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# Urban Tree Mortality: a Primer on Demographic Approaches

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**Abstract**

Realizing the benefits of tree planting programs depends on tree survival. Projections of urban forest ecosystem services and cost-benefit analyses are sensitive to assumptions about tree mortality rates. Long-term mortality data are needed to improve the accuracy of these models and optimize the public investment in tree planting. With more accurate population projections, urban forest managers can also plan for cycles of tree planting, death, removal, and replacement to achieve canopy cover goals. Analytical tools from demography, such as life tables and mortality curves, could be used to improve our understanding of urban tree mortality and longevity. Demographic approaches have been widely used in forest ecology to quantify population dynamics and project future changes in wildland systems. This report is a primer on demographic concepts applied to urban trees, with terms and analytical methods adapted to the cultivated urban landscape. We include an overview of the uses of urban tree mortality rate data for research and management, a summary of lessons learned from ecological monitoring in other systems, and a discussion of opportunities for long-term urban forest monitoring by researchers and practitioners.

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Cover Photo: Trees in Center City Philadelphia, PA, as seen from the Philadelphia Museum of Art. Photo by Lara A. Roman, U.S. Forest Service.

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Chinese pistache (*Pistacia chinensis*) street trees in the West Oakland neighborhood of Oakland, CA. Photo by Lara A. Roman, U.S. Forest Service.

# Introduction

Urban forests are said to provide environmental, socioeconomic, and human health benefits, collectively referred to as ecosystem services (Dwyer et al. 1992, Nowak and Dwyer 2007, Tzoulas et al. 2007). Ambitious tree planting efforts in cities across the United States and around the world aim to expand the delivery of these benefits. Campaigns to plant a million trees have been a major force in urban sustainability initiatives (Young and McPherson 2013). The value of tree planting is justified, in part, by models that quantify and monetize tree benefits (McPherson et al. 2005, Nowak et al. 2008, U.S. Forest Service 2015b). The ecosystem services concept has become mainstreamed in urban forest management among municipal arborists and urban greening nonprofit organizations (Silvera Seamans 2013, Young 2013).

However, realizing the public value of urban forest programs depends on quantifiable locality-specific performance (Pataki et al. 2011), including tree survival (Roman et al. 2014b). Altering the mortality rate assumptions in urban forest ecosystem services models drastically affects projected tree values (e.g., 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 urban forest management that is designed to achieve canopy cover goals? Answers to these questions depend on the rates and processes of urban tree death. Thus, understanding urban tree mortality is critical to accurately modeling tree population changes over time and quantifying ecosystem services (Roman 2014).

This report is a primer on demographic concepts and analytical approaches relevant to the study of urban tree mortality. **Demography**—the statistical study of populations—is used to analyze mortality trends and project future changes in systems ranging from human societies to endangered wildlife communities and naturally occurring forests. The same concepts and calculations used by actuaries (to determine risk of death for issuing life insurance policies) and conservation biologists (to assess species extinction risk) can be used to study tree death in cities. Many terms have been used to describe tree mortality and longevity in the urban forestry literature (Roman and Scatena 2011, Roman et al. 2014b). To make meaningful comparisons across programs, it is important to use consistently defined terms.

This primer is intended to aid municipal arborists and urban greening organizations that undertake their own tree monitoring studies to evaluate mortality as a metric of program success (Roman et al. 2013). This report also serves to introduce demographic terms and concepts to urban forest researchers. We begin by briefly reviewing the importance of tree mortality to urban forest planning and population models. Throughout the report, we situate urban tree mortality within the larger body of literature on tree mortality in wildland systems. We define demographic terms in the context of the cultivated urban forest landscape. Specifically, we focus on the rates of tree death and associated lifespan metrics. We then outline an approach to studying urban tree mortality that relies on long-term monitoring to produce the data necessary for demographic analyses, and call for partnerships among researchers and practitioners to produce such long-term studies.

## Why Study Urban Tree Mortality Rates?

Ecosystem services projections and cost-benefit analyses have used a variety of mortality assumptions (Table 1). Urban tree mortality rates vary by size class and time since planting, among other factors (Koeser et al. 2014, Nowak et al. 2004, Roman et al. 2014a). Some models accounted for such differences (Table 1), but rates were occasionally stated without differentiation by tree size or age (McPherson et al. 1999). McPherson et al. (2008) did not reference

**Table 1.**—Mortality rate assumptions used in ecosystem services projections for urban tree planting. Most studies 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 tree cohorts, but one example (Nowak et al. 2002) assessed benefits based on an inventory of current tree stocks. Annual mortality rates used in each study are noted when available, and these assumed rates varied by years planted, age class, condition class, and tree locations.

| 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 to 5, and 2% annual mortality thereafter | |
| Low mortality | 35 | 17 (83) | Low-mortality scenario: 1% annual mortality years 1 to 5, and 0.5% annual mortality thereafter | |
| Modesto, CA | n/a | n/a | 1.4% annual mortality averaged across all age classes | McPherson et al. (1999)[a] |
| Sacramento, CA | | | | Hildebrandt and Sarkovich (1998)[b] |
| High mortality | 30 | 42 (58) | Residential shade trees; long-term mortality assumptions based on field records of survival during young tree establishment | |
| 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) |
| California urban forests | 15 | 25 (75) | 3% annual mortality years 1 to 5; 1% annual mortality thereafter | McPherson and Simpson (2001) |
| Chicago, IL | 30 | 35 (65) | Variable mortality for different tree site types (e.g., parks, residential yards, residential street trees) | McPherson (1994)[a] |
| Brooklyn, NY | n/a | n/a | Annual mortality for different condition classes: dead 100%; dying 50%; critical 13.08%; poor 8.86%; fair 3.32%; good–excellent 1.92% for 0–7.6 cm d.b.h.,1.46% for >7.6 cm d.b.h. | Nowak et al. (2002) [c, e] |
| New York City, NY | | | | Morani et al. (2011) [d, e] |
| 4% Avg. ann. mort. | 100 | 95.4 (4.6) | Trees planted over 10 years and grown out with assumed diameter growth rates | |
| 6% Avg. ann. mort. | 100 | 99.1 (0.9) | | |
| 8% Avg. ann. mort. | 100 | 99.8 (0.2) | Annual mortality for different d.b.h. 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 | |

[a] 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." This information was used to calculate carbon release from dead trees. Mortality rate assumptions in McPherson (1994) were also based on information provided by local managers.

[b] Hildebrandt and Sarkovich (1998) illustrated graphically that young trees have higher mortality rates than established trees, and provided information about observed young tree losses, but annual mortality rate values for different age classes were not provided. Note that "mortality" in this context also includes failure to plant because this is a shade tree giveaway program (Roman et al. 2014b).

[c] 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 1 year. The Brooklyn study is an example of the i-Tree Eco (formerly UFORE) model for assessing current ecosystem services from a plot-based inventory; other i-Tree Eco reports use these same methods.

[d] 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 percent, 6 percent, or 8 percent, with size-specific mortality rates proportional to those reported in the table.

[e] Morani et al. (2011), Nowak et al. (2002), and others using the i-Tree Eco model and prototype i-Tree Forecast (Nowak et al. 2013) referenced Nowak (1986) empirical data from mortality of maple *(Acer)* species street trees in Syracuse, NY, with rates differentiated by size and health condition classes.

Southern magnolia (*Magnolia grandiflora*) street trees in the West Oakland neighborhood of Oakland, CA. Photo by Lara A. Roman, U.S. Forest Service.

specific field data to support mortality scenarios, but other projections have used information supplied by local managers (Table 1). The mortality data cited in i-Tree Eco (formerly UFORE) and the prototype i-Tree Forecast models (Morani et al. 2011; Nowak et al. 2002, 2013) originated with a single study of maple (*Acer* spp.) street trees in Syracuse, NY (Nowak 1986). The Syracuse street tree mortality results, grouped by size class and condition rating, were then applied to different site types and land uses in other cities.

The mortality scenarios used in ecosystem services analyses and 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, or alternative mortality rate scenarios, 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 in New York, NY. As Morani et al. (2011: 1045) noted, however, uncertainty in mortality rates is "the main limit for the population projector."

Improving the accuracy of urban forest population projections is one reason to study tree mortality in cities. Urban tree mortality research can also help managers to identify leading risk factors for death, and then target program modifications that could enhance survival (Roman et al. 2014b, Vogt et al. 2015). Municipal foresters and local urban greening organizations have also expressed interest in tree mortality and monitoring as a way of quantifying success (Roman et al. 2013). Some practitioners have used monitoring as a proactive approach toward tree maintenance and management. For example, municipalities have undertaken systematic repeated inventories as part of pruning cycles and tree risk management, and planting initiatives have used monitoring data to give feedback to those responsible for tree care, such as residents and contractors (Roman et al. 2013). From a basic research perspective, urban tree mortality is central to urban forest population dynamics, and understanding mortality is essential to assessing temporal patterns in urban forest structure. Similarly, in natural forest ecosystems, tree monitoring and mortality research are crucial to understanding long-term dynamics, which are integral to sustainable management and model accuracy (Lines et al. 2010).

Before proceeding further, we must note that although urban forests can be broadly defined to include all trees in cities and urbanized areas (Konijnendijk et al. 2006, Piana and Troxel 2014), our report focuses on intentionally planted trees in street and residential lawn settings. Urban tree mortality rates differ by planting site type and land use (Koeser et al. 2014, Lawrence et al. 2012, Nowak et al. 2004). City trees in remnant patches or afforested parks of native and naturalized species, or forested areas on vacant lots and private lands (Zipperer et al. 1997), 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 report is most relevant to heavily altered human-dominated portions of the urban forest: street and lawn trees.

As we draw on demographic concepts used in natural, wildland forests, we must bear in mind essential differences in the life cycle for wildland and urban trees. Wildland forests have natural processes of seed dispersal and germination; large amounts of seeds are 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 (Das et al. 2007, Pedersen 1998, Waring 1987), eventually succumbing to death through contributing factors such as wind, insects, or pathogens (Franklin et al. 1987, Harcombe and Marks 1983). 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 (and remnant forest patches in urban landscapes), urban street and yard trees are typically produced by nurseries and planted as saplings or whips in sidewalk cut-outs, planting strips, or manicured lawns. Urban environments pose many challenges for trees, such as compacted and contaminated soil (Craul 1999), construction (Hauer et al. 1994), and vandalism (Nowak et al. 1990), but urban trees may also have advantages not present for wildland trees, including fertilizer, irrigation, 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 and reduce liability from falling limbs or infrastructure conflicts (Harris et al. 2004); this preemptive action kills the tree before the mortality spiral is completed. Additionally, some healthy city trees are removed due to human aesthetic 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. For example, although demographic methods are traditionally applied to a single species, we follow the convention of urban foresters, whereby "urban tree population" refers to all trees within a given area or from a particular planting program (Roman et al. 2014a). However, we acknowledge that urban tree populations are human-constructed systems with multiple species, and that species functional groups, such as drought tolerance, can be relevant to mortality analyses (Roman et al. 2014b). With sufficient sample size, separate mortality models can be developed for different species.

This primer is intentionally focused narrowly on mortality rates and associated lifespan metrics, so we do not address statistical techniques to assess factors related to mortality. For readers interested in such research, we point to several recent studies that have used multivariate analyses to investigate urban tree mortality (Ko et al. 2015; Koeser et al. 2014; Lawrence et al. 2012; Roman et al. 2014a, 2014b; Staudhammer et al. 2011; Vogt et al. 2015). Such techniques are widespread in the ecological and epidemiological literatures and can aid in interpreting the most relevant factors for mortality in a particular tree planting initiative or geographic area, as well as provide evidence for population projections (Table 1). Similarly, we do not cover spatial aspects of urban tree mortality, although spatial patterns can be an important component of population dynamics in wildland forests (e.g., Das et al. 2008) and probably urban forests as well. The techniques discussed in this report for quantifying mortality rates are a necessary first step for managers and researchers to determine what levels of loss are typical for urban trees as they age and grow.

London planetree (*Platanus* × *acerifolia*) street trees in Philadelphia, PA, including a tree that fell during an ice storm and was removed. Photo by Lara A. Roman, U.S. Forest Service.

# Age-based Life Tables, Survivorship Curves, and Mortality Curves

In discussing tree mortality, urban foresters can use terms established in population biology and demography: annual mortality rates, annual survival rates, life tables, survivorship curves, mean life expectancy, and population half-life. We have defined these and other terms in the glossary. 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 in this primer, as both are relevant to urban forest systems.

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, age-based life tables are appropriate for projects that monitor planting **cohorts**—trees planted around the same time (e.g., trees planted in the same year). Cohort survival data can also be depicted by using survivorship and mortality curves (Box 1).

As an example of an age-based life table for urban trees, we present survival data from Sacramento County, CA. These data are adapted from a study that monitored a cohort of shade trees that were distributed by the Sacramento Tree Foundation and the Sacramento Municipal Utility District in 2007 (Roman et al. 2014b). These trees were planted in residential lawns. Although the "clock" in an age-based life table typically begins at birth, in the context of urban trees, the "clock" begins at planting. Notation and formulas for the age-based life table, as applied to an urban tree planting cohort, are summarized in Table 2, and exemplified in the Sacramento cohort in Table 3. The life table presented here considers only trees that were planted and omits trees that were distributed to residents but never planted (Roman et al. 2014b). The time interval between age classes in this example, and throughout the rest of our discussion of age-based life tables, is set to 1 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 periods, which requires slight changes to the equations in Table 2 (Carey 1993). The "clock" could also begin at a different time to suit situations encountered by urban forest managers (e.g., begin counting time at the end of a 2-year contractor guarantee).

**Box 1: Classic Survivorship and Mortality Curve Shapes**

A graph of $ln\ (l_x)$ vs. $x$ is called a **survivorship curve**. For age-based cohort mortality, survivorship $l_x$ is the proportion surviving to age $x$. In the survivorship curve, $ln\ (l_x)$ is the natural logarithm of survivorship; the natural log has base $e$, where $e$ is a constant approximately equal to 2.718. Survivorship curves are conventionally ln transformed to depict more clearly changes in the proportion of individuals surviving over time. The shape of the survivorship curve depends on how the mortality rate $q_x$ 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. The sigmoid survivorship curve and U-shaped mortality curve represent high mortality for both old and young individuals, with low mortality rates in between.

Mortality curves for nonurban trees are typically drawn by using size-based (not age-based) mortality rates, and the shape of the curve has implications for management and predictive modeling (more on size-based mortality curves beginning on p. 12). 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.

Note that throughout this primer, $x$ means time, with the exception of this graph, where $x$ can also mean age class for mortality curves.

Survivorship and mortality curves depicted here are adapted from Harcombe (1987).

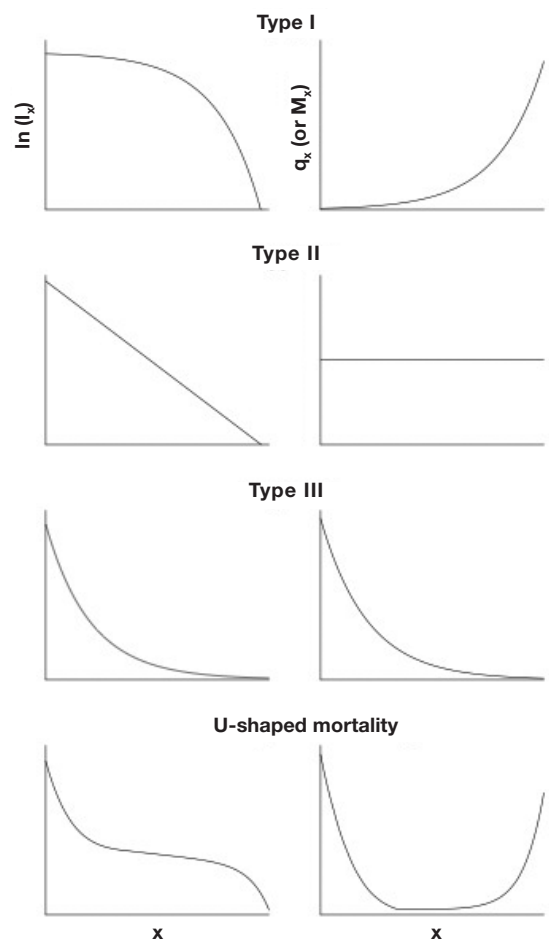

**Table 2.**—Notation and formulas for the age-based urban tree life table. Notation mostly follows Carey (1993), with censoring terms from Klein and Moeschberger (1997).

| 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 $x+1$ |
| $D_x$ | number of deaths in the interval $x$ to $x+1$ |
| $W_x$ | number of individuals right 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, and that causes of mortality and censoring are independent<br>$Y_x = K_x - (W_x / 2)$<br>(eqn. 1, after Klein and Moeschberger 1997: 138) |
| $q_x$ | proportion dying from $x$ to $x+1$, annual mortality rate<br>when there is no censoring:<br>$q_x = D_x / K_x$ (eqn. 2)<br>to compensate for censoring:<br>$q_x = D_x / Y_x$ (eqn. 3) |
| $p_x$ | proportion surviving from $x$ to $x+1$, annual survival rate<br>$p_x = 1 - q_x$ (eqn. 4) |
| $l_x$ | proportion of the cohort surviving from planting to age $x$, commonly called survivorship to age $x$; $l_0 = 1$ (i.e., 100 percent) by definition; eqns. 5 and 6 are equivalent when there is no censoring<br>when there is no censoring:<br>$l_x = K_x / K_0$ (eqn. 5)<br>to compensate for censoring:<br>$l_x = (l_{x-1})(p_{x-1})$<br>$= \prod_{i=1}^{x} (1-(D_{i-1}/Y_{i-1}))$<br>(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<br>$L_x = (l_x + l_{x+1})/2$<br>(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 possible year of life $w$:<br>$T_x = L_x + L_{x+1} + L_{x+2} + \cdots + L_w$<br>$= \sum_{i=x}^{w} (L_i)$<br>(eqn. 8, after Carey 1993: eqn. 2-4) |
| $e_x$ | life expectancy at age $x$:<br>$e_x = T_x / l_x$<br>(eqn. 9, after Carey 1993: eqn. 2-5) |

Our example from Sacramento (Table 3) tracks a cohort for the first 5 years, and illustrates the key components of an age-based life table. Note that the first three columns ($K_x, D_x, W_x$) 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 $x + 1$ (i.e., 1 year).

Note that survivability, a commonly used term in urban forestry, does not have a standard definition in demography and population biology. Indeed, survivability is also not generally defined in the urban forestry literature; survivability could mean annual survival, survivorship, or something else. Stating clear definitions—with equations and time periods specified—for terms such as annual survival, survivability, and survivorship (Roman et al. 2014b) is essential to comparing survival outcomes across cities and programs.

## Censored Data

A complication in the Sacramento example (Table 3), and in demographic studies in general, is that the survival outcome of all individuals could not be determined. We have only partial field data for a few of the Sacramento trees, in situations where we could not secure permission to enter the backyard every year. These incomplete observations are referred to as **censored data**. Specifically, some 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. (In fact, all trees that remain alive at the study's conclusion are right censored, because we do not know when they will die.) For example, in medical research, when scientists permanently lose contact with patients during the course of the study, their information is considered right censored. There are many different approaches to censored data in survival analysis (Klein and Moeschberger 1997). We used a simple method to compensate for right censoring ($Y_x$ in Table 2) that depended on two assumptions: the causes of mortality and censoring were independent, and censoring times (i.e., times when individuals dropped from the study) were uniformly distributed.

**Table 3.**—Life table example from field observations. Age-based life table from a cohort of residential lawn trees in Sacramento County, CA (Roman et al. 2014b). The time of planting is $x = 0$. See Table 2 for notation and definitions. Note that 0.122 is equivalent to 12.2 percent.

| 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 | 370 | 45 | 0 | 370 | 0.122 | 0.878 | 1.000 |
| 2008 | 1 | 325 | 20 | 2 | 324 | 0.062 | 0.938 | 0.878 |
| 2009 | 2 | 303 | 18 | 5 | 300.5 | 0.060 | 0.940 | 0.824 |
| 2010 | 3 | 280 | 13 | 3 | 278.5 | 0.047 | 0.953 | 0.775 |
| 2011 | 4 | 264 | 9 | 6 | 261 | 0.034 | 0.966 | 0.739 |
| 2012 | 5 | 249 | n/a | n/a | n/a | n/a | n/a | 0.713 |

There are other forms of censoring, however. In urban forestry studies like the Sacramento example with private properties and incomplete observations, the analytical methods must ensure that censored data do not bias calculated vital rates. Interval censoring was present for every dead shade tree in the Sacramento 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. In other words, as long as death is known to occur between one summer field season and the next, the age-based life table works well. But interval censoring becomes problematic for the Sacramento life table when the interval length was longer than 1 year. For example, consider a tree seen alive in 2009, of unknown status in 2010 and 2011, and observed missing in 2012. From the available information, we do not know which year the tree died. Our cohort life table (Table 3) had three such cases (<1 percent of the 370 trees total). To retain these three cases in the life table, we assumed that those death events occurred at the midpoint of the longer interval. This is the simplest way to deal with interval-censored data; however, using the midpoint can lead to biased estimates (Giolo et al. 2009).

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 than the life table can provide, we used the 'interval' package in the free statistical software R (Fay and Shaw 2010, R Development Core Team 2014), with specific planting dates and observation dates included (Roman et al. 2014b) (Fig. 1). In addition to dealing with interval censoring, this survivorship curve also accounts for the precise number of days between planting date and the first field observation for every tree, as well as the exact length of time between later field observations (e.g., when summer field work was not exactly 365 days apart). Trees were planted in the Sacramento program between January and December 2007, with annual mortality observations and field work each summer from 2008 through 2012. The first "year" in the cohort life table (Table 3) is therefore a range of about 6 to 21 months (Roman et al. 2014b). Thus there is a slightly different 5-year survivorship calculated from the cohort life table (Table 3, 71.3 percent, or stated equivalently, 0.713) and the survivorship curve (Figure 1a, 70.9 percent, or equivalently, 0.709). The survivorship estimate from 'interval' is more precise because exact dates were included and censored information was appropriately incorporated. Nonetheless, the cohort life table is a simple approach that both researchers and practitioners in urban forestry may find useful to summarize cohort mortality data without specialized statistical analysis, or when exact planting dates are unavailable. Values in the age-based life table can be calculated in common spreadsheet programs (e.g., Microsoft® Excel).

To further demonstrate the utility of assessing mortality data with more sophisticated survival analysis, we have also presented a graph comparing survival outcomes on stable versus unstable properties in Sacramento (Roman et al. 2014b) (Fig. 1b). Stable properties are owner-occupied,

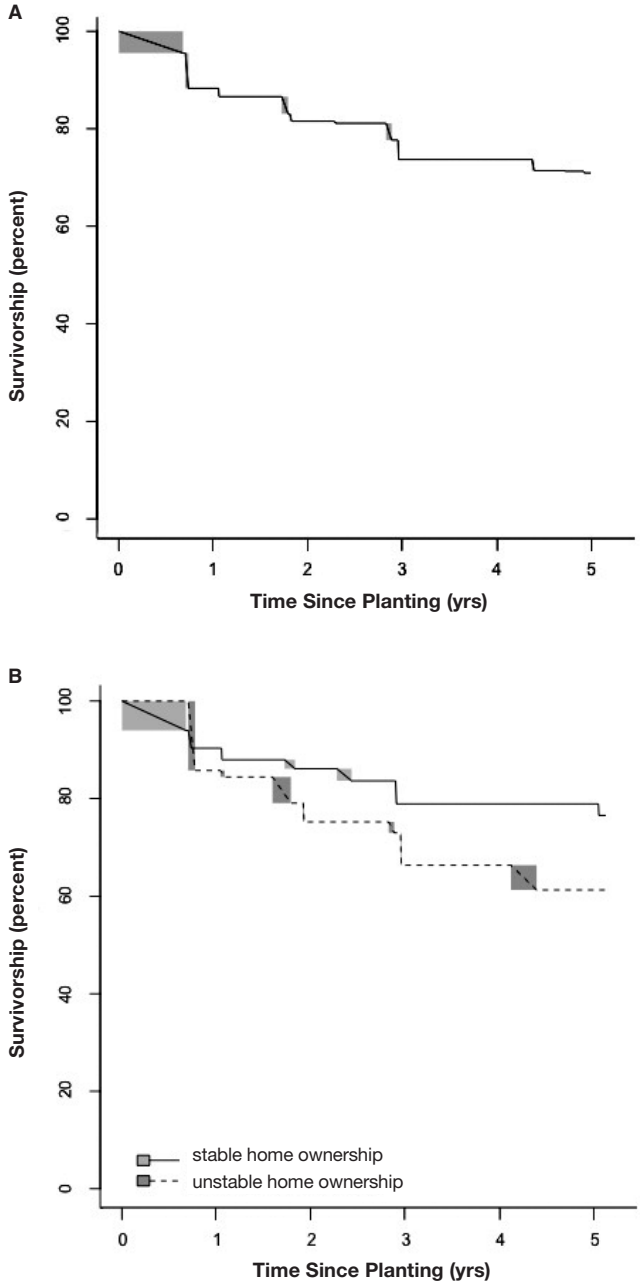


**Figure 1.**—Survivorship curves for trees in single-family residential yards (n=370), Sacramento County, CA. Survivorship was assessed from Kaplan-Meier survival analysis with Turnbull (1976) estimator for censored observations (Fay and Shaw 2010). The gray rectangles indicate the range of possible values given censoring. Survivorship at 5 years equals 70.9 percent for all trees (A). The most important variable for 5-year survival was stable home ownership (B); survivorship at 5 years was 61.3 percent for unstable properties and 76.6 percent for stable properties. A weighted logrank test (Fay and Shaw 2010) shows significant difference in the survival curves ($p < 0.01$). From Roman et al. (2014b).

and had the same resident throughout the study, whereas unstable properties had foreclosures, home sales, were renter occupied, or a combination thereof. Based on multivariate analysis with a data mining approach (specifically, conditional inference trees), stable home ownership was found to be the best predictor of shade tree establishment success. Trees on properties with stable home ownership were significantly more likely to survive, and more likely to have a higher maintenance rating. Residential yard tree planting programs therefore must be cognizant of resident turnover and associated changes in stewardship. The Sacramento Tree Foundation has used these and other findings from that 5-year survival study to guide program modifications (Roman et al. 2014b).

## Calculating Annual Mortality from Survivorship

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

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

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

where $K_0$ and $K_x$ are the population sizes at the beginning and end of time interval $x$, respectively. As with the cohort life table, $x$ is years. The fraction $K_x/K_0$ is 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. This equation is especially useful when a cohort is not visited annually. For example, Nowak et al. (1990) reported 66 percent survivorship 2 years after planting for street trees in Oakland, CA. Annual survival was therefore 81.2 percent, and annual mortality 19.8 percent, with the assumption of constant mortality: survivorship at 2 years = 0.66 = $0.812^2$. Annual survival was estimated from survivorship data reported in other street tree studies in a similar fashion in Roman and Scatena (2011) (Table 4). Even when a cohort is observed annually, as with the Sacramento example, eqn. 10 can be used to report annual survival over the entire study period. Based on 70.9 percent survivorship at 5 years (Fig. 1), annual survival of yard trees in Sacramento was 93.4 percent, with 6.6 percent annual mortality (Roman et al. 2014b).

**Table 4.**—Estimated annual street tree survival rates, mean life expectancy, and population half-life. Adapted from Roman and 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). Note that 0.035 is equivalent to 3.5 percent.

| Source | 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, PA, field survey, survivorship 2–10 years after planting | 0.034–0.045 (0.955–0.966) | 22–29 | 15–20 |

# Lifespan Metrics

Information about the typical lifespan for urban trees can help managers anticipate future tree losses. Although the "average lifespan" has been frequently discussed in urban forestry (Moll 1989, Skiera and Moll 1992), the term is not clearly defined, and is rarely based on field data. The idea that street trees have an average lifespan of 7 years comes from a questionnaire sent to cities across the United States in which municipal foresters reported typical tree lifespans (Moll 1989). Another report claimed that downtown trees have an average lifespan of 13 years (Skiera and Moll 1992). Nowak et al. (2004) estimated a 15-year average lifespan by using observed mortality rates for different tree size classes and land uses in Baltimore, MD, although the exact formula used to calculate lifespan was not specified. Demographers have different standard terms for lifespan that can be used in urban forestry: the mean life expectancy and population half-life.

With the constant mortality assumption (Type II), typical lifespan can be easily quantified. The mean life expectancy is the average age at death across the population. The mean life expectancy $e_0$ from the time of planting, $x = 0$, is:

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

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

As a clarification, the notation $e_x$ (with the subscript) is life expectancy from age $x$, whereas $e$ (without a subscript) is a constant approximately equal to 2.718 and is the base for the natural logarithm. We use $e_x$ following conventional notation in demography (Carey 1993, Seber 1982).

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

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

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

For street trees, a meta-analysis (Roman and Scatena 2011) estimated annual survival rates and lifespan metrics (Table 4). Survivorship data from 11 previous studies were pooled in a linear regression analysis, with the assumption of constant mortality, to estimate annual survival. Mean life expectancy was estimated at 19–28 years, and population half-life 13–20 years. The 7- or 13-year average lifespans of street trees suggested by Moll (1989) and Skiera and Moll (1992) are therefore not supported by the available field data. 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), and this technique can be used to estimate tree lifespan metrics for other planting programs.

However, mortality may not be constant over time, which complicates the calculation of mean life expectancy. Urban trees are said to have an establishment period, the first several years after planting when trees are more vulnerable to mortality (Miller and Miller 1991, Richards 1979). When mortality is not constant, mean life expectancy can still be determined, but the cohort life table must be completed to the oldest age classes. To determine life expectancy, several additional columns are added to the cohort life table: $L_x$, $T_x$, $e_x$ (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, $x + e_x$. For example, actuaries calculate life

expectancy for people of various ages and characteristics to determine life insurance premiums. Classic examples of calculating $e_x$ are given in Carey (1993) and Seber (1982), which focused 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 earlier by using the constant mortality assumption (eqn. 11). Mean life expectancy at birth can be thought of as the average age of death. Calculating $e_0$ by using eqn. 9 (Table 2) when annual mortality is not constant requires having life table data for the cohort until the last individual has died.

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. For urban trees, however, age-based life tables and survivorship curves are still relevant. Built into the mortality assumptions of many urban forestry ecosystem services projections (Table 1) are estimates of annual mortality rates for planting cohorts. As an example, we used the assumed low- and high-mortality scenarios in the Million Trees initiative for Los Angeles, CA (McPherson et al. 2008) to construct age-based life tables (Table 5) and survivorship curves (Fig. 2). The environmental benefits for the Los Angeles example were projected for 35 years after planting. Most tree losses were assumed to occur in the first 5 years after planting (McPherson et al. 2008). The

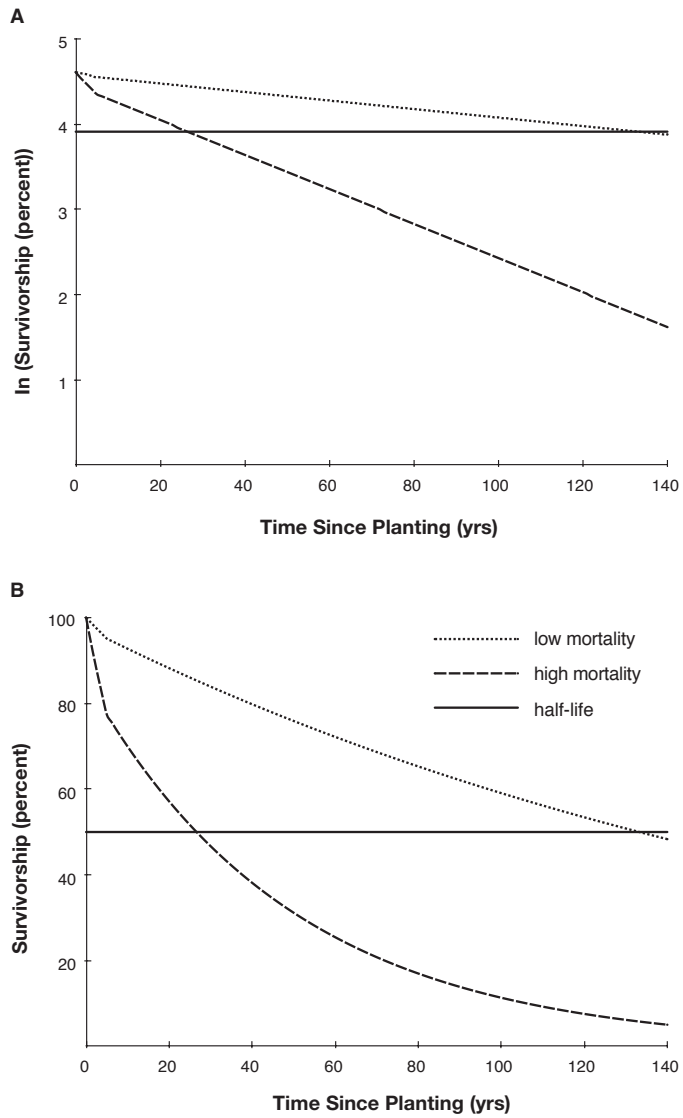


**Figure 2.**—Assumed survivorship curves used in ecosystem services projections for the Million Trees LA program in Los Angeles, CA, with (A) and without (B) natural log (ln) transformation. Adapted from McPherson et al. (2008). See Table 5 for annual survival rates.

**Table 5.**—Life table example from a population projection. Age-based life table for annual mortality rate assumptions used in cost-benefit analysis for the Million Trees LA program in Los Angeles, CA. Adapted from McPherson et al. (2008). The terms survivorship, annual survival rate, and annual mortality rate are defined in Table 2 (eqns. 2, 4, 5). For years 5 and beyond, annual mortality remains constant at either 2 percent (high-mortality scenario) or 0.5 percent (low-mortality scenario). These assumptions are compared in the text to empirical observations for young tree mortality in Los Angeles (McPherson 2014).

| | 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 |

low-mortality scenario was projected to yield $1.95 billion in benefits, whereas the high-mortality scenario would reduce benefits by 32 percent, to $1.33 billion (McPherson et al. 2008). These mortality assumptions were checked by monitoring trees planted through the Los Angeles program; annual mortality rates during the establishment phase were 4.4 percent for street trees, 3.1 percent for yard trees, and 4.6 percent for park trees (McPherson 2014). These empirical mortality data, combined with growth observations, were used to revise simulations for ecosystem services (McPherson 2014). The observed mortality during establishment is closer to the high-mortality scenario in the original projections (McPherson et al. 2008), although lower than mortality for Sacramento yard trees (Roman et al. 2014b).

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 LA projected benefits (McPherson et al. 2008) can be calculated by using the assumed annual survival rates (Table 5). For example, in the high-mortality scenario, determine the half-life by solving for $x$ :

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

In this situation, we are calculating the time at which survivorship $l_x$ equals 0.5 (or 50 percent) (eqn. 12). With the high-mortality scenario, half-life would be 27 years, based on 95 percent annual survival for the first 5 years followed by 98 percent survival thereafter. With the low-mortality scenario, half-life would be 133 years (using 0.99 and 0.995 instead of 0.95 and 0.98, respectively).

These half-life values are illustrated more intuitively in Figure 2. Both of these half-life estimates for Los Angeles are considerably higher than the half-life estimated in the street tree meta-analysis (Roman and Scatena 2011) (Table 4). A half-life of 133 years seems unrealistic, and indeed, as previously mentioned, the mortality rates observed during establishment in Los Angeles are closer to the high-mortality scenario (McPherson et al. 2008). For comparison, for another large-scale planting initiative in New York City, 73.8 percent of street trees remained alive 8 to 9 years after planting (Lu et al. 2010). 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 their canopy cover goals, it is essential to have accurate mortality rates (Roman 2014), as well as growth rates and allometric equations (McHale et al. 2009, McPherson and Peper 2012). The 7-year average lifespan figure appears overly pessimistic, and the low-mortality scenario in the Los Angeles population projections appears overly optimistic. For reasonable forecasts of planting program performance, we need realistic projections based on field observations, rather than overly optimistic or pessimistic speculations.

Note that the lifespan metrics described in this section are relevant only to age-based mortality data, not the size-based mortality data discussed in the next section.

## Size-based Mortality Curves

In contrast to age-based mortality curves, an alternative is to display mortality curves by diameter at breast height (d.b.h.) size or size classes (Harcombe 1987). Forest ecologists have observed Type III or U-shaped size-based tree mortality rate curves (Box 1), with different mortality curve shapes from various forest systems and species (e.g., Coomes and Allen 2007, Lorimer et al. 2001, Metcalf et al. 2009). In a recent large study of forests across the eastern United States with records of more than 430,000 stems (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 overstory or canopy trees, typically 1 to 3 percent or even less (e.g., Condit et al. 1995, Harcombe and Marks 1983, Lorimer et al. 2001), absent catastrophic disturbances (Lugo and Scatena 1996).

Urban tree mortality distributions using size classes can be created for repeated inventories, such as a repeated census of street trees in a particular neighborhood, or repeated monitoring of plots. We used the annual mortality rates reported for randomly located plots in Baltimore by Nowak et al. (2004) to construct a mortality curve (Fig. 3). Annual mortality was highest for small trees 0–7.6 cm (0–3 in) d.b.h. (9.0 percent) and lowest for midsized trees 30.6–45.7 cm (12.1–18 in) d.b.h. (0.5 percent), then rose again for larger trees (>45.7 cm or 18 in), with some fluctuations

(1.8–3.3 percent). This pattern corresponds to a U-shaped curve. Street trees in Oakland, CA, had a Type III mortality distribution (Roman et al. 2014b) (Fig. 4), in which mortality was again higher in the smallest size class (5.6 percent) than in other size classes (0.0–1.6 percent).

Even though the general mortality curve shapes are similar to trees in wildland environments, with both having about 1 percent mortality for midsized trees, there are essential differences in the mortality process. As previously mentioned, large urban trees are sometimes cut down and removed before they fully succumb to pathogens, injuries, and stresses. These hazard removals may increase mortality rates for urban trees with large d.b.h., and cause a more distinct upswing at the tail of the U-shaped curve, in comparison to wildland trees. On the other end of the mortality curve, very small trees in urban street and lawn settings are generally newly planted, with fairly high mortality rates during the establishment phase (Miller and Miller 1991, Richards 1979). 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 minimum size that would exclude newly planted or small-stature urban trees (e.g., limiting forest inventories to trees ≥12.7 cm or 5 in). Other studies include young seedlings (e.g., 3-year-old seedlings in Cleavitt et al. 2011) that would commonly be in the nursery production stage for urban forests. Classes of d.b.h. 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. For some plot-based urban tree inventories (e.g., i-Tree Eco methods; U.S. Forest Service 2015b), woody stems are included if their d.b.h. is at least 2.54 cm (1 in). Therefore some plants considered shrubs or saplings in classic forest ecology studies are included as trees in urban forest research. Indeed, even the 2.54 cm (1 in) threshold excludes some small newly planted

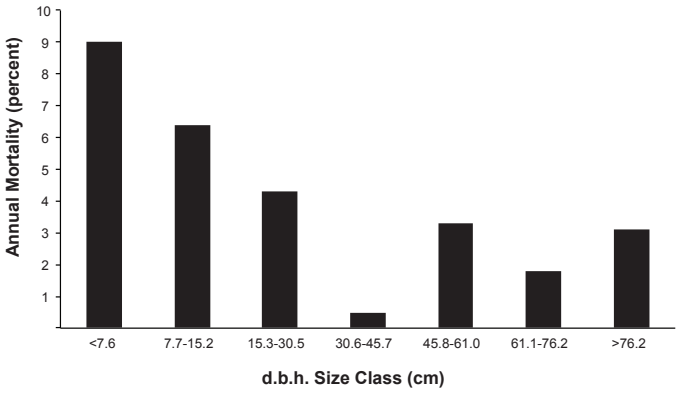


**Figure 3.**—Urban tree mortality rates for Baltimore, MD, based on size class. Trees (n=1,396) were observed over a 2-year study period in plots located randomly throughout Baltimore, and included many species, land uses, and site types. Adapted from Nowak et al. (2004).

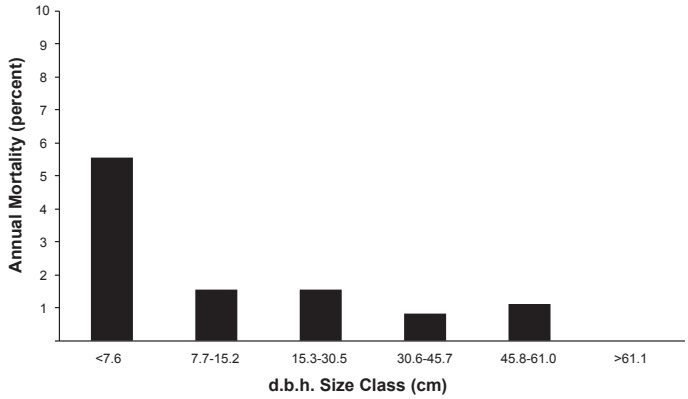


**Figure 4.**—Street tree mortality rates for Oakland, CA. Trees (n=940) were observed over a 5-year study period, and included many species and land uses. Planting site types were sidewalk cut-outs, planting strips, and medians. From Roman et al. (2014a).

Dead street tree in the West Oakland neighborhood of Oakland, CA. Photo by Lara A. Roman, U.S. Forest Service.

Red maple (*Acer rubrum*) yard tree from the Sacramento Shade program in Sacramento County, CA. Photo by Lara A. Roman, U.S. Forest Service.

trees, such as those included in the Sacramento (Roman et al. 2014b) and Oakland (Roman et al. 2014a) studies. Thus these saplings are particularly important to understand population dynamics in the urban setting. Size-based mortality curves and life tables will aid our understanding of urban tree mortality trends, but as with other aspects of wildland forest demography, we must adapt these tools to urban systems, including explicitly defining "tree" for monitoring purposes.

## Tracking Population Dynamics

Urban forests change over time through inputs to and outputs from the system. In human and animal populations, those fluxes are birth, death, immigration, and emigration. In the urban forest, we have losses from mortality and removal, and inputs from planting and natural regeneration of seedlings. Across all site types in the urban forest, Nowak (2012) concluded that only one out of three trees in cities is planted. Regeneration is thus critical to maintaining canopy in remnant forest patches on public and private property, as well as afforested vacant lots (Zipperer et al. 1997). In the heavily managed street tree environment, however, natural regeneration is negligible, so the main source of new trees is planting.

Tracking the inputs (from planting) and the losses (from death and removal) in street tree populations aids our understanding of the impact of planting initiatives. An example of such fluxes comes from 5 years of annual monitoring in Oakland (Roman et al. 2014a). This study assessed net changes in street tree population counts, in relation to annual planting and mortality. The classic demographic balancing equation, which quantifies inputs and losses of individuals from a population, was adapted to the street tree context (Roman et al. 2014a) (Fig. 5). The West Oakland neighborhood has been the focus of recent planting efforts by both the City of Oakland and a local nonprofit organization, Urban Releaf. These planting programs seek to provide socioeconomic benefits and address environmental injustices in an underserved community.

After an initial neighborhood street tree inventory in 2006, an annual census was conducted to observe tree mortality, removals, and new plantings. There was an overall population increase during the 5-year study period: 995 live street trees in 2006, and 1,166 in 2011, for an increase of 17 percent. The annual mortality rate across all size classes was 3.7 percent, which is within the range of typical street tree mortality rates from the literature review discussed earlier (Roman and Scatena 2011). Thus the mortality rates are within the "normal" range and the population is on the rise. However, mortality of small, young trees suppressed population growth. Roughly half of the 2006 trees were small, with trunks 7.6 cm (3 in) in diameter or less (Roman et al. 2014b). As previously mentioned, annual mortality in that smallest size class was 5.6 percent, about four times the rate for all the other size classes (Fig. 5). Additionally, in a complementary analysis of planting cohorts, survival of new young trees added to the system was fairly low, with only 83 percent of new trees surviving for 2 years. Although the age-based and size-based mortality rates organize the data differently, they both lead to the same conclusion: most of the tree losses came from recently planted, small trees. The planting campaigns in this neighborhood were barely outpacing young tree deaths, and could have had a larger impact if young tree survival had been enhanced. These

findings support the assertions in Richards (1979) that the death of young street trees drives population cycles.

Monitoring studies to track fluxes in urban tree populations, such as the Oakland example, can provide critically needed empirical evidence for models and managers. Miller and Marano (1984) and Bartch et al. (1985) presented street tree simulations, using tree inventory data combined with user-defined planting, growth, mortality, and removal rates. These models were designed to help meet management objectives related to costs and desired benefits, enabling urban forest managers to project the impacts of various planting and removal actions. Local managers continue to conduct monitoring to assess population dynamics. In a survey about practitioner-driven monitoring (Roman et al. 2013), one respondent summarized the motivation for data collection as follows: "The sense that we were losing trees as fast as they were being planted made [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." This statement essentially calls for analyzing demographic change in the urban tree population. The adapted balancing equation (Roman et al. 2014b) (Fig. 5) provides a conceptual and computational approach for applying demographic methods to street tree population simulations.

## 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 can provide mortality and growth rates to build life tables, survivorship and mortality curves, and balancing equations, as well as more advanced analyses such as matrix models (Caswell 2001). 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, institutional structures, and program operations. However, only data tracking the fate of individual trees are suitable for the analyses discussed here.

Although urban forest inventory systems have been developed to provide managers with quantitative data on forest composition and structure (McPherson et al. 2005, Nowak et al. 2008), 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 finding plots and individual trees during subsequent visits (Roman et al. 2013), and determining the appropriate

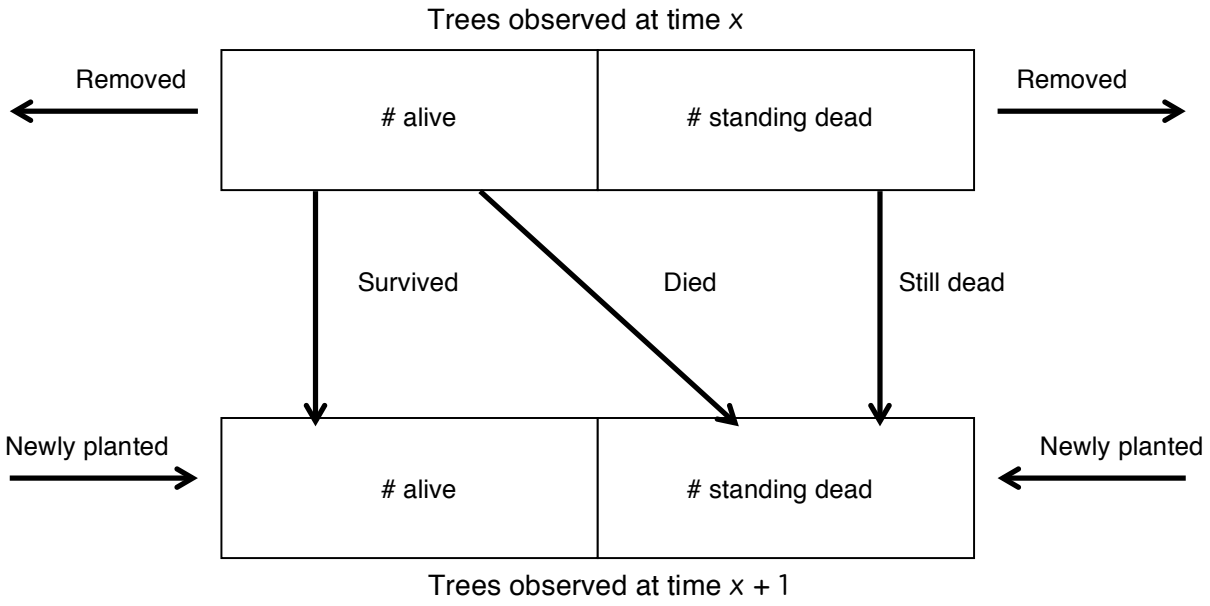


**Figure 5.**—Diagram illustrating street tree balancing equations for live and standing dead street trees (after Roman et al. 2014a). Arrows indicate transitions between pools of live and standing dead trees from time *x* to time *x*+1. The bottom right arrow indicates newly planted trees that were observed dead during the summer field work (but were presumably alive when planted).

