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Original article

Predictors of street tree survival in Philadelphia: Tree traits, biophysical environment, and socioeconomic context

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ABSTRACT

Street trees are highly visible components of the urban forest, providing ecosystem services to city residents, but tree mortality can result in a loss of these essential benefits. Here, we focus on how street tree survival in Philadelphia, Pennsylvania (US) has been influenced by 1) tree traits and attributes, 2) the local biophysical environment, and 3) socioeconomic context. We used a longitudinal monitoring approach with a 6-year time interval and 103 randomly-located, fixed-length permanent plots, containing a range of tree size classes, to assess street tree survival rates, predictors of survival, and the balance of tree losses and gains. Street trees surveyed in this study experienced an annual mortality rate of 3.68% and followed a Type III mortality curve, with the highest annual mortality rates in the smallest size class. On the surveyed plots, there was a net decrease of live trees over the study period (2657 to 2542; 4.33% decline). Multivariate logistic regression revealed significant associations between street tree survival and tree crown vigor, species traits (expected mature stature and drought tolerance), site type, land use type, % impervious surface in a 15 m buffer, parcel-level demolition events, % of residents in poverty, % unoccupied households, and % renter-occupied households. Additionally, the final model included a multiplicative interaction term between crown vigor and tree size with separation in pattern of small (DBH $<$ 7.6 cm) from larger (DBH $\geq$ 7.6 cm) trees. Our findings suggest that targeted management interventions may enhance overall street tree survival, such as focusing on small trees, as well as trees in sidewalk cut-outs, commercial or industrial areas, or near demolition activity. Longitudinal monitoring studies such as ours provide valuable information about how urban forest systems change over time, with assessments that are valuable for management by municipal foresters and other street tree stewards.

1. Introduction

Street trees are a vital component of the urban forest, providing numerous ecosystem services (Nowak et al., 2001). In neighborhoods with high building density, small parcel set-backs, and minimal private yard space, these trees comprise some of the only greenspace residents encounter (McPherson et al., 1994; Pham et al., 2017). Street trees tend to be a major focus of planting initiatives and management by municipal arborists (Hauer and Peterson, 2016) due to their aesthetic appeal and high visibility (Roy, 2017). Street tree stewardship involving volunteers can provide opportunities for social connection, civic engagement, and education (Hauer et al., 2018; Johnson et al., 2018). However, in order to maximize the ecosystem services provided by trees, a common motivation for tree planting initiatives (Eisenman et al., 2021), each tree needs to survive and reach its mature size (Widney et al., 2016).

Applying demographic concepts and analyses adapted from traditional forest ecology can be a valuable tool in understanding street tree population[1] dynamics, particularly mortality (Roman et al., 2016).

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[1] The term "population" is used throughout this study to describe all trees in an area, though the trees are not all the same species, as is common convention among urban forestry studies (Roman et al., 2016).

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Trees in wildland forests or urban forest natural area patches generally experience new inputs via recruitment and losses via death from various stressors and disturbance events (Das et al., 2007; Piana et al. 2019; Johnson et al., 2021). In contrast, street tree systems are anthropogenically-constructed, with population cycles largely driven by direct human intervention (Roman et al., 2014a), with inputs from planting and losses via a combination of decline and failure, as well as human-initiated removal (Hilbert et al., 2019). Tree decline and associated failure may occur as trees approach the end of their functional lifespan, but can also be influenced or exacerbated by disturbances such as pests and pathogens or severe weather (Poland & McCullough, 2006; Staudhammer et al., 2011; Franceschi et al., 2023). Pulse disturbance events, such as fires and pest outbreaks, can cause severe and rapid tree mortality and canopy loss in both urban and rural settings (Gaiser et al., 2020; Konijnendijk et al., 2021). High rates of mature street tree failure can drastically change municipal forest systems and reduce the benefits associated with large trees (Hauer et al., 2020a). Street tree populations can also be limited by high rates of mortality in the years immediately following planting, during the establishment phase (Richards, 1979; Roman et al., 2014a).

Urban tree mortality also occurs through pre-emptive removal strategies associated with tree risk management or pest and disease outbreaks (Hilbert et al., 2019; Klein et al., 2019; Hauer et al., 2020a). Indeed, the *Ulmus americana* losses associated with *Ophiostoma* spp. in the northeastern region of the United States (US) in the mid-20th century were due not only to direct death from this infamous fungus, but also by eradication programs to remove threatened yet still living trees (Bukowski, 2019). Such local management decisions about disturbance threats can drastically impact mortality over time (Vannatta et al., 2012). Furthermore, city streetscapes are often utilized by pedestrians and exist in close proximity to built infrastructure, therefore municipal and utility arborists commonly remove trees deemed to be a risk to people, property, or power lines (Hilbert et al., 2019; Klein et al., 2019). Urban trees can also be removed for reasons unrelated to tree health, including human preferences (Conway, 2016) and construction and development activities (Morgenroth et al., 2017; Roman et al., 2022; Steenberg et al., 2018).

In this study, we focus on street tree mortality, both because of its importance to urban forest management and because a lack of empirical data limits conceptual understanding of street tree systems and calibration of ecosystem services projection models (Hilbert et al., 2019). We quantify street tree survival and mortality rates, predictors of survival, and the balance of tree losses and gains. Focusing on predictors of survival, we ask the following multi-faceted question: How is street tree survival influenced by 1) tree traits and attributes, 2) the local biophysical environment, and 3) socioeconomic context? The outcomes of this research can be used to guide conceptual and practical understanding of street tree population dynamics, thereby informing management decisions and highlighting areas for further investigation.

2. Methods

2.1. Approach

Our approach explicitly used a permanent plot design to facilitate longitudinal data collection on individual trees, adapting precedent from forest ecology (Reams et al., 2005; van Doorn et al., 2011; Condit et al., 2014). While other urban tree mortality studies have employed circular plots for monitoring (typically following UFORE/i-Tree Eco methods, e.g., Staudhammer et al., 2011; Lawrence et al., 2012), we used linear plots along city blocks to better capture street tree systems. Notably, although 67% of US cities and towns have inventories, such inventories may be outdated (Hauer and Peterson, 2016), and are generally oriented towards managing work orders (e.g., planting, inspection, pruning, removal), not evaluating change over time, and there are various data management challenges associated with using traditional street tree inventories for mortality studies (van Doorn et al., 2020). As with prior urban tree mortality studies (reviewed in Hilbert et al., 2019), our approach connected field data with variables from other sources related to socioeconomic status, stewardship, land cover, tree traits, and construction to investigate potential predictors of mortality/survival.

2.2. Study city

The study was conducted in Philadