000)	71 (0.8987)		0.9104
2 yrs. Later	116 (0.8056)	37 (0.7708)	50 (0.9259)			0.8341
3 yrs. Later	109 (0.7569)	36 (0.7500)				0.7535
4 yrs. Later	105 (0.7292)					

Table 7. Final logistic regression model for 2006-2011 tree survival (n=924), with estimated odds ratios (OR) and 95% confidence intervals (CI) for each parameter: a) overall model results, and b) varying odds ratio estimates across levels of DBH size class and leaves health due to interaction term. Parameters from the 2006 inventory are defined in Table 1.

(a)

parameter	OR estimate	95% CI	p-value	baseline
DBH size class	12.093	3.646, 40.108	<0.001	smallest size class, 0.01-7.6 cm
foliage condition	2.298	1.870, 2.822	<0.001	lowest health condition (dying)
location site				planting strip
cut-out	1.614	1.093, 1.472	0.016	
median strip	0.509	0.176, 1.472	0.212	
interaction: DBH size class * foliage condition	0.522	0.334, 0.816	0.004	

(b)

parameter	OR estimate	95% CI	p-value
OR for a unit increase in DBH size class when leaves health level is:			
Dying	12.093	3.646, 40.108	<0.001
Poor	6.316	2.895, 13.777	<0.001
Fair	3.298	2.163, 5.030	<0.001
Good	1.723	1.175, 2.526	0.005
OR for a unit increase in foliage condition when DBH size class (cm) is:			
[0.01,7.6]	2.298	1.870, 2.822	<0.001
[7.7,15.2]	1.200	0.777, 1.853	0.411
[15.3,30.5]	0.627	0.266, 1.476	0.285
[>30.5]	0.327	0.090, 1.195	0.091

Figure 1. West Oakland study site (red), originally inventoried in 2006 by the USDA Forest Service and Urban Releaf. The site is bounded by bounded by 35th St., Martin Luther King, Jr. Way, West Grand Ave., and Peralta St.

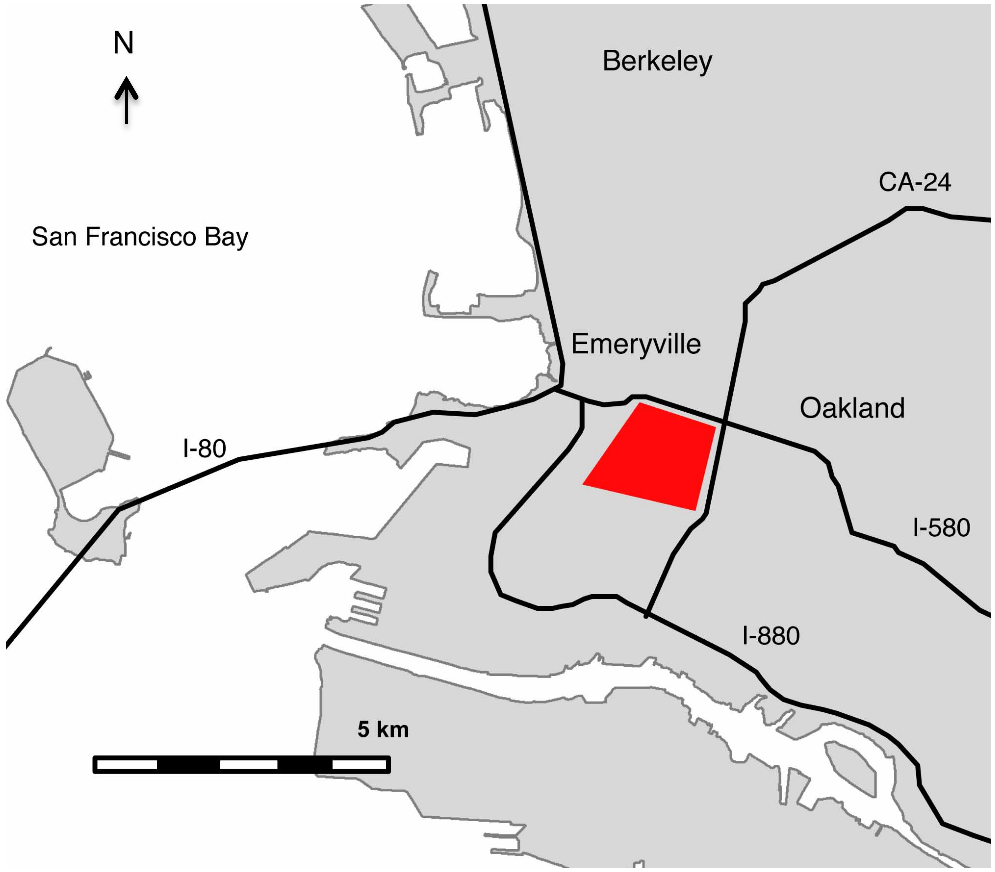


Figure 2. Diagram illustrating street tree balancing equations for live and standing dead street trees (Table 2, eqn. 2 and eqn. 3). Terms are defined in Table 2.

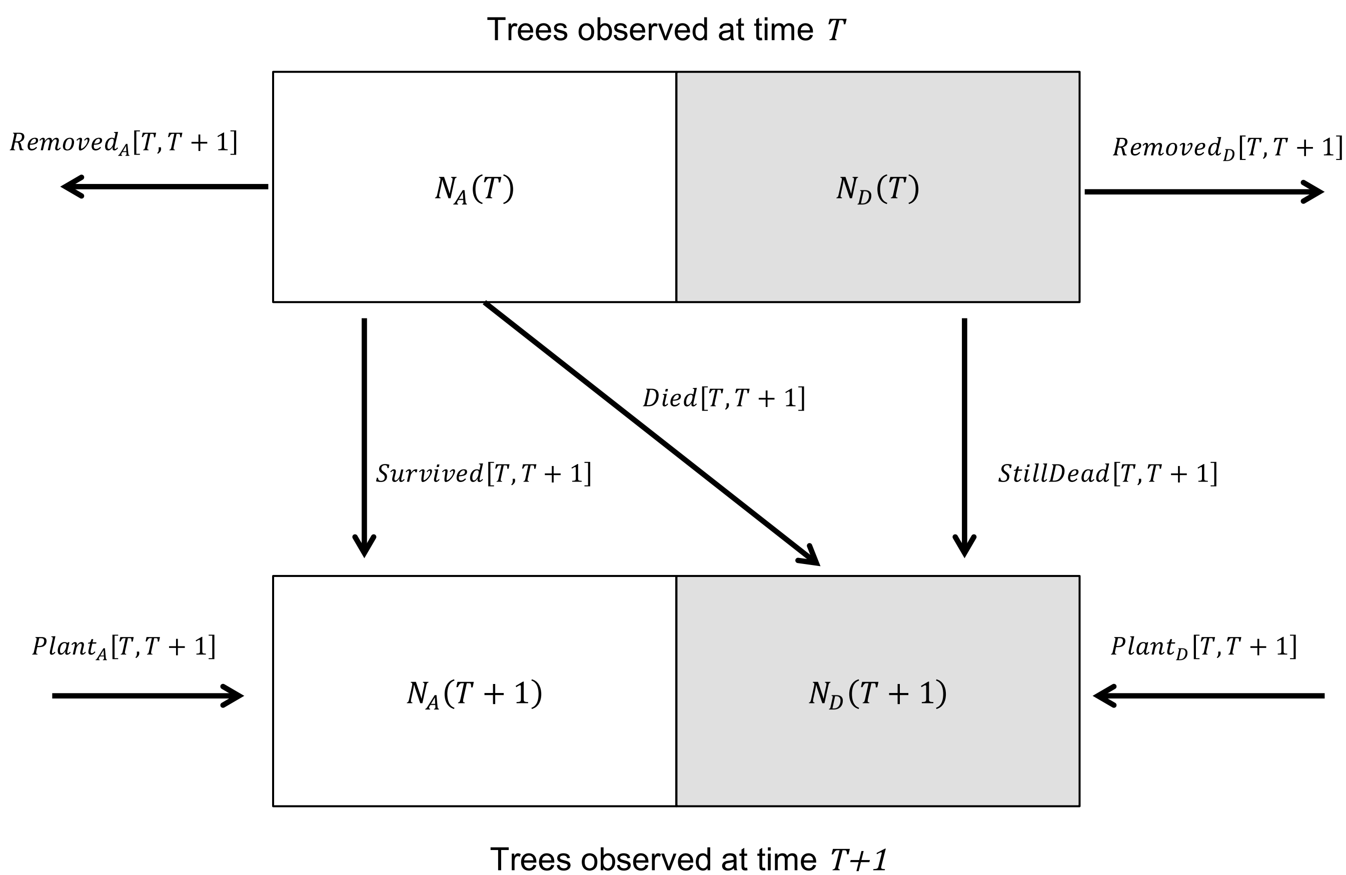


Figure 3. Street trees in West Oakland, CA, with examples of live trees (a, b), a dead tree (c), and a block without trees (d).

(a)

(b)

(c)

(d)

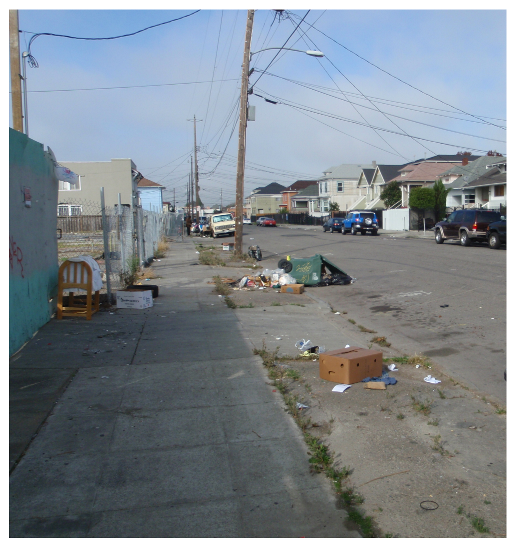
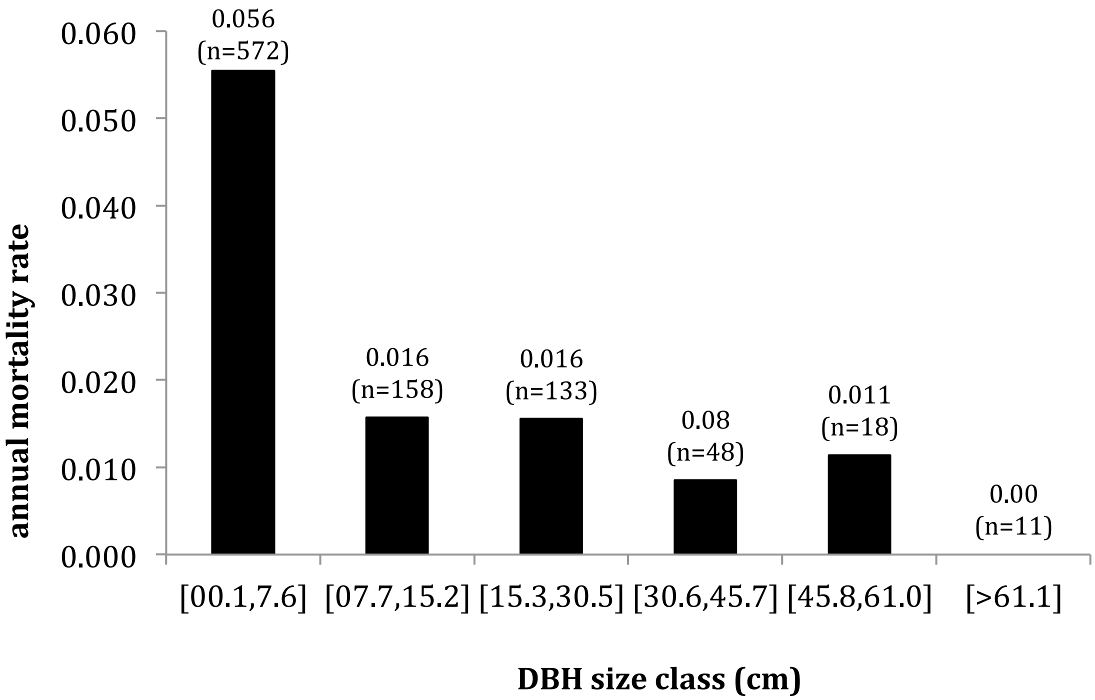


Figure 4. Size-class mortality curves for West Oakland street trees, using annual mortality calculated from the 2006-2011 observation interval (eqn. 10). Total n=940.



Appendix 1: Supplemental field data collection details**.**

Several quality assurance and quality control steps were necessary to adapt the 2006 inventory system to multi-year monitoring. Some trees that were assumed to have been omitted from the 2006 inventory were retroactively added as alive in 2006. In these cases, tree size (>10 cm DBH in 2008) was taken as evidence that they were already in the ground in 2006. The urban forestry initiatives in this neighborhood plant small saplings, therefore it seemed reasonable to assume that mid-size and large trees were omitted in 2006. In the first monitoring year (2007), we also confirmed species, land use, and planting location information from the initial inventory, correcting errors where necessary.

Standing dead status during monitoring years 2007-2011 was defined by an absence of green leaves and live buds. This is a lower threshold of health than the "dead or dying" condition rating in i-Tree (Table 1). Trees from the 2006 inventory recorded as health rating 1 (dead or dying) for both foliage and wood were categorized as standing dead by our definition. However, because health rating is subjective, and different individuals were involved during the inventory vs. monitoring years, this approach to connect our standing dead definition and 2006 health categories was imprecise. There were 2 trees from the 2006 inventory with dying health ratings for foliage and wood that we recorded alive in 2007; however, we also noted that these trees were nearly dead. For simplicity in this analysis, because no backwards transitions were allowed from standing dead to alive, we retroactively re-categorized those 2 trees as alive in 2006.

To facilitate ease of finding trees each year in the study, tree location was recorded with several complementary systems: street addresses, manual notes on a map of GPS coordinates from the 2006 inventory, and order on the block. Tree order on the block was a system of numbering each tree every year in progression from north to south, or east to west, for one side of the street on a given block. The ordering system was used to facilitate database sorting for convenience during field work.

Appendix 2: Species distribution in 2006 inventory**.**

Among the 995 live trees in 2006, the most common species were *Platanus x acerifolia* (12.06%), *Magnolia grandiflora* (10.75%), *Prunus cerasifera* (10.35%), *Pyrus calleryana* (7.49%), *Pyrus kawakamii* (6.53%), and *Fraxinus oxycarpa* (6.43%). All other species represented <5% of the live tree pool in 2006.

Chapter 3

Determinants of establishment survival for residential trees in Sacramento County, CA

Abstract

Urban forests provide valuable ecosystem services that motivate tree planting campaigns, and tree survival is a key element of program success and projected benefits. We studied mortality in a shade tree give-away program in Sacramento, CA, monitoring a cohort of 436 young trees for five years on single-family residential properties. We used Random Forests to identify the most important risk factors at different life history stages, and survival analysis to evaluate post-planting survivorship. Our analysis included socioeconomic, biophysical, and maintenance characteristics. In addition to field observations of tree planting status, survival, and maintenance, we also collected property ownership information (renter vs. owner-occupancy, homeowner change, and foreclosure) through the Multiple Listing Service and neighborhood socioeconomic characteristics from the U.S. Census. We found that 84.9% of trees were planted, with 70.9% survivorship at five years post-planting. Planting rates were higher in neighborhoods with higher educational attainment, and on owner-occupied properties with stable residential ownership. Five-year survival was also higher for properties with stable homeownership, as well as for tree species with low water use demand. When we incorporated maintenance characteristics from the first year of field observations, factors related to tree care were important to survival. Many residents did not adhere to recommended maintenance practices. Our results illustrate the critical role of stewardship and consistent homeownership to young tree mortality on residential properties, and suggest that survival assumptions in urban forest cost-benefit models may be overly optimistic.

Keywords

Random Forests, socio-ecological system, survivorship, tree mortality, urban ecosystem

Introduction

Urban ecosystems are emergent phenomena shaped by biophysical and socioeconomic forces (Grove and Burch 1997; Pickett et al. 1997; Grimm et al. 2000; Alberti et al. 2003; Cook et al. 2011). Urban forests – the trees in cities, towns, and urbanized landscapes (Konijnendijk et al. 2006) – produce valuable ecological services (Dwyer et al. 1992; McPherson et al. 1994; Nowak and Dwyer 2007). Trees in cities improve air and water quality, reduce stormwater runoff, and mitigate the urban heat island (McPherson et al. 1999; Akbari et al. 2001; McPherson et al. 2005; Nowak et al. 2006; Nowak et al. 2008). Urban trees also increase property and rental values (Laverne and Winson-Geideman 2003; Donovan and Butry 2010; Donovan and Butry 2011), and contribute to vibrant, safe, and walkable neighborhoods (Kuo 2003; Southworth 2003; Southworth 2005; Wolf 2005; Tilt et al. 2007). Urban forest structure and species composition are influenced by climate, legacy effects of past land use and pre-settlement habitat, and socioeconomic characteristics (McBride and Jacobs 1986; Loeb 1992; Nowak 1993; Hope et al. 2003; Martin et al. 2004; Ramage et al. 2012). While these studies focused on the forces that affect tree species diversity in cities, the same drivers are relevant to other aspects of the urban forest. We investigated urban tree mortality through the lens of integrated human and biophysical factors.

Tree mortality affects ecological functions and services in both natural (Franklin et al. 1987; Lines et al. 2010; Dietze and Moorcroft 2011) and urban forests. The estimated environmental, socioeconomic, and human health benefits from urban forest cost-benefit models are sensitive to mortality rate assumptions (Hildebrandt and Sarkovich 1998; McPherson et al. 1998; McPherson and Simpson 2001; McPherson and Simpson 2003; McPherson et al. 2008; Morani et al. 2011). These models monetize the values of urban forest ecosystem services and have motivated tree planting initiatives (Silvera Seamans 2013). However, there is a lack of longitudinal data documenting the magnitude of mortality losses in urban tree planting programs. Such data is essential to understand tree population dynamics and associated ecological functions, and to sustainably manage urban forest resources; similar arguments have been made to explain the importance of tree mortality studies in natural forest ecosystems (Lines et al. 2010; Dietze and Moorcroft 2011).

Numerous risk factors for urban tree death have been reported. Biophysical factors include species, size/age class, planting site characteristics (Nowak et al. 1990; Nowak et al. 2004; Lu et al. 2010; Lawrence et al. 2012), soil compaction and altered microbial activity (Craul 1999; Scharenbroch et al. 2005), water stress (Whitlow et al. 1992; Nielsen et al. 2007), extreme weather events (Hauer et al. 1993; Duryea et al. 1996; Duryea et al. 2007; Staudhammer et al. 2011), and pests and diseases (Dreistadt et al. 1990; Poland and McCullough 2006; Laćan and McBride 2008). Human factors include socioeconomic status, land use, vacancy, unemployment, rental status, vandalism, community involvement in tree planting, and construction damage (Sklar and Ames 1985; Nowak et al. 1990; Hauer et al. 1994; Pauleit et al. 2002; Boyce 2010; Lawrence et al. 2012). The relative importance of each of these factors may change as trees age and grow. During the establishment period – the first several years after planting – urban trees may be especially vulnerable to mortality losses (Richards 1979; Miller and Miller 1991). Planting and maintenance techniques are considered particularly critical to establishment success for street and lawn trees (Nowak et al. 1990; Ferrini et al. 2000; Struve 2009; Boyce 2010; Lu et al. 2010). Young tree mortality losses may drive urban forest population cycles and tree replacement needs (Richards 1979; Chapter 2). Recognizing the importance of early tree survival to the success of tree planting campaigns, urban forest researchers and practitioners are both interested in quantifying the rates and causes of mortality (Leibowitz 2012; Chapter 4).

We propose a conceptual model for urban tree mortality as a result of integrated socio-ecological drivers and patterns (Figure 1). This model adapts previous conceptual frameworks for an interdisciplinary approach to the study of urban ecosystems (Grimm et al. 2000; Alberti et al. 2003) and residential landscapes (Cook et al. 2011), as well as models for the sustainable management of urban forests (Clark et al. 1997; Dwyer et al. 2003). In our diagram, socioeconomic and ecological forces produce multi-scalar patterns in human and biophysical characteristics related to the urban trees (Alberti et al. 2003; Cook et al. 2011). These characteristics influence tree maintenance and management practices. The variables in these multi-scalar human and biophysical patterns are potential predictors of urban tree death. Mortality rates and associated population dynamics affect ecological functions and services (Hildebrandt and Sarkovich 1998; McPherson et al. 1998; McPherson and Simpson 2001; McPherson and Simpson 2003; McPherson et al. 2008; Morani et al. 2011). These ecosystem services in turn affect urban forest governance and policy (Silvera Seamans 2013), which are among the human drivers of urban forest patterns.

Our conceptual model (Figure 1) provides a framework for our study of establishment-phase tree mortality in the Shade Tree Program in Sacramento County, CA. This program serves as an example of large-scale tree give-away initiatives (i.e., trees distributed by the program and planted by participants). An investigation of tree mortality in Sacramento has relevance to other large-scale urban tree planting and distribution programs (e.g., “million tree” campaigns in Los Angeles, CA; Denver, CO; New York City, NY; and Philadelphia, PA). In our study, we monitored a cohort of shade trees for five years. We were interested in two distinct outcomes to represent life history stages of trees in the Sacramento program: planting status and post-planting survival. Planting status is of particular interest to urban tree give-away programs, which cannot guarantee that every tree delivered will be planted. Our research objectives were to 1) quantify tree planting rates and survivorship during the establishment period, and 2) identify the primary determinants of establishment survival.

Methods

Our study was designed as an exploratory investigation of young tree death in the Sacramento Shade Tree Program. We included many biophysical and socioeconomic risk factors for mortality based on past literature and discussions with program staff, with the intent to identify the several most important factors in this study system. We used Random Forests (RF; Breiman 2001), an ensemble method based on Classification and Regression Trees (CART; Breiman et al. 1984). RF and CART are nonparametric techniques that have previously been used to study tree mortality in natural and harvested forests (Dobbertin and Biging 1998; Fan et al. 2006; Shifley et al. 2006; Tyler et al. 2008; Solarik et al. 2012), ecological habitat types, land use classification and species distributions (De’ath and Fabricius 2000; Vayssières et al. 2000; Gislason et al. 2006; Prasad et al. 2006; Cutler et al. 2007; Stella et al. 2011), and human disease (Lemon et al. 2003; Bureau et al. 2005; Fonarow et al. 2005; Szabo de Edelenyi et al. 2008). Additionally, these methods can accommodate a mix of variable types, including unordered categorical variables. After our initial exploratory assessment through RF to identify the most important variables, we used chi-squared tests to assess differences in planting rates, and survival analysis to test for differences in five-year survivorship.

Study system

The Sacramento, CA area has a Mediterranean climate with mild, wet winters and hot, dry summers. There are typically 73 days per year with maximum temperature ≥ 32.2 C (NOAA 2013). The Shade Tree Program is intended to produce a specific ecosystem service: reducing building energy use through tree shade during the hot summer months (Hildebrandt and Sarkovich 1998, McPherson et al. 1998). The program is a partnership between a local non-profit organization [Sacramento Tree Foundation (STF)] and a utility company [Sacramento Municipal Utility District (SMUD)]. SMUD and STF have distributed over 500,000 trees since 1990 (C. Cadwallader, pers. comm.); the Shade Tree Program is the largest and longest-running shade tree initiative in the United States. The program operates in the SMUD service area, which includes Sacramento County and small parts of Placer and Yolo Counties.

In this tree give-away program, trees are distributed upon request to residents, property owners, and property managers, who are then responsible for planting and maintenance. SMUD and STF staff refer to individuals who receive trees as "shade tree customers"; we use that term throughout the remainder of this paper. Most trees are distributed to single-family residential (SFR) properties and planted in yards, but some are also distributed to multi-family residential properties, schools, and businesses. We limited our investigation of tree mortality to SFR properties. In these cases, tree stewardship is entirely the responsibility of individual homeowners/residents.

After requesting free trees, customers are visited by STF staff and receive a 15-45 minute consultation about shade benefits, species, planting site selection, and tree stewardship. The agreed-upon planting locations are recorded on a map in a signed Tree Care Agreement. SMUD projects energy savings based on the recorded distance and orientation to the building, along with expected mature tree size and assumed mortality rates. Customers also receive an educational folder. After tree delivery, shade tree customers are responsible for planting and maintenance. Follow-up contact with program staff is not required, but customers are invited to call STF for additional tree care advice, and to participate in free urban forest stewardship classes. Trees delivered by STF are small saplings [containerized trees in 20L buckets (5 gallon trees), approximately 2m tall]. SFR properties typically receive no more than 10 trees.

The economic context during our study period is relevant to tree establishment and mortality in the program. The recent recession affected many homes in the Sacramento region, which had the fifth highest foreclosure rate in the U.S. in 2007 (RealtyTrac 2008) and ninth highest in 2008, with 5.2% of housing units undergoing foreclosure that year (RealtyTrac 2009). Sacramento continued to have high rates of foreclosure for several years. In repossessed bank-owned properties, lawn care typically becomes the responsibility of the bank or lending agency (Keith 2011).

Study sample

We monitored a sample of 436 trees on SFR properties randomly selected from the 13,594 shade trees distributed by STF from Jan.-Dec. 2007. Trees were distributed across the city of Sacramento and surrounding suburbs and small towns (Figure 2). The sample included 30 species (Table 1). In lieu of species-specific analysis, we classified species by water use demand in California's Central Valley (UC Cooperative Extension 2000; Table 1). Water use demand is especially relevant because the Sacramento region has seasonal summer drought. We also classified species by mature tree size because recent SMUD mortality assumptions have included higher mortality rates for small stature trees (M. Sarkovich, pers. comm.).

STF records included the tree delivery date. We used the delivery date as a close approximation of the planting date. Because trees were delivered and planted throughout 2007, we accounted for differences in time elapsed since planting by counting the number of days between delivery date and 2008 field observation date (Table 2). We also used delivery date to categorize trees as planted during the rainy season (Oct-Apr) vs. dry season (May-Sep.). Trees planted during the dry season may need additional care and watering due to seasonal summer drought.

Data collection

Field data

Field work was conducted May-Aug. each year, 2008-2012, with occasional additional field visits through Nov. for unresponsive residents. We contacted study participants each spring by mail and telephone to request access to the properties. Multiple contact attempts were often required to gain access to back yards. When residents were unresponsive, we made at least 3 attempts to visit the property, and left an informational flier about the study. Unresponsive properties were visited again the subsequent year, while residents who opted-out were not visited again.

Because STF distributes trees but does not plant them, we distinguished between residential planting rates and post-planting mortality. Trees classified as never planted were either observed in container during the summer 2008 field work, or were observed missing and determined to have never been planted. Trees that were missing in 2008 may have been planted and subsequently removed; whether a missing tree was planted was determined based on conversations with residents and observations at the properties. Post-planting mortality was a combination of standing dead and removed trees. Other urban forest studies have also defined mortality as a combination of standing dead and removed trees (Nowak et al. 2004; Lu et al. 2010; Roman & Scatena 2011; Chapter 2). Standing dead trees were defined by the complete absence of green leaves and live buds. Trees observed alive during a given field year were visited again the following year and classified as alive, standing dead, or removed.

During the 2008 field season, we observed tree health and planting site characteristics. Tree health rating was recorded separately for foliage and wood following i-Tree Streets methods (itreetools.org). We noted whether the tree was planted in the correct sited location, as depicted on the Tree Care Agreement. We also recorded whether the tree was planted in the front or back yard. Previous studies have found that front yards receive distinct landscaping care as a showcase to the neighborhood (Richards et al. 1984; Larsen and Harlan 2006; Daniels and Kirkpatrick 2006; Larson et al. 2009). Lastly, we recorded planting location ground cover (Table 2).

Adherence to recommended maintenance served as a general indication of residents' propensity to follow STF instructions, along with our observations of whether the trees were planted in the correct location. Maintenance characteristics at the planting site were recorded based on STF recommended practices and educational materials: irrigation, staking, mulching, and trunk wounds (Table 2). The maintenance issues we included have plausible biological mechanisms for affecting tree survival, with anecdotal evidence from STF staff (Table 2; L. Leineke, pers. comm.). Mulching, staking and watering are central elements of arboriculture

(Harris et al. 2004) and are also emphasized in STF tree stewardship materials. With the seasonal drought in the Sacramento region, STF recommends watering all trees for at least three years during the dry season. Because our field visits took place during the summer, when there was little to no rainfall, brown lawns and dry soil were used to indicate lack of watering. Due to the limitations of our field work – with only a few minutes of observation for each tree in 2008 – we did not distinguish between varying levels of irrigation beyond presence/absence of watering (i.e., we only recorded the most extreme cases of lack of watering). With regards to staking, residents should remove the nursery stake when trees are planted; tags from the nursery stake can girdle the trunk. STF provides structural stakes and ties with tree delivery. Mulch is not provided with tree delivery, but it is available for free from SMUD. We also recorded presence of trunk wounds from weed wackers or mowers. STF staff have seen the nursery stake and trunk wound problems on young shade trees, and the issues are mentioned in their printed educational materials and on their website.

In our analysis of risk factors, we used a composite rating for maintenance that gave more weight to irrigation observations. All trees that had no evidence of irrigation were classified as "poor" maintenance. Trees were classified as "good" maintenance when there was evidence of irrigation, nursery stakes were removed, structural stakes were present, mulch was present, and there was no trunk wound (i.e., maintenance generally followed STF instructions). Of the latter four maintenance characteristics, trees classified as "adequate" had one or two problems, along with some evidence of irrigation. When there were three or four problems and some evidence of irrigation, a tree was classified as "poor" maintenance.

Neighborhood socioeconomic data

To incorporate neighborhood socioeconomic characteristics into our analysis, we used U.S. Census data from the American Community Survey (ACS). ACS produces multi-year estimated averages of social, economic, and housing information at the census tract level (geographic areas with 1,500-8,000 people). We used 2007-2011 averages (U.S. Census Bureau 2013) because this overlaps with our study period. Specifically, we used educational attainment (percent of population with bachelor's degree or higher), median income, and median housing value and (Table 2).

For RF, we used raw census tract data, but for survival analysis, we categorized these socioeconomic variables into low, medium, high and very high (Table 2). Category break-points for income were based on category bounds used by the U.S. Census, with some categories collapsed due to small sample sizes in census tracts with very high or very low values.

Homeownership data

STF relies on residents and property owners to plant and care for trees, therefore changes in ownership and occupancy may affect tree health and survival. Houses may be vacant and unmaintained when ownership and occupancy change, new residents may have different levels of tree stewardship, and new residents may also make different landscaping choices. To account for potential homeownership effects, we collected homeowner data to determine foreclosure status, change in residential ownership, and owner vs. renter occupancy. Homeownership data is publically available through the Sacramento County Assessor, and we obtained this data in bulk using the Multiple Listing Service (MLS), a proprietary service for realtors.

For purposes of our study, we defined foreclosed properties as those that were repossessed by banks and investment companies. We did not include properties given foreclosure notices that were not repossessed, nor did we include short sales, because these records were not readily available in the MLS database for all properties during the entire study period. We also recorded change in residential ownership, defined as new non-bank owners. Renter status is not formally recorded with the County Assessor, and records for the homeowner tax exemption are unreliable, therefore we inferred renter status based on the tax address of the owner (D. Covill, pers. comm.). We interpreted a mismatch between the tax address of the owner and the physical address of the house as a rental property. When the tax address was a P.O. Box, this was interpreted as owner-occupied. For all three of these homeownership issues, we recorded information for each year, beginning in 2007 (after the date of STF staff site visit) and ending with the last year the tree was observed (either the year when the tree was observed dead/removed, or 2012 as the last year observed alive).

We combined these three homeowner issues into one metric to identify continuously owner-occupied properties that had the same homeowner throughout the study period; such properties had the same customers responsible for tree stewardship who originally requested the tree. When a property was owner-occupied, with no change in ownership (i.e., no foreclosures or sales to new residents), this was categorized as stable homeownership. Unstable cases had renter-occupancy, foreclosure, and/or new owners.

Data analysis

Random Forests (RF)

RF is an extension of CART (Brieman et al. 1984), a nonparametric binary recursive partitioning method. With classification trees, CART uses a binary outcome; in our case, planted vs. not planted, and survived vs. died. Classification tree analysis produces a tree-like diagram representing a series of hierarchical splits from the “root” of the tree. Classification trees partition data into homogeneous subsets in terms of the explanatory variables. Classification trees are sometimes used to identify predictors of a binary outcome to aid in decision-making, such as clinical rules to guide interventions with human disease (Lemon et al. 2003), and identifying habitats likely to harbor endangered species (Bourg et al. 2005). RF uses random subsets to fit many classification trees to a data set, with a random subset of the available predictors used at each node (Breiman 2001). RF is referred to as an ensemble technique, with results for each predictor averaged across all the trees. Ecological studies are increasingly using RF (Prasad et al. 2006; Cutler et al. 2007).

We used RF because it is well-suited to exploratory studies and can accommodate a variety of variable types. RF also reduces the overfitting problem of CART (Brieman 2001; Strobl et al. 2008), and is a robust technique for data sets with more predictor variables than observations (“large p small n” data sets). We used conditional inference RF with the ‘party’ package in R (Hothorn et al. 2006; Strobl et al. 2009) because this method is unbiased towards different variable types, and because our explanatory variables were likely highly correlated. This package uses random sub-sampling without replacement. We used 1000 trees and five predictors at each node.

We used permutation importance to identify the most important explanatory variables. With the permutation technique, the values of a predictor variable are randomly permuted to break the association with the response. Variable importance is a measure of prediction accuracy for observations before and after permuting, averaged over all classification trees (Strobl et al. 2008; Strobl et al. 2009). Following the recommendation of Strobl et al. (2009), we selected variables for further consideration that had a higher importance value than the absolute value of the most negative score, and we ranked variable importance for each test without inferring additional information from the raw importance score outputs. Because RF results are averaged over all trees, this technique does not produce the easily interpretable tree classification diagrams from CART. To aid in interpretation of RF results, we therefore reported planting and survival outcomes for the variables identified as most important.

For RF analysis (Table 3) on planting status (model P), we considered socioeconomic characteristics (homeowner stability and neighborhood income, housing value, and educational attainment) and program records that may relate to tree stewardship (number of trees delivered). STF staff speculated that customers receiving more trees may provide less care per tree (J. Caditz, pers. comm).

With regard to RF models for post-planting mortality (Table 3), we considered three different time frames: planting through fifth year of observation (2007-2012; model P-5), first year after planting (2007-2008; model P-1), and second-through-fifth year deaths (2008-2012; model 2-5). We separated the first and second-through-fifth year mortality because more trees were lost in the first year post-planting, which is a potential time for intervention from STF staff (C. Cadwallader, pers. comm.), and because we collected tree maintenance and health characteristics in 2008 which could be used in model 2-5. For all of these mortality models, we included tree biophysical characteristics (species water use demand, mature tree size, days since planting, season planted), socioeconomic characteristics (homeowner stability and neighborhood income, housing value, and educational attainment), and number of trees distributed. For model 2-5, we also included tree health metrics and stewardship indicators observed during 2008 field work (ground cover, foliage and wood health condition, yard side, correct location, maintenance rating). In this model, we only included trees that survived through the first year (survived to summer 2008), omitting trees for which we could not closely observe tree health and maintenance (e.g., we saw the tree alive over the back yard fence, but could not gain access to the yard). Trees with unknown status in 2012 were omitted from both models P-5 and 2-5.

Chi-squared tests for planting status

We used chi-squared tests on the variables identified through RF analysis as most important for planting status. For binary explanatory variables, we used Pearson's chi-squared, and for ordered categorical variables, we used the Cochran-Armitage chi-squared test for trend. The latter technique tests for a linear trend in risk as the explanatory variable increases (Jewell 2004). We conducted these tests in STATA (StataCorp 2009) using $\alpha = 0.05$.

Survival analysis for post-planting mortality

We used the variables identified through RF analysis (model P-5 only) to test for differences in survivorship over the five-year study period. Because we had annual field observations, we could not determine the exact date of death for the trees in our sample; most mortality observations were interval censored between two annual field dates. Furthermore, we had right censored data (e.g., unknown status in the fourth and fifth years) and a few cases of interval censored data over longer time spans (e.g., tree observed alive in the third year, unknown status fourth year, confirmed tree death by fifth year, with residents uncertain when tree died). We used Turnbull's (1976) procedure for the Kaplan-Meier or product-limit estimator to calculate survivorship curves (Gomez et al. 2009) and the weighted logrank test from the 'interval' package in R (Fay & Shaw 2010), which was recently used in a study of seedling mortality (Cleavitt et al. 2011). We constructed survivorship curves by day to account for variation in both planting dates and annual field work dates. We treated the tree delivery date as time zero, with the assumption that this is a close approximation of the planting date. We used $\alpha = 0.05$.

We used the results from survival analysis to report overall survivorship and annual survival over the five-year establishment period (for all trees, and for categories of the most important variables from model P-5). Annual survival was estimated using $p = (l_x)^{1/x}$, where p is annual survival rate and l_x is survivorship to year x (after Sheil et al. 1995), assuming constant annual survival. Annual mortality is 1 - p. Although survival may not be constant across the entire five-year period, we used the annual survival estimate to compare our field observations to establishment-phase mortality assumptions in other studies. For this calculation, we used the five-year survivorship based on 'interval' output at day 1825.

Results

Overall planting and survivorship

We obtained at least partial field data for all trees during the five-year study period (Figure 3). Of the 436 shade trees delivered to SFR properties, 15.1% were not planted; 23% of these were observed in container, and 77% were missing and never planted. Of the 370 trees planted, 12.2% died by the first summer of field observation (2008). The 2008 post-planting tree deaths included 27% that were observed standing dead and 73% removed by the time of our field visit. We had some trees with unknown status in subsequent years due to unresponsive residents and opt-outs. Considering the trees that survived until summer 2008, 249 survived until the fifth year (2012), with 13 right censored trees of unknown status. Illustrations of tree status are provided in Figure 4.

The survivorship curve for all trees shows a steady decline during the establishment period (Figure 5). Unlike the monitoring outcomes diagram (Figure 3), this curve takes into account the different planting dates in 2007 and the different field dates each year 2008-2012. At five years after planting, overall survivorship was 70.9%, with estimated 6.6% annual mortality.

Determinants of tree planting and post-planting mortality

For RF analysis concerning planting status, neighborhood educational attainment and homeowner stability were the most important variables (Tables 3, 4). More trees were planted in areas with more educated populations (p=0.0140) and properties with stable owner-occupied residents (p=0.0007, Table 4).

For post-planting mortality, homeowner stability was the most important variable in each time period considered (Table 4). Over the entire five-year study (model P-5), species water use demand, neighborhood income, season planted, mature tree size, and days since planting were also important. Higher survival rates were observed for stable properties, species with low water use demand, trees planted during the rainy season, species with smaller mature size, and trees with fewer days since delivery (Table 6). However, only homeowner stability had a statistically significant difference with the weighted logrank test (Table 6, Figure 6). Neighborhood income did not show a consistent trend.

Focusing on the first year after planting (model P-1), homeowner stability was again the most important variable, followed by days since delivery, neighborhood income, neighborhood educational attainment, and mature tree size (Table 4, Appendix 1). Considering survival of trees that were still alive at the first summer field observation (model 2-5), variables related to tree maintenance were important: homeowner stability, yard side, number of trees delivered, and maintenance rating (Table 4, Appendix 1).

Discussion

Our observed tree losses in Sacramento were higher than urban tree mortality projections used in previous studies for this area (Hildebrandt & Sarkovich 1998; McPherson et al. 1998) and for other cities (McPherson 1994; McPherson & Simpson 2001; Morani et al. 2011; McPherson et al. 2008). While we found 6.6% annual mortality for young trees during the establishment phase, McPherson & Simpson (2001) assumed 3% mortality during the first five years after planting for urban forests across CA, and McPherson et al. (2008) assumed 1-5% mortality in the first five years for Los Angeles, CA. Our observed mortality rates are closer to the high mortality scenario considered by Morani et al. (2011) in New York City, NY, which assumed 4-8% mortality for the smallest size class (0-7cm, roughly similar to our 0-5 years post-planting age class). Additionally, we distinguished between failure to plant rates and post-planting mortality. This is a major distinction for tree give-away programs, which are a component of large-scale urban forestry initiatives in several cities (e.g., Los Angeles, CA; Philadelphia, PA), yet planting rates are overlooked in the literature. We found that 15.1% of trees were not planted in Sacramento.

While it is possible that our failure to plant and post-planting mortality rates were relatively high due to the economic recession and homeowner changes during our study period, our findings are actually consistent with previously collected SMUD data. SMUD calculates program success in terms of "survivability", defined as the proportion of trees survived out of those delivered. Lindeleaf (2007) reported 54% survivability five years after tree distribution (1996-2001). Using SMUD terminology, our five-year survivability was 59% (Appendix 2). SMUD's estimates of shade tree energy benefits previously assumed 58-70% survivability at 30 years after planting (Hildebrandt and Sarkovich 1998). More recently, SMUD assumed 50-57% survivability at 30 years and 69-72% at five years (M. Sarkovich, pers.comm.). Both our field study and the SMUD report indicate that survival projections for tree give-away programs – and perhaps for tree planting programs more generally – are overly optimistic. This finding has implications for urban forest management. Tree planting initiatives use monetized forecasts of ecosystem services to engage the public and justify their programs (Silvera Seamans 2013). Although our specific findings are most relevant to residential tree give-away programs, the broader issue of accounting for tree losses in the million tree campaigns has already caught the public's attention (Foderaro 2011; Miller 2011; Marritz 2012). Realistic expectations of young tree mortality are also important to plan for population cycles of tree planting, death, removal and replacement in urban forest ecosystems (Richards 1979; Bartsch et al. 1985; Nowak et al. 2004; Brack 2006; Nowak 2012; Chapter 2).

With respect to the causes of tree losses in the Shade Tree Program, we found strong evidence that stewardship is the most critical determinant of young tree planting and survival. Our homeowner stability metric ranked as the most important variable in all three RF models for post-planting tree survival, and the second-most important variable for planting status. Trees were more likely to be planted (Table 5) and more likely to survive (Table 6, Appendix 1) on owner-occupied properties that had the same homeowner during the study period. We interpret the homeowner stability metric as an indicator of consistent tree stewardship by the individual(s) who requested the shade trees. Properties with renters, foreclosure (Whitaker 2011), and/or a change in residential ownership likely experienced inconsistent landscape maintenance, with some properties experiencing long gaps in tree care. Other variables related to tree care were also important in our RF analysis for model 2-5, which included observed site and maintenance characteristics in the first year after planting. Higher survival was observed for trees in front yards, those with higher maintenance ratings, and those on properties that had fewer trees delivered (Table 8). Front yards may have higher survival because they serve as a showcase to the neighborhood (Richards et al. 1984; Larsen and Harlan 2006; Daniels and Kirkpatrick 2006; Larson et al. 2009), and potentially receive greater care than backyards. Anecdotally, we also observed some vacant and foreclosed properties with maintained front yards, but unmaintained and overgrown back yards. The higher survival for properties with only a few trees delivered fits with STF speculation that shade tree customers receiving many trees may invest less time per tree. Our maintenance rating, while admittedly imprecise due to a few minutes of field observation, nonetheless ranked as an important variable. Less than a quarter (22.7%) of the trees we observed alive in the first year after planting had maintenance characterized as “good”, and only 74.6% were in the correct sited location, which indicates that most customers did not adhere to recommended tree care practices. Our finding that maintenance effects dominate during the establishment phase concurs with previous research on street trees in New York City, NY (Boyce 2010). In order to increase survival rates in Sacramento, more extensive staff site visits and increased communication with shade tree customers after the site visit may be necessary. Over the past several years, STF has revised their outreach program to include mailed and emailed “tree care tips”, which remind customers about tree maintenance. STF also recently implemented systematic phone calls after the site visit; such follow-up communication was previously more sporadic. We suggest that other urban forest monitoring projects should consider a composite maintenance rating, perhaps specific to their programs’ educational materials and maintenance guidelines, in order to evaluate adherence to recommended practices and potential effects on tree survival (Chapter 4).

Along with the stewardship indicators described above, biophysical factors were also important in the RF model for five-year tree survival (Tables 4, 6). We did not conduct species-specific analysis of survival outcomes due to the wide assortment of species in our sample, with insufficient sample size per species (Table 1). By including functional groups for urban forest management, we were still able to include some species-related information that is useful to practitioners. We observed higher survival for species with low water use demand, and higher survival for trees planted during the rainy season. Although neither of these factors was statistically significant with weighted logrank tests, the importance of these variables in RF analysis indicates that species selection and seasonal patterns are related to young tree survival in this area with summer drought. We also observed that trees with larger mature size died more often (Tables 4, 6, Appendix 1). However, this variable was not highly ranked in variable importance, and water use demand may be the more appropriate way to group species for mortality risk.

We also found that time since delivery (i.e., number of days between tree planting and 2008 field date) was the second-most important variable for first-year survival (Table 4), but was less highly ranked in other models. This points to the relevance of the exact time since planting in newly planted trees, especially in the first year. Trees in large-scale urban forestry programs are often planted during a range of months (e.g., all year-round, or spring and fall seasons only), while field work is often limited to summer. High mortality in the first few years after planting means that a difference of several months in the time elapsed since planting can impact survival calculations. Hence we do not refer to raw proportions of trees alive (Figure 3, Appendix 1) as "survival rates" – they lack the precision of the survival analysis over the entire study period (Figure 4) and are presented for descriptive purposes in interpreting RF results. As the decades pass after planting, precise calculations by day may become less critical in assessing survival.

In terms of neighborhood socioeconomic characteristics from U.S. Census data, income and educational attainment were important in some of our models. While educational attainment was significantly associated with tree planting status (Table 5), with higher planting rates in more well-educated areas, income did not show a consistent pattern for five-year survival outcomes (Table 6). However, our insights into these patterns are limited by our use of neighborhood-scale data, rather than household-scale surveys. We collected these socioeconomic data at the Census tract scale because frequent changes in homeownership made interviews and surveys with every household participating in our study impractical. Additionally, income and housing value during our study period were likely affected by the foreclosure crisis in Sacramento (Immergluck and Smith 2010), therefore other studies may find different relationships between tree mortality and neighborhood-scale socioeconomic patterns. In a study of street tree mortality along a major road in Oakland and Berkeley, CA, young tree mortality rates were higher in neighborhoods with lower socioeconomic status (Nowak et al. 1990). Although income, housing value, and educational attainment were not ranked as the most important variables in our study, they deserve further analysis in future urban tree mortality research, particularly at the household scale.

In summary, our results showed a mix of maintenance, homeownership, and biophysical factors influencing urban tree survival. This supports our overall conceptual framework (Figure 1) and agrees with other research on urban environments as integrated socio-ecological systems (Grimm et al. 2000; Alberti et al. 2003; Cook et al. 2011). Our findings are also consistent with arboricultural practices, which emphasize tree maintenance and climate-appropriate species selection (Harris et al. 2004), and with previous urban forestry research emphasizing the role of maintenance for young tree survival (Nowak et al. 1990; Struve 2009; Boyce 2010; Lu et al. 2010). While human factors are essential to understanding urban tree death, urban forests are not divorced from biological influences (Ramage et al. 2012). Our exploratory investigation of young tree death in Sacramento found that stewardship, consistent homeownership, and climate-appropriate planting strategies affect urban tree establishment. However, there are factors potentially related to urban tree mortality which we were not able to include, such as household values and cultural influences, neighborhood norms, and soil conditions (Figure 1). Future urban forestry research across different age and size classes, and different land uses, may find other dominating risk factors for tree death. To sustainably manage the urban forest (Clark et al. 1997), with planting programs that achieve the desired ecosystem services, it is essential to collect long-term field data to both improve the accuracy of projected environmental benefits and identify major impediments to tree survival (Leibowitz 2012).

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Table 1. Sacramento shade tree species classified by water use demand in California's Central Valley (L = low, M = medium, H = high; UC Cooperative Extension 2000) and expected mature tree size [S = small (<7.6m height), M = medium (7.7-13.7m), L = large(>13.7m)].

Species	**Water use demand**	**Mature size**	**Percent of SFR trees planted (n=370)**
Acer buergerianum	M	M	7.6
Acer campestre	M	M	1.1
Acer rubrum [1]	H	L	15.1
Acer truncatum	M	M	3.2
Betula nigra	H	L	1.6
Betula platyphylla japonica	H	M	3.8
Carpinus betulus	M	M	0.3
Celtis australis	M	L	0.3
Cercis canadensis	M	S	5.4
Crataegus phaenopyrum	M	S	0.8
Koelreuteria bipinnata	M	M	0.3
Koelreuteria paniculata	M	M	1.1
Lagerstroemia indica	L	S	17.0
Malus sp.	M	S	0.5
Nyssa sylvatica	M	M	1.6
Pistacia chinensis	L	M	6.5
Platanus racemosa	M	L	0.3
Platanus x *acerifolia*	M	L	2.7
Pyrus calleryana [2]	M	M	17.8
Quercus castaneafolia [3]	M	L	0.5
Quercus coccinea	M	L	0.8
Quercus douglasii [4]	L	L	0.3
Quercus lobata	L	L	1.4
Quercus macrocarpa [5]	M	L	0.5
Quercus robur	M	L	0.5
Quercus rubra	M	L	3.8
Quercus shumardii	M	L	0.3
Tilia americana	M	L	0.3
Tilia cordata	M	M	1.4
Zelkova serrate	M	L	3.3

[1] *A. rubrum* includes columnar cultivar.
[2] *P. calleryana* includes 'Capital', 'Chanticleer', and 'Redspire' cultivars.
[3] *Q. castaneafolia* is not listed in WUCOLS. Our classification was based on local expertise at STF (L. Leineke, pers. comm.) and the University of California, Berkeley (J.R. McBride, pers. comm.).
[4] *Q. douglasii* was categorized as "very low" in WUCOLS, but "low" in STF promotional materials.
[5] *Q. macrocarpa* is classified in the next version of WUCOLS, to be released later in 2013 (L. Costello, pers. comm.).

Table 2. Explanatory variables for RF analyses of shade trees on SFR properties. For the maintenance characteristics, all variables (except the number of trees distributed) were observed during field work in summer 2008.

Variable	Description	Notes & sources
Biophysical characteristics		
Days since planting	Number of days between tree delivery and field observation 2008	Used to account for the range of planting dates in 2007; we used STF records of tree delivery date used in lieu of precise planting date; value ranged from 182 to 634 days (mean 371)
Season planted	Tree delivery and planting during dry season (May-Sep.) or rainy season (Oct.-Apr.)	
Water use demand	Water use demand for this species in California's Central Valley: low, medium, high	Based on Water Use Classification for Landscape Species (Costello et al. 2000)
Mature tree size	Expected mature tree size: small (<7.6m height), medium (7.7-13.7m), large (>13.7m)	SMUD shade tree cost-benefit calculations assumed higher mortality for small trees (M. Sarkovich, pers. comm.)
Ground cover (2008)	Ground cover at the planting site: maintained grass, mulch, bare soil, rocks/gravel, other maintained vegetation, other unmaintained vegetation	Adapted from i-Tree Eco (itreetools.org)
Health condition (2008)	Numeric code for the health of the tree, with separate ratings for wood (structural health) and foliage (functional health): dying, poor, fair, good	Adapted from i-Tree Streets (itreetools.org)
Socioeconomic characteristics		
Neighborhood level (from US Census American Community Survey 2007-2011)		
Income	Median family income, in 2011 inflation-adjusted US dollars; for survival analysis, this was categorized as low (<50,000), middle (50,000-74,999), high (75,000-99,999), very high (≥100,000)	Median for Sacramento County: $65,720 (U.S. Census Bureau 2013)

Home value	Median home value, in 2011 inflation-adjusted US dollars	Median for Sacramento County: $285,000 (U.S. Census Bureau 2013)
Educational attainment	Percent of population with bachelor's degree or higher; for survival analysis, this was categorized as low (<20%), middle (20-29%), high (30-39%), and very high (≥40%)	Median for Sacramento County: 27.7% (U.S. Census Bureau 2013)
Property level (from Multiple Listing Service)		
Residential homeowner stability	A composite metric combining information about three issues: renter vs. owner-occupancy status, foreclosure status, and change in residential ownership; properties classified as stable had the same residential owner, while properties classified as unstable had renter occupancy, foreclosure, and/or home sales	Used property records for 2007-8 for RF models P and P-1; Used property records for 2007-2012 for P-5 and 2-5
Planting & maintenance practices		
Number of trees distributed	Number of trees delivered to the property by STF for this Tree Care Agreement	STF staff suspected that residences receiving more trees may provide poorer tree care (J. Caditz, pers. comm.)
Yard side	Front yard, back yard	Front yard trees may receive distinct care and landscaping from back yards (Richards et al. 1984, Larsen and Harlan 2006, Daniels and Kirkpatrick 2006, Larson et al. 2009)
Planted in correct location	Tree planted in the sited location agreed to by STF staff and shade tree customer	Correct location assessed by comparing observed planting site to Tree Care Agreement property map
Maintenance rating	Composite rating for tree maintenance: good, adequate, poor	See text for rating system explanation

Table 3. Outcomes and risk factors used in the four RF models. See Table 2 for descriptions of the explanatory variables.

Model name	Outcome	Sample	Risk factors
P	Planted status	Trees on single-family residential properties (n=436)	Neighborhood socioeconomic characteristics, residential homeowner stability, number of trees distributed
P-5*	5th year mortality	Trees on single-family residential properties that were planted (n=357)	Same as model P, plus days since planting, water use demand, and mature tree size
P-1	1st year mortality	Trees on single-family residential properties that were planted (n=370)	Same as model P-5
2-5*	5th year mortality	Trees on single-family residential properties that survived to 1st year of field observations (n=291)	Same as model P-5, plus 2008 ground cover, 2008 foliage and wood health condition, yard side 2008, correct location 2008, maintenance rating 2008

* Models P-5 and 2-5 excluded 13 right censored observations with unknown status 2012. Model 2-5 additionally excluded 21 trees lacking tree health rating and maintenance data from 2008 (i.e., trees which could be seen alive over a backyard fence, but not closely observed).

Table 4. RF results for each model: planting status of delivered trees (n=436, model P), five-year survival of planted trees (n=357, model P-5), one-year survival of planted trees (n=370, model P-1), and four-year survival of trees which survived through the first year (n=291, model 2-5). Variables are listed in rank order from permutation importance (Strobl et al. 2008; Strobl et al. 2009). The variables in brackets have importance values less than the absolute value of the lowest negative score; variables without brackets were considered most relevant for our results (Strobl et al. 2009). Next to each of the most important variables, we indicated whether we observed higher survival (+) or lower survival (-) as the value increased (e.g., higher survival in neighborhoods with higher educational attainment, and lower survival for species with higher water use demand). For binary variables, we noted which category had higher survival (e.g., higher survival for front yard trees).

P	P-5*	P-1	2-5*
neighborhood educ. attain. (+)	homeowner stability (stable +)	homeowner stability (stable +)	homeowner stability (stable +)
homeowner stability (stable +)	species water use demand (-)	days since planting (-)	yard side (front +)
[neighborhood income]	neighborhood income**	neighborhood income**	number of trees delivered (-)
[number of trees delivered]	season planted (rainy +)	neighborhood educ. attain. (+)	maintenance rating (+)
[neighborhood housing value]	mature tree size (-)	mature tree size (-)	[days since planting]
	days since planting (-)	[season planted]	[foliage health]
	[neighborhood housing value]	[number of trees delivered]	[mature tree size]
	[number of trees delivered]	[species water use demand]	[neighborhood income]
	[neighborhood educ. attain.]	[neighborhood housing value]	[species water use demand]
			[correct location]
			[neighborhood educ. attain.]
			[neighborhood housing value]
			[ground cover]

* Models P-5 and 2-5 both excluded 13 right censored trees with unknown status 2012. Model 2-5 also excluded trees for which maintenance characteristics could not be observed in the first summer of field work (2008).
** Survival results were inconsistent across neighborhood income levels.

Table 5. Planting rates for variables identified as most important from RF analysis (Table 4). Reported p-values are for Pearson's chi-squared (homeowner stability) and the chi-squared test for trend for neighborhood educational attainment (Jewell 2004).

Explanatory variable	Planting rate, %
Homeowner stability (p=0.0007)	
stable (n=327)	88.4
unstable (n=109)	74.3
Neighborhood educational attainment (p=0.0140)	
low (n=159)	79.2
middle (n=76)	85.5
high (n=98)	88.8
very high (n=103)	89.3
All trees (n=436)	84.9

Table 6. Survivorship and annual survival estimates for the five-year establishment phase, considering only planted trees and the variables identified as most important from RF analysis (model P-5, Table 4). Survivorship was assessed from Kaplan-Meier survival analysis with Turnbull (1976) estimator for censored observations. Survivorship and estimated annual survival were based on five years after planting, which takes into account the varied planting dates and field observation dates. Unlike RF, we included all right censored trees with unknown status 2012 (n=370 for survival analysis, n=357 for RF). Reported p-values are for the weighted logrank test for interval-censored data (Fay and Shaw 2010).

Explanatory variable	Survivorship 2007-2012, %	Est. annual survival, % (annual mortality)
Homeowner stability (p=0.0019)		
stable (n=222)	76.6	94.8 (5.2)
unstable (n=148)	61.3	90.7 (9.3)
Species water use (p=0.2142)*		
low (n=93)	77.1	94.9 (5.1)
medium (n=201)	66.6	92.2 (7.8)
high (n=76)	72.2	93.7 (6.3)
Neighborhood income (p=0.7877)		
low (n=85)	70.5	93.2 (6.8)
middle (n=136)	73.5	94.0 (6.0)
high (n=106)	66.1	92.1 (7.9)
very high (n=43)	74.2	94.2 (5.8)
Season planted (p=0.0967)		
dry (n=197)	67.7	92.5 (7.5)
rainy (n=173)	75.5	94.5 (5.5)
Mature tree size (p=0.1625)		
small (n=88)	77.7	95.1 (4.9)
medium (n=165)	68.5	92.7 (7.3)
large (n=117)	66.4	92.1 (7.9)
Days since planting (p=0.2401)		
182-370 days (n=192)	71.7	93.6 (6.4)
371-634 days (n=178)	68.1	92.6 (7.4)
All trees (n=370)	70.9	93.4 (6.6)

* We also tried comparing low water use to combined medium and high water use trees. In this case, p=0.09598.

Figure 1. A conceptual model for urban tree mortality as an outcome of integrated socio-ecological systems, adapted from previous frameworks for urban ecosystems (Grimm et al. 2000; Alberti et al. 2003), residential landscapes (Cook et al. 2011), and urban forests (Clark et al. 1997; Dwyer et al. 2003). Many of the human and biophysical characteristics can operate at multiple scales.

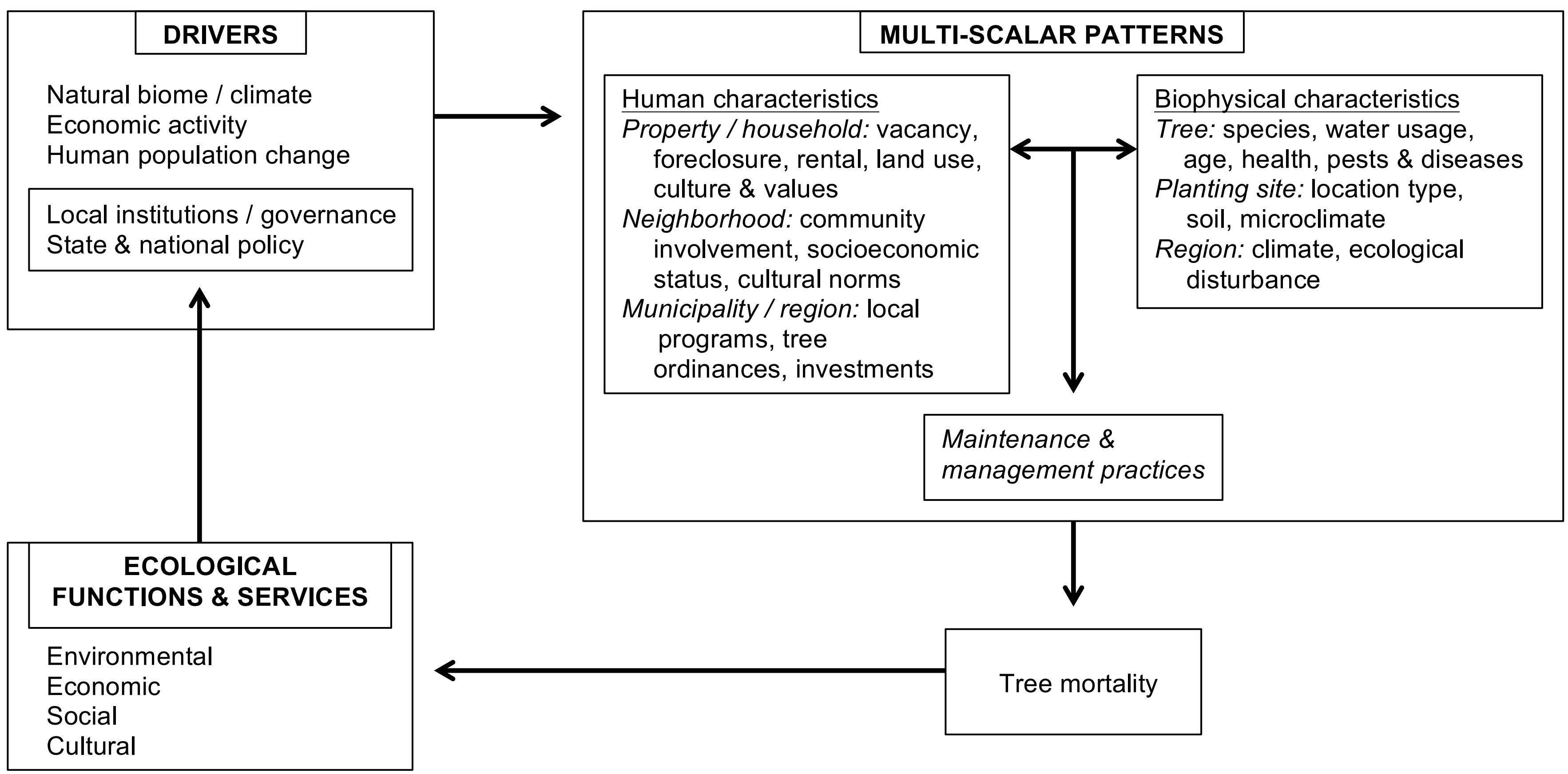


Figure 2. Map of shade tree locations in Sacramento County (medium grey) and Placer County (light grey). Black dots represent planted trees; white dots represented unplanted trees.

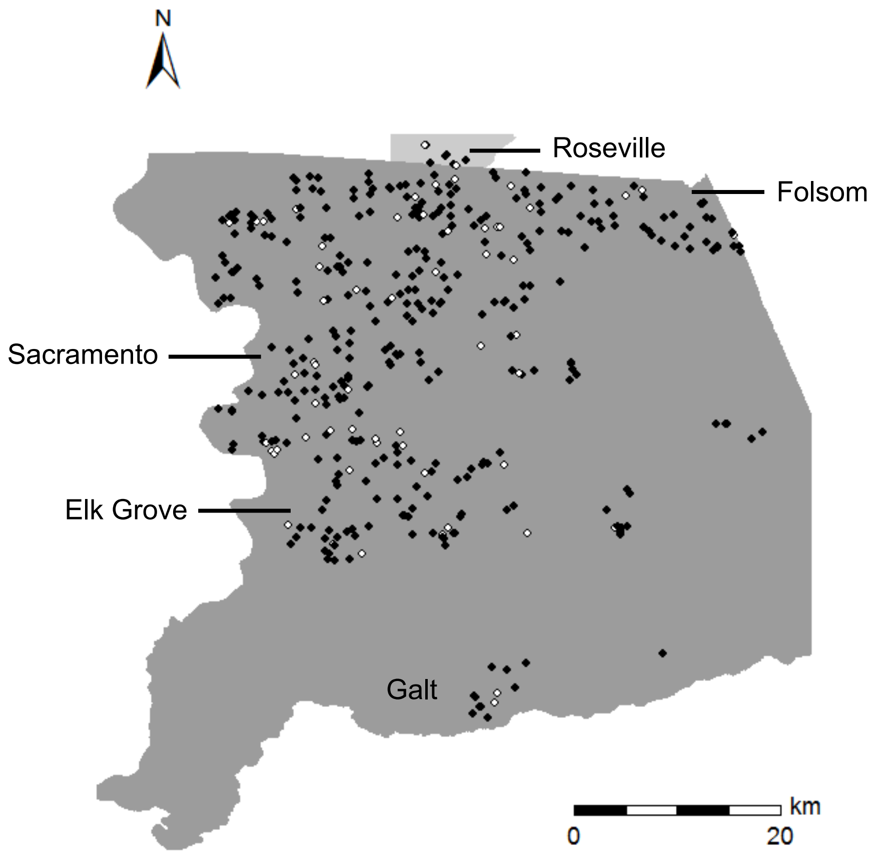


Figure 3. Monitoring outcomes for Sacramento shade trees on SFR properties. Trees were distributed Jan.-Dec. 2007, and field observations of mortality status took place each summer 2008-2012. The numbers reported in this diagram do not represent precise survival rates (see Figure 5 for post-planting survivorship curve).

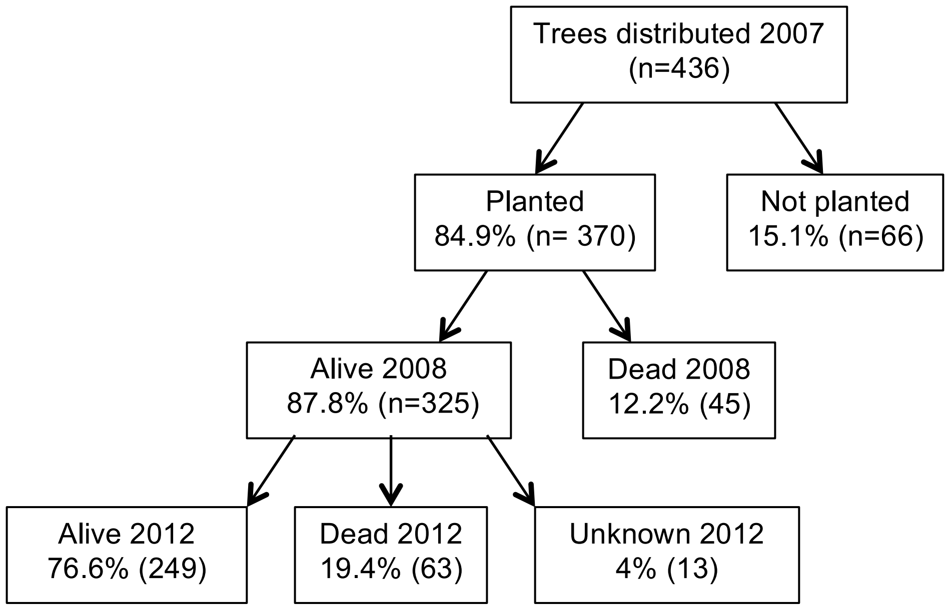


Figure 4. Examples of tree status: live trees in 2012 (a, b), standing dead tree 2008 (c), and trees that were never planted (d).

(a)

(b)

(c)

(d)

Figure 5. Overall survivorship for all planted shade trees (n=370). Survivorship was assessed from Kaplan-Meier survival analysis with Turnbull (1976) estimator for censored observations (Fay and Shaw 2010). The grey rectangles indicate the range of possible values given censoring.

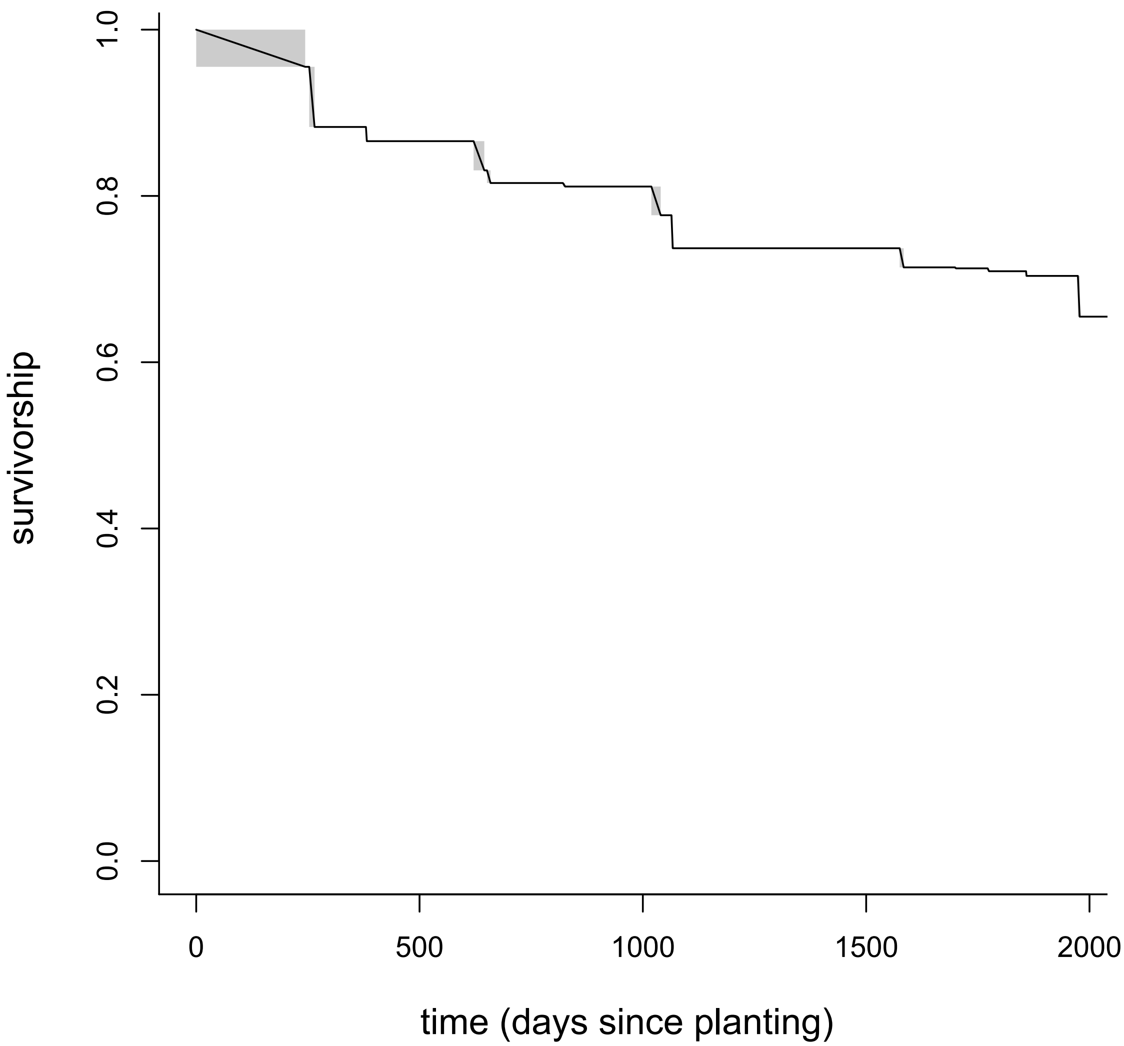
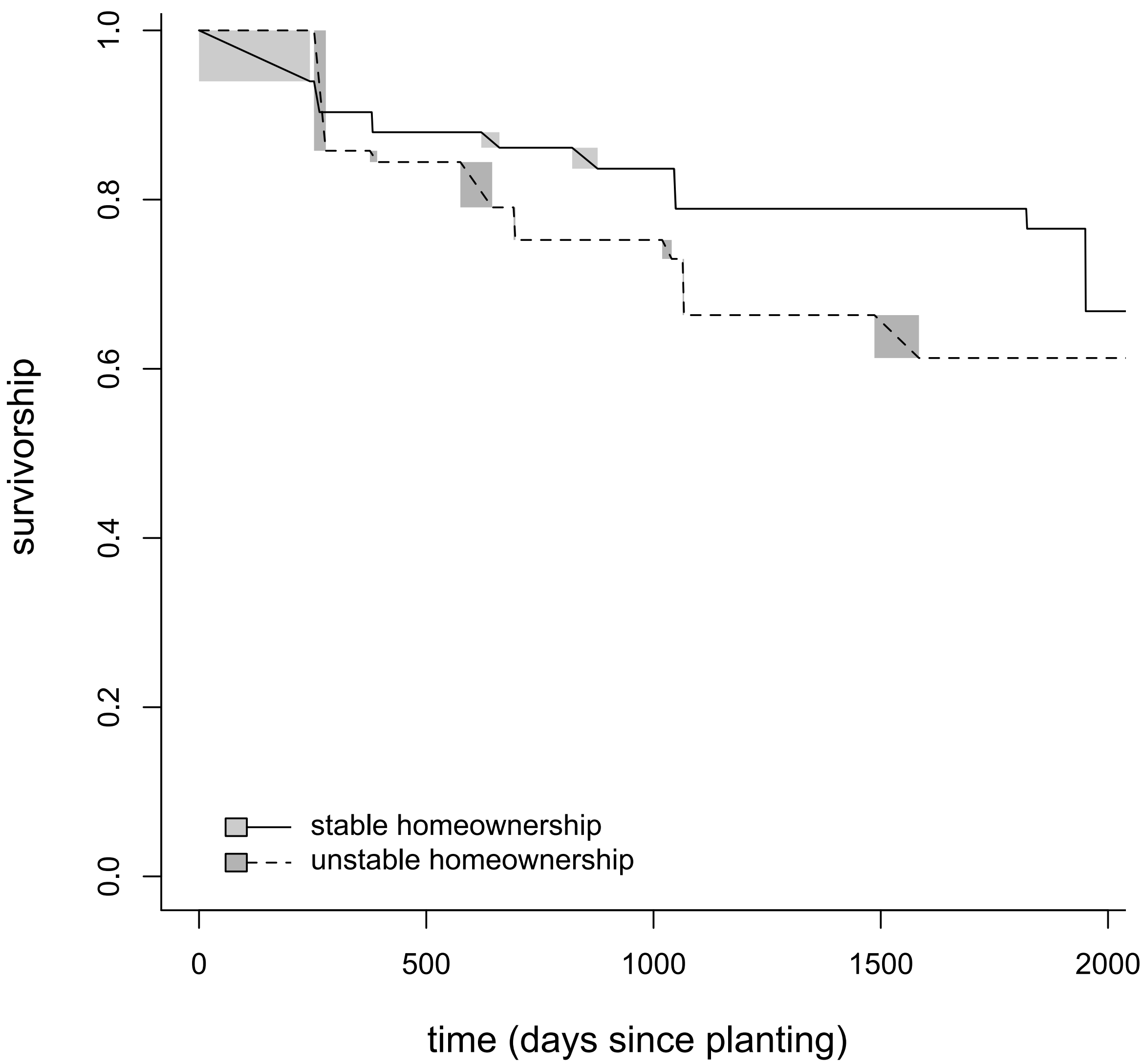


Figure 6. Shade tree survivorship, comparing trees located on properties with stable vs. unstable homeownership over the five-year study period. Survivorship was assessed from Kaplan-Meier survival analysis with Turnbull (1976) estimator for censored observations. A weighted logrank test (Fay and Shaw 2010) shows significant difference (n=370, p = 0.0019). The grey rectangles indicate the range of possible values given censoring.



Appendix 1: Percent of trees alive for models P-1 and 2-5.

Table 7. Percent of trees alive in the first year of field observation (2008) out of the total planted, for the most important variables identified from RF analysis (model P-1). These results do not take into account the varied planting dates and field observation dates, and are presented for descriptive purposes only to aid in interpretation of RF results (Table 4).

Explanatory variable	**% alive 2007-2008**
Homeowner stability	
stable (n=289)	91.7
unstable (n=81)	74.1
Days since planting	
182-370 days (n=192)	91.1
371-634 days (n=178)	84.3
Neighborhood income	
low (n=85)	85.9
middle (n=136)	88.2
high (n=106)	88.7
very high (n=43)	88.4
Neighborhood educational attainment	
low (n=126)	82.5
middle (n=65)	89.2
high (n=87)	90.8
very high (n=92)	91.3
Mature tree size	
small (n=88)	92.0
medium (n=165)	89.1
large (n=117)	82.9
All trees (n=370)	87.8

Table 8. Percent of trees observed alive in the fifth year (2012) that were alive during the first year of field observation (2008), for the most important variables identified from RF analysis (model 2-5). This table omits right censored trees with unknown status 2012, and trees lacking maintenance data 2008. These rates do not take into account the varied planting dates and field observation dates, and are presented for descriptive purposes only to aid in interpretation of RF results (Table 4).

Explanatory variable	**% alive 2008-2012**
Homeowner stability	
stable (n=176)	86.4
unstable (n=115)	72.2
Yard side	
front (n=128)	85.9
back (n=163)	76.7
Number of trees delivered	
1-3 (n=139)	83.5
4-6 (n=94)	80.9
>6 (n=58)	74.1
Maintenance rating	
good (n=66)	89.4
adequate (n=150)	80.7
poor (n=75)	73.3
All trees (n=291)	80.8

Appendix 2: Survivability definition.

SMUD defines survivability as the proportion of trees alive divided by the total distributed. Unknown or "hidden" trees are omitted from this calculation. With our field data (Figure 3), the five-year survivability is 249/(436-13) = 59%. SMUD defines mortality as 1 – survivability, therefore SMUD's reported mortality rates include trees that were not planted. Understanding our different mortality and survival terminology was an integral part of early conversations with STF to develop the field data collection plan and record tree status (see dissertation preface). The survivability rates and mortality rates from SMUD's previous studies (Lindeleaf 2007) are difficult to compare to other reported mortality studies that only consider post-planting tree death.

Chapter 4

Identifying common practices and challenges for local urban tree monitoring programs across the United States

Abstract

Urban forest monitoring data are essential to assess the impacts of tree planting campaigns and management programs, and to assess trends in tree mortality, growth, and health. Local practitioners have monitoring projects that have not been well documented in the urban forestry literature. To learn more about practitioner-driven monitoring efforts, we surveyed 32 local urban forestry organizations across the United States about the goals, challenges, methods, and uses of their monitoring programs. Non-profit organizations, municipal agencies, state agencies, and utilities participated. One-half of the organizations had six or fewer urban forestry staff. Common goals for monitoring included evaluating the success of tree planting and management, taking a proactive approach towards tree care, and engaging communities. Programs employed a wide range of monitoring methods. The most commonly recorded data were species, health condition rating, mortality status, and diameter at breast height. Challenges included limited staff and funding, difficulties with data management and technology, and field crew training. Programs used monitoring results to inform tree planting and maintenance practices, provide feedback to individuals responsible for tree care, and manage hazard trees. Participants emphasized the importance of planning ahead: carefully considering what data to collect, setting clear goals, developing an appropriate database, and planning for funding and staff time. To improve the quality and consistency of monitoring data across cities, researchers can develop standardized protocols and be responsive to practitioner needs and organizational capacities. Urban foresters can also learn from the best practices of long-term monitoring in other ecological systems.

Keywords

citizen science, Forest Inventory and Analysis, i-Tree, Long-Term Ecological Research, participatory research, questionnaire

Introduction

The proliferation of urban forest inventory systems in the past few decades has allowed practitioners and researchers to quantify forest structure and function, estimate ecosystem services, and manage tree maintenance issues (Miller 1996; McPherson et al. 1999; Nowak and Crane 2000; Brack 2006; Keller and Konijnendijk 2012). Standardized inventory systems have enabled comparisons of tree density, species composition, and cost-benefit ratios across cities (McPherson and Simpson 2002; McPherson et al. 2005; Nowak et al. 2008). While these inventories have enhanced our understanding of urban forests, they provide a snapshot in time, and can quickly become outdated in a changing, complex urban landscape. Long-term monitoring data are essential to understand change over time in urban forests – including trends in tree mortality, growth, longevity, and health – and to assess the impacts of tree planting campaigns and management programs.

Although urban forest researchers and arborists have long recognized the value of monitoring data and systematically updated inventories (Weinstein 1983; Baker 1993; McPherson 1993; Clark et al. 1997; Dwyer et al. 2002, Rysin et al. 2004), we do not yet have coordinated programs to conduct longitudinal studies. The need for long-term monitoring was raised at a recent conference on urban tree growth and longevity (Leibowitz 2012). There have been several long-term monitoring programs in wildland (i.e., non-urban) forest ecosystems in the United States, including the Forest Inventory and Analysis (FIA) program of the United States Department of Agriculture (USDA) Forest Service and Long-Term Ecological Research (LTER) sites sponsored by the National Science Foundation. The LTER sites were developed with a recognition that many ecological phenomena operate over decades, and longer, requiring long-term investment in data collection (LTER Network 2011). The FIA program (fia.fs.fed.us) serves as a census for forest ecosystems in the United States (Smith 2002), with recent integration of the Forest Health Monitoring (FHM) program (fhm.fs.fed.us) and annual field measurements (McRoberts et al. 2005) to generate longitudinal data. The FIA and FHM programs have generated rich data sets for researchers to study tree mortality, growth, and health (e.g., Lessard et al. 2001, Shaw et al. 2005; Woodall et al. 2005; Komroy et al. 2008; Klos et al. 2009; Lines et al. 2010). Globally, the Center for Tropical Forest Science (CTFS) is a network of dozens of tropical and temperate plots, all following the same methods to re-census trees every five years (Condit 1995; www.ctfs.si.edu).

Although the above forest monitoring programs focus primarily on non-urban systems, the methods and analytical tools can be adapted to urban systems. This is already happening with FIA urban pilot programs (Cumming et al. 2008). The Forest Service has also collected repeated plot-based data using i-Tree Eco in Baltimore, MD and Syracuse, NY (Nowak et al. 2004; D.J. Nowak, personal communication). There are two LTER sites in urban environments: Baltimore, MD (beslter.org) and Phoenix, AZ (caplter.asu.edu). In Phoenix, annual tree surveys are underway (caplter.asu.edu). While urban forest researchers begin to pursue long-term data collection, local practitioners are also engaged in monitoring. Two examples have been published in an online open-access journal (Boyce 2010; Lu et al. 2010), but other local monitoring programs exist that are not well documented in the literature. Standardized protocols for urban tree monitoring would promote data sharing among professionals and researchers, and advance local monitoring projects already underway (Leibowitz 2012). Tree monitoring partnerships would provide urban forestry professionals with improved tree growth and mortality information to evaluate the success of planting and management programs, while expanding the data sets available to researchers. Standardization would also enable comparisons across municipalities.

To assist in the development of standardized urban forest monitoring protocols, we sought to learn more about the goals and operations of practitioner-driven monitoring. We disseminated a questionnaire to urban forestry organizations across the United States, specifically targeting local organizations that already conduct monitoring programs. The survey assessed: 1) common goals and motivations for monitoring; 2) the range of methods employed; 3) common challenges; and 4) uses of monitoring data. We also asked participants to offer suggestions for other local organizations seeking to collect monitoring data, and for researchers aiming to develop standardized protocols.

Methods

Study design and participant recruitment

We targeted local urban forestry organizations in the United States that have collected urban tree monitoring data; only organizations with longitudinal data on individual trees were relevant to our research. Throughout this paper, we use the term 'monitoring' to refer to systematically collected data on the same trees over time (i.e., longitudinal data), and 'inventory' to refer to a one-time snapshot of urban forest characteristics. Organizations with lists of planted trees lacking follow-up records, static inventories, or sporadically updated inventories were not included in this study.

To understand practitioner-driven monitoring efforts, we specifically sought monitoring programs developed and led by local urban forestry organizations, rather than researcher-driven monitoring studies (e.g., Cumming et al. 2001; Nowak et al. 2004; Cumming et al. 2007; Cumming et al. 2008). Eligible organizations were identified through researcher and peer recommendations. We began with a list of a dozen organizations that were known or suspected to have relevant monitoring programs. This list was generated by L.A. Roman and E.G. McPherson, with mainly West coast and East coast organizations, supplemented by suggestions from colleagues in the South and Midwest. Next, we used a snowball or chain referral sampling technique, asking for peer recommendations from staff at the local organizations already identified. Sixty-seven organizations were identified through this process.

We recruited participants via email in Feb.-Apr. 2012, followed by a phone call to explain the study purpose and verify whether the organization had relevant urban tree monitoring programs. Seventeen organizations did not have relevant monitoring programs, 16 were unresponsive to our recruiting attempts, and 34 agreed to participate in the study. We emailed questionnaires to staff at each of the 34 recruited organizations, with several reminder emails and phone calls as needed. Questionnaire design and recruitment techniques were adapted from Dillman's Tailored Design Method (Dillman 1999). Thirty-two organizations completed the survey (Table 1), a 94% participation rate of those recruited. Most participants completed the survey via email, and one dictated responses over the phone.

Survey format

The survey contained organization-level and program-level questions. Some organizations had more than one distinct monitoring program; in these situations, the program-level questions were repeated. Surveys were customized to each organization with the name(s) of their program(s). Forty-five distinct monitoring programs were included from the 32 participating organizations. Organization-level questions included the type of organization, number of paid urban forestry staff, challenges with urban tree monitoring, experiences sharing monitoring methods and results, and recommendations for other local organizations and researchers undertaking monitoring projects. The number of full-time equivalent paid staff was limited to individuals working on urban forestry and urban greening issues. This enabled more meaningful comparison of staff at different organizations (e.g., municipalities reported the number of urban forestry employees in the parks and/or streets division, rather than the total staff across all departments). Program-level questions included motivations for the specific monitoring program(s), processes of developing field methods, types of data collected, and uses of monitoring data.

The survey included both multiple-choice and open-ended questions. Multiple-choice questions were usually presented as “check all that apply”, including an option for “other”, to account for categories we had not anticipated. We re-coded responses in the “other” category to fit the original categories whenever possible (i.e., we determined that the participant’s explanation for the “other” response fit a category already listed). In a few cases, several participants gave similar responses for the “other” explanation, and we created new response categories.

Data analysis

Open-ended questions were qualitatively assessed for common themes, counting the number of times participants mentioned similar ideas. Themes were not pre-determined. The open-ended questions were independently analyzed by one of the authors and a research assistant, with later discussion to resolve discrepancies in the interpretations. Differences in interpretation typically related to lumping vs. splitting topics. Direct quotations from participants are included to provide a deeper view of their experiences and perspectives. Quotes are presented anonymously, with spelling errors corrected.

We present results for both the open-ended and multiple-choice questions as a percent of the total number of organization-level (n_{org}) or program-level (n_{prog}) responses. In some cases, responses were left blank, and in those situations we divided by the total number of actual responses for that particular question. For both multiple choice and open-ended questions, percentage totals are typically >100%, because respondents were not forced to chose only one option.

Results

Types of organizations represented

Participating organizations (Table 1, n_{org}=32) were mainly non-profits (53%) and municipalities (38%), with a smaller proportion of state governments (9%) and utilities (6%). Most non-profit organizations are focused on urban forestry and urban greening; two are neighborhood associations. The organizations serve a range of geographic areas: cities/municipalities (72%), counties (31%), regions (25%), neighborhoods (22%), states (6%), and other (6%). The number of full-time equivalent urban forestry staff of these organizations also varies widely, with 50% of organizations having six or fewer staff (Table 2).

Goals and motivations for monitoring

The most common goal (51%) for urban tree monitoring programs (n_{prog}=43) was to track tree survival, health, and/or growth, and measure program success. 'Success' itself was generally not clearly defined by respondents, but tree survival and health were implied. Some programs also aimed to evaluate factors related to survival, such as species, planting stock, and stewardship. Two participants explained their program goals as follows:

> *[Our organization] had an assumed survival rate when I started, but nothing to back it up. I wanted to have a legit number that we can claim as the success of our planting and care work.*
>
> *The sense that we were losing trees as fast as they were being planted made me/us want to see whether that was true, so getting some data together was essential to know if we were in fact gaining or losing ground.*

Another common motivation was conducting monitoring as a proactive approach towards tree care, maintenance, and management (44%). Monitoring data collection was sometimes done at the same time as, or in preparation for, tree maintenance work. Related to this, 14% of programs conduct monitoring to effectively manage mature and hazardous trees for pruning, disease, and litigation issues. Twenty-one percent of programs conducted tree monitoring to educate and engage volunteers, residents, and communities.

Tree monitoring programs were sometimes required by grants or contract obligations; 16% of programs mentioned this as part of their motivation for conducting monitoring. Of all programs (n_{prog}=45), 51% had external funding, and of these (n_{prog}=23), monitoring was required for 48%.

<u>Monitoring methods</u>

Programs (n_{prog}=41) developed their field methods for urban forest monitoring using a mix of in-house program staff (46%) and external assistance (17%). Participants worked with paid consultants, university or USDA Forest Service researchers, and other local urban forestry organizations. Some programs (12%) adapted their monitoring methods from the i-Tree inventory system (www.itreetools.org), which was developed by the Forest Service. Field work was carried out (n_{prog}=45) mostly by program staff (62%), followed by volunteers (42%), arborists (36%), researchers (16%), interns (16%), and contractors (4%). Thirty-three percent of programs (n_{prog}=45) developed a field manual for their monitoring project.

The most commonly recorded tree characteristics for urban tree monitoring programs (n_{prog}=45) were species (96%), health condition rating (89%), mortality status (76%), diameter at breast height (DBH; 71%), and specific health problems (67%). Many other tree size metrics, maintenance issues, and site characteristics were recorded (Table 3). Half (53%) of the programs (n_{prog}=45) exclusively monitor trees planted by their organization, while others monitor only trees not planted by their organization (9%) or both (38%). Street trees were the most common (86%) type of tree location included in these programs (n_{prog}=44), followed by public park trees (45%), institutional trees (34%), residential yard trees (25%), conservation areas (7%), and other (14%). The most common way to record tree location was street address (78%), with many other techniques employed (Table 4; n_{prog}=45); tree location was often recorded in several ways.

The sampling designs for these monitoring programs (n_{prog}=45) also varied widely. Seventy-three percent used a complete survey of all trees in a particular program or neighborhood, 16% used a stratified random sample, 9% used a simple random sample, 7% used a convenience sample, 7% used a targeted sample, and 4% used another sampling technique. In terms of observation intervals (n_{prog}=44), 64% of programs used a fixed time interval, 43% used a one-time monitoring of recently planted trees, 18% used a rolling schedule (e.g., visit 20% of all trees every year, to reach all trees in 5 years), and 30% used another observation interval.

Monitoring data were managed using a wide assortment of software (n_{prog}=45), including Excel (49%), Access (44%), GIS (22%), i-Tree (18%), Lucity (7%), TreeKeeper (4%), and other (20%). Thirty-seven percent of programs (n_{prog}=43) have a paid staff person dedicated to management of tree monitoring databases.

Challenges with monitoring

Resource limitation was the most common challenge (63%) to urban tree monitoring at these organizations (n_{org}=32). Specifically, 50% mentioned lack of staff time and 25% mentioned lack of dedicated funding. Data management and technology challenges were also common (47%), such as time-intensive data entry and management, identifying appropriate software for long-term tree records, and adapting other technologies for tree monitoring. Twenty-eight percent of organizations had challenges developing protocols, including deciding what data to collect, subjectivity of tree health ratings, and instituting quality assurance and quality control. Twenty-five percent had difficulties with field crew recruitment and/or training, especially for volunteers and student interns. Twenty-five percent had problems implementing the field work, such as reliably locating tree and plots, and getting access to private properties. One participant captured the problem of locating trees:

> *[We] actually need to have the capacity to revisit these same trees and distinguish them from others planted in the same field. Coordinates aren't accurate enough to achieve this, a map must be made, which is tricky in itself.*

Another participant summarized many of the common challenges:

> *Not knowing what to monitor, no one to monitor, not knowing what questions to ask of the monitoring.*

Organizations had many solutions to these challenges (n_{org}=32). Twenty-five percent improved the process of recruiting and training field crews, particularly organizations relying on volunteers and student interns. Twenty-five percent explained their solutions to lack of funding, including incorporating monitoring and staff time into organizational budgets, seeking external grants, and using volunteers. Thirteen percent prioritized data collection to meet immediate management needs, such as hazard tree issues. Other solutions were staff and volunteer dedication (9%), and advice from external consultants or peers (9%). Twenty-two percent of organizations noted that challenges remain and have not been solved.

Uses of local monitoring data

We asked participants whether monitoring programs influence management at their organizations; 78% said yes (n_{prog}=45). Of these (n_{prog}=35), 60% said that monitoring informs tree planting techniques and maintenance practices. Forty-three percent said that monitoring affects tree species selection, helping to maximize diversity and select appropriate species. Twenty-three percent used monitoring to provide feedback to individuals responsible for tree care, such as residents, volunteers, contractors, and municipalities. Twenty percent used monitoring data to manage mature and hazardous trees, often connected to liability concerns.

Data analysis at these programs (n_{prog}=42) involved summary statistics (81%), overall survival and/or growth rates (69%), comparisons of survival and growth across groups (50%), spatial analysis (31%), statistical analysis such as chi-squared and ANOVA (19%), and other techniques (17%). Data analysis (n_{prog}=36) was carried out by program staff (83%), interns (8%), researchers (8%), volunteers (8%), and consultants (3%). Thirty-seven percent of programs (n_{prog}=43) have a paid staff person dedicated to database management for the monitoring project. Sixty percent of programs (n_{prog}=45) produced written reports on their monitoring projects; two of these were published in *Cities and the Environment*, an online open access journal (Boyce 2010; Lu et al. 2010).

Sharing monitoring methods and results

We asked participants whether their organizations shared information about their tree monitoring program(s) with other urban forestry organizations; 56% said yes (n_{org}=32). Information was shared through a variety of mechanisms. Fifty-six percent of those who share information (n_{org}=18) did so through direct communication with colleagues at other organizations, 33% shared through state or regional networks, and 22% shared at conferences. One participant described the uses of a regional network:

> *We have an email network of foresters in [our area]. It is very helpful to post questions. If one person has any questions, they can post a question and others will reply. We have used this feature to ask questions and give advice, share reports, manuals, specs and countless other things. It is invaluable.*

Participants (n_{org}=29) described the value in sharing monitoring methods and results across cities. Fifty-five percent valued the opportunity to learn from the best practices and methods in other cities and programs. One participant whose organization has not shared monitoring information with others captured the perceived benefits:

> *Our monitoring methods are imperfect, and we would love to learn from other programs that may operate similarly to us - what methods work for them, how they deal with similar challenges.*

Twenty-one percent commented that sharing methods and approaches can save time and increase efficiency:

> *It increases efficiency - you don't have to "re-create the wheel" for each tree planting/monitoring program. We can learn from other's experience.*

Organizations also valued the ability to share findings across cities and programs (21%), with some specifically noting the value of standardized methods for meaningful comparison of data (17%).

Suggestions for other practitioners and researchers

We asked participants to offer guidance to another local urban forestry organization seeking to develop a tree monitoring program. Most recommendations (n_{org}=31) addressed the importance of advance planning. Fifty-two percent of respondents emphasized the importance of thinking carefully about methods and data collection. Forty-two percent said that monitoring programs should have clear goals and intended uses of the data. Forty-two percent mentioned the importance of a good database, especially the initial inventory or planting records. Twenty-nine percent suggested planning ahead for budgeting, funding, staffing needs, and field crew time. A participant captured many of these common recommendations:

> *They need to know what the purpose is for the information. If you're taking the time to do it, what's the point? This helps drive what data you collect. Know who is going to do the work, and make sure they have the time and experience to do it properly.*

We also asked participants how researchers can be useful to enhance their urban forest monitoring program(s). Forty-four percent (n_{org}=27) asked researchers to provide best practices and methods for monitoring, including standardized protocols. One participant noted that small organizations have limited capacity, and would appreciate input from researchers on best practices and protocols:

> *Our small organization does not have the capacity to do this research ourselves and search for and interview other programs. By providing info on what other programs do, suggesting protocols, and providing guidance, researchers could help us improve our work.*

Twenty-two percent of organizations suggested that researchers should develop tools for monitoring, such as technology and software solutions. Nineteen percent requested that researchers continue to produce information on tree benefits and ecosystem services, which help justify funding for urban forest programs. Fifteen percent would like researchers to provide accurate estimates of tree mortality, growth, and canopy change. Eleven percent noted that university and/or government researchers have already been useful.

Finally, we asked for recommendations with the development of standardized urban tree monitoring protocols. Thirty-one percent (n_{org}=29) suggested that protocols should be adaptable to different organizational capacities and needs, and flexible for different situations:

> *It would be helpful if the standardized protocols are developed with various respondents' program designs/capacities in mind, that information is supplied suggesting the relevance/appropriateness of suggested protocols to the diversity of programs.*

Another suggestion (21%) was to be inclusive and involve practitioners:

> *This is a great place to start. Update everyone as to your findings and get everyone together to talk about it.*

Some participants (21%) stressed the importance of keeping protocols simple for users, rather than “complicated and academic”:

> *Keep it simple. The more complex, the less likely all groups planting trees will be able to use the protocol.*

Discussion

Common goals and motivations for practitioner-driven urban forest monitoring emerged from our analysis. These goals were often echoed in later responses about field methods and uses of the data. For example, some programs that evaluate trees planted by their organization tracked tree survival and health, and used the results to inform planting practices. Other programs that manage mature urban trees tracked potential hazard trees, and used the results to prioritize maintenance. Regularly updating inventories for hazard tree management has been advocated previously in the arboriculture literature (Weinstein 1983; Anderson and Eaton 1986). However, not all programs had clear linkages between monitoring goals, field methods, and uses of the data. At the same time, when we asked participants to offer guidance for other organizations embarking on tree monitoring programs, the most common recommendations were to carefully consider what data to collect and have clearly articulated goals.

Research ecologists have similarly stressed the importance of clear questions and objectives in long-term monitoring (Lindenmayer and Likens 2010). Monitoring is not a goal in and of itself, but rather, a means to answering questions (Lovett et al. 2007; Lindenmayer and Likens 2010). Other attributes of effective ecological monitoring are dedicated leadership and institutional commitment, strong partnerships among scientists, resource managers, and policy-makers, careful selection of core variables to measure, frequent use of the collected data, plans for long-term data accessibility, and an adaptive monitoring framework that responds to new technologies and research questions (Lovett et al. 2007; Lindenmayer and Likens 2009; Lindenmayer and Likens 2010). Monitoring projects that lack strong research questions and plans for data analysis may become "snowed by a blizzard of ecological details" from a poorly focused "laundry list" of measurements (Lindenmayer and Likens 2010). The "data-rich but information-poor" scenario in environmental monitoring programs (Ward et al. 1986) has led to monitoring programs being criticized as unscientific (Lovett et al. 2007; Lindenmayer and Likens 2009). While these comments are focused on monitoring for academic and research purposes, long-term ecological datasets often address basic research goals while generating useful data for environmental managers and policy-makers (Magurran et al. 2010; Lindenmayer and Likens 2010). The same guidelines for effective monitoring apply to urban forests, where long-term monitoring can produce data for both researchers and practitioners; similar ideas were raised by our survey participants.

Linking planting grants to monitoring and maintenance funds would be one step forward towards addressing the hurdle of resource limitations faced by many local monitoring programs. One-quarter of the programs we surveyed were required to monitor due to grant obligations. Urban forestry initiatives should tout exemplary records of tree survival and health, rather than shear numbers of trees planted. With increased interest in urban tree monitoring from funders, more local organizations may begin monitoring, or formalize their existing programs. Additionally, regulatory-based programs, such as California's cap and trade offset program (California Air Resources Board 2011), allow for urban tree planting as a mitigation measure because of projected ecosystem services (McHale et al. 2007; Poudyal et al. 2011). Regulatory protocols that provide guidance based on best management practices are including provisions for reporting survival and growth. Reliable funding sources have also been a concern in long-term environmental and ecological monitoring (Caughlan and Oakley 2001; Lovett et al. 2007), and dedicated funding from national agencies has been important for long-term ecological research in the United States (e.g., LTER and FIA). Finding consistent funding for long-term urban tree monitoring is likely to require new partnerships among federal and state agencies, industries, and non-profits.

Survey participants encountered other challenges with urban forest monitoring that were previously raised by Baker (1993): consistency in field crew training, accurately recording tree location, and managing data. Existing inventory software often did not meet participants' needs for long-term data collection. Researchers can significantly improve the quality and consistency of monitoring data across cities by developing standardized protocols, offering technology solutions, and being responsive to practitioner needs and organizational capacities. There is wide variation in the methods currently used in practitioner-driven urban forest monitoring, with a range of measurements recorded, tree location types included, and sampling designs. Standardized monitoring protocols can extend from existing urban forest data standards (www.unri.org/standards) and inventory methods (Miller 1996; McPherson et al. 1999; Nowak and Crane 2000; Brack 2006; Keller and Konijnendijk 2012), with special attention to issues that are unique to long-term data collection, such as managing longitudinal datasets and accurately recording tree location and DBH growth. Technology solutions for monitoring could include mobile interfaces for data collection and remote sensing to reduce the need for costly ground-based approaches. In offering suggestions for standardized protocols, survey respondents urged researchers to "keep it simple", rather than "complicated and academic", to enable more organizations to participate. Researchers must remain cognizant of the fact that many local organizations engaged in monitoring have a small number of urban forestry staff (one-half with six or fewer, one-quarter with three or fewer), and that most local organizations do not have staff dedicated to database management. Developing, implementing, and analyzing long-term monitoring projects are significant challenges for organizations with few staff and limited resources. By providing standards and best practices, researchers can enhance the institutional capacity of these organizations to generate rigorous data that addresses their management needs. Standardization would also promote the sharing of information amongst practitioners. While our survey participants recognized many values in sharing monitoring approaches and results, only about half currently share their results and methods with other organizations.

Collaboration between researchers and practitioners will be essential to develop effective monitoring standards and implement long-term data collection. Dialogue between researchers, managers, and arborists has been central to urban forestry for many decades, recognizing the strengths that each party brings to collaborations, as well as the difficulties in two-way communication (Shigo 1976; Dwyer 1987). Participants in our survey requested that researchers have an inclusive process to develop standards, and create flexible protocols adaptable to different organizations' needs. To maintain an inclusive approach, we presented preliminary results at the International Society of Arboriculture's 2012 Annual Conference and Trade Show, with a special symposium on urban tree monitoring featuring presentations by the USDA Forest Service, several non-profits and municipalities, and a panel discussion. The panel discussion helped to launch an advisory committee of practitioners and researchers that will collaboratively develop protocols.

Collaborative, community-based, and participatory approaches are increasingly common in other disciplines, such as public health (O'Fallon and Dearry 2002; Leung et al. 2004; Minkler and Wallerstein 2011), city planning (Forester 1999; Rotmans and Van Asselt 2000), and natural resource management (Fortmann 2008; Wilmsen 2008). Following from the principles of community-based participatory research (Israel et al. 1998; O'Fallon and Dearry 2002), local urban forestry organizations should be involved with setting goals, developing methods, collecting data, and disseminating results. For example, Wolf and Kruger (2010) used structured discussions among urban forest managers, professionals, and researchers in the Pacific Northwest to identify and prioritize research topics. Urban forestry practitioners can contribute their professional expertise and insights into local conditions, thereby enhancing the quality of the research. Continued dialogue between researchers and practitioners will be necessary to ensure that future urban forest monitoring projects are both scientifically rigorous and useful for local management concerns.

Furthermore, data collection for urban forest monitoring programs often involves local residents and volunteers; one-fifth of organizations we surveyed stated that community engagement was a motivation for monitoring, and nearly one-half of the programs relied on volunteers for data collection. Volunteer-based data collection and citizen science in urban forestry can promote environmental awareness and create a more informed constituency (Bloniarz and Ryan 1996; Cooper et al. 2007; Abd-Elrahman et al. 2010). Citizen science is also employed in long-term ecological monitoring in other systems (Silvertown 2009; Dickinson et al. 2010; Magurran et al. 2010; Dickinson et al. 2012), such as the Audobon Society's Christmas Bird Count (birds.audubon.org/christmas-bird-count). Volunteer-based data collection has increased the geographic scale and magnitude of ecological monitoring projects. While data collected by volunteers has the potential for error and bias, the extent of this error is poorly understood (Dickinson et al. 2010). Errors can be minimized with data validation procedures whereby scientists follow-up on data entries flagged as potential problems (Bonter and Cooper 2012; Gardiner et al. 2012). Bloniarz and Ryan (1996) found that with adequate training, volunteer-based urban tree inventories can produce mostly accurate data at lower cost than professional arborists. Our survey participants also noted that effective volunteer and intern training is essential to producing high-quality data. Training for urban forest monitoring can build on training materials already developed for inventory systems, such as i-Tree workshops (www.itreetools.org) and webinars for OpenTreeMap (http://www.azavea.com/products/opentreemap), in addition to training techniques employed by participants in our questionnaire.

To the best of our knowledge, this is the first comprehensive survey of local urban tree monitoring programs. As such, it provides information to establish a baseline for best practices in urban forest monitoring. Our results and conclusions may be biased due to the limited sample size; there may be other urban tree monitoring programs in the United States that we unintentionally omitted. Nevertheless, we assert that by including 32 organizations with a range of characteristics and monitoring methods, we have gathered sufficient information to assess the goals, challenges, methods, and uses of practitioner-driven monitoring. The observations gleaned from our survey can inform the next generation of urban tree monitoring, with researchers and practitioners collaborating for long-term data collection.

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Table 1. Participating organizations (n_{org}=32).

Organization name	City	Organization type
Canopy	Palo Alto, CA	Non-profit
Casey Trees	Washington, DC	Non-profit
Chicago Dept. of Streets & Sanitation, Bureau of Forestry	Chicago, IL	Municipal agency
City of Austin, Urban Forestry Program	Austin, TX	Municipal agency
City of Bowling Green, OH	Bowling Green, OH	Municipal agency / Utility
City of Charleston, Dept. of Parks	Charleston, SC	Municipal agency
City of Pasadena, Public Works, Div. Parks & Natural Resources	Pasadena, CA	Municipal agency
East Hill Council Neighbors	Grand Rapids, MI	Non-profit
Friends of Greenwich Streets	New York City, NY	Non-profit
Friends of the Urban Forest	San Francisco, CA	Non-profit
Friends of Trees	Portland, OR	Non-profit
Mass. Dept. Conservation and Recreation	MA	State agency
Missouri Dept. Conservation, Community Forestry	MO	State agency
Morris Arboretum	Philadelphia, PA	Non-profit
New Jersey Tree Foundation	NJ	Non-profit
NYC Parks and Recreation	New York City, NY	Municipal agency
Ohio DNR	Silver Lake, OH	Municipal agency / State agency
Our City Forest	San Jose, CA	Non-profit
Pennsylvania Horticultural Society	Philadelphia, PA	Non-profit
Sacramento Municipal Utility District	Sacramento, CA	Utility
Sacramento Tree Foundation	Sacramento, CA	Non-profit
Savannah Tree Foundation	Savannah, GA	Non-profit
Seattle Dept. of Parks & Recreation	Seattle, WA	Municipal agency
The Park People	Denver, CO	Non-profit
Tree People	Los Angeles, CA	Non-profit
Tree Trust	Minneapolis & St. Paul, MN	Non-profit
Trees Pittsburgh	Pittsburgh, PA	Non-profit
Urban Resources Initiative	New Haven, CT	Non-profit
Village of Downers Grove	Downers Grove, IL	Municipal agency
Village of Lombard, Illinois	Lombard, IL	Municipal agency
Village of Mount Prospect, Illinois	Mount Prospect, IL	Municipal agency
Village of Oak Park, Illinois	Oak Park, IL	Municipal agency

Table 2. Number of full-time equivalent paid staff working on urban forestry and urban greening at participating organizations (n_{org}=32).

	Non-profit organizations (17)	Municipalities, utilities, and state governments (15)	All organizations
min	0	2	0
25th percentile	3	3	3
median	4	7	6
75th percentile	21	22	22
max	90	174	174

Table 3. Data collected by urban tree monitoring programs (n_{prog}=45).

Data collected	Percent
Tree characteristics	
Species	96%
Health condition rating	89%
Mortality status	76%
Diameter at breast height	71%
Specific health problems	67%
Height	38%
Canopy width	31%
Canopy dieback	27%
Maintenance issues	
Pruning	56%
Watering	47%
Mulching	47%
Infrastructure conflicts	42%
Staking	36%
Other tree care issues	9%
Site characteristics	
Location type	47%
Land use	36%
Ground cover	27%
Soil characteristics	13%
Canopy cover	4%
Other site characteristics	13%
Other	13%

Table 4. Methods of recording tree location (n_{prog}=45)

Method	Percent
Street address	78%
GPS	42%
Site maps	31%
Tree tags	16%
Google maps	13%
Reference point	11%
Map cell number	4%
Other	18%