London planetree (*Platanus* × *acerifolia*) trees at the Rodin Museum in Philadelphia, PA. Photo by Lara A. Roman, U.S. Forest Service.

observation intervals. As previously mentioned, some local urban forestry organizations in the United States already gather mortality data (Roman et al. 2013). These practitioner-driven studies rarely result in peer-reviewed journal articles (although two notable exceptions are Boyce 2010 and Lu et al. 2010), but sometimes lead to internal program reports.

To promote data-sharing among professionals and researchers, and to advance monitoring efforts already underway, standardized protocols for urban tree monitoring have been developed (Urban Tree Growth and Longevity Working Group 2015). Standardization can enable comparisons across and within cities, and avoid duplicated efforts to develop monitoring methods. For example, a protocol for tracking recently planted street trees (Vogt and Fischer 2014) was recently applied across several United States cities, in collaboration with local nonprofit organizations. Such partnerships can provide urban forestry professionals with improved mortality information to evaluate the success of planting and management programs, while expanding the long-term datasets available to researchers. Participatory research approaches (McKinley et al. 2013, Minkler and Wallerstein 2008) will be critical to these research-practice partnerships for urban forests (Campbell et al. 2014).

## 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, 2010). Urban forest practitioners who collect monitoring data also stressed the importance of clear objectives and uses of the data (Roman et al. 2013).

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 U.S. Forest Service and Long-Term Ecological Research (LTER) sites sponsored by the National Science Foundation. The FIA program serves as a census for forest ecosystems in the United States (Smith 2002, U.S. Forest Service 2015a), with recent integration of the Forest Health Monitoring program (U.S. Forest Service, Forest Health Monitoring 2015) and annual field measurements (McRoberts et al. 2005) to generate longitudinal data. Many studies of tree mortality have used FIA data (e.g., Lines et al. 2010, Woodall et al. 2005). 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 5 years (Condit 1995, Smithsonian Tropical Research Institute 2015). Although these programs focus primarily on nonurban forests, the methods and analytical tools have been adapted to urban systems, such as the emerging Urban FIA program (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 (Cary Institute of Ecosystem Studies 2014) and

Phoenix, AZ (Arizona State University 2015). In Baltimore, repeated i-Tree plot data have been collected (Nowak et al. 2004), and in Phoenix, annual tree surveys were conducted across an urban to rural gradient (Martin and Stutz 2013). To gather comprehensive longitudinal data on urban trees, it is essential that urban foresters and urban ecologists coordinate efforts, with clear research questions and strong partnerships, learning from the experiences of forest ecologists working in long-term monitoring programs.

The practitioner-driven monitoring initiatives discussed earlier can complement these national programs for plot-based monitoring. Long-term data collected by practitioners have the potential to produce higher volumes of information, but because local organizations sometimes rely on volunteers and citizen scientists, there are concerns about data quality (Roman et al. 2013). Although ecological data collected by volunteers have the potential for errors, those errors can be minimized with effective training and data validation procedures (Bonter and Cooper 2012, Gardiner et al. 2012). Indeed, ecological monitoring programs with citizen science have become increasingly popular (Dickinson et al. 2012, Silvertown 2009). Monitoring programs incorporating local practitioners and volunteers can provide higher volumes of data across broad geographic regions, while expanding opportunities for public engagement in research and fostering environmental stewardship (Silva and Krasny 2014).

## Looking Ahead: Opportunities for Future Research

To promote research-practice dialogue and data sharing regarding urban tree mortality, researchers are collaborating with municipal foresters and nonprofit organizations through the Urban Tree Growth and Longevity Working Group (Leibowitz 2012, Scharenbroch et al. 2014). Many municipal foresters, urban greening nonprofit organizations, and states are interested in collecting long-term monitoring data. Additionally, research articles on urban tree mortality have become more frequent in the past few years. This confluence of interests from researchers and professionals can generate the observational data called for in earlier population projections (McPherson et al. 1999, Morani et al. 2011). As researchers and communities gather more long-term urban tree monitoring data, we will be better equipped to understand the patterns and processes of urban tree death.

However, there are challenges 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 (Lindenmayer and Likens 2010). To facilitate permanent plots, the initial inventory or tree planting information should include detailed site maps, geospatial coordinates, site photos, or tagged tree identification numbers. Such practical considerations are often overlooked with static snapshot urban tree inventories (Roman et al. 2013). Additionally, demographic analyses and mortality studies of wildland and plantation forests are traditionally applied to a single species, in systems with natural reproduction. Our examples of urban street and lawn trees lump many species together, and are open systems with human-driven planting and removal. Although we can adapt demographic tools, we must be aware of the different assumptions and interpretations for urban trees in the cultivated urban landscape. 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 understand urban forest population dynamics through the lens of empirical evidence. Tree monitoring and mortality have long been recognized as important components of sustainable urban forest management (Baker 1993, Clark et al. 1997). Urban tree demography is essential to understanding temporal changes in urban forest systems, just as tree demography is used to analyze change in natural forest systems (e.g., van Doorn et al. 2011). This primer lays the foundation for future studies that will use demographic approaches in urban forestry for basic and applied research.

In this primer, we only scratched the surface of population biology methods. For further reading, we recommend the following: Harcombe (1987) and Sheil et al. (1995) for classic overview articles, and textbooks by Seber (1982), Carey (1993), Caswell (2001), and Morris and Doak (2002).

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# Glossary

**annual mortality rate:** proportion of individuals dying over a year, from time $x$ to $x + 1$

**annual survival rate:** proportion of individuals surviving over a year, from time $x$ to $x + 1$

**censored data:** incomplete observations in longitudinal data

**cohort:** trees planted around the same time (for example, trees planted in the same year)

**demography:** the statistical study of populations and vital rates, such as birth and mortality, or in the case of urban trees, planting, death, and removal rates

**life table:** a table with survival and mortality information organized by age or size classes

**longitudinal:** involving repeated observations on the same individuals over time

**mean life expectancy:** the average age at death across the population

**mortality curve:** a graphic representation of mortality rate over time or across size classes

**population half-life:** the time at which half of a planting cohort has died

**survivorship:** proportion of a cohort surviving from planting to year $x$ (in other words, cumulative survival)

**survivorship curve:** a graphic representation of survivorship over time, typically displaying natural logarithm-transformed data; survivorship curves are relevant to display survival for planting cohorts

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Realizing the benefits of tree planting programs depends on tree survival. Projections of urban forest ecosystem services and cost-benefit analyses are sensitive to assumptions about tree mortality rates. Long-term mortality data are needed to improve the accuracy of these models and optimize the public investment in tree planting. With more accurate population projections, urban forest managers can also plan for cycles of tree planting, death, removal, and replacement to achieve canopy cover goals. Analytical tools from demography, such as life tables and mortality curves, could be used to improve our understanding of urban tree mortality and longevity. Demographic approaches have been widely used in forest ecology to quantify population dynamics and project future changes in wildland systems. This report is a primer on demographic concepts applied to urban trees, with terms and analytical methods adapted to the cultivated urban landscape. We include an overview of the uses of urban tree mortality rate data for research and management, a summary of lessons learned from ecological monitoring in other systems, and a discussion of opportunities for long-term urban forest monitoring by researchers and practitioners.

KEYWORDS: survivorship, survival rate, urban forest, tree monitoring, street tree, longitudinal, life table

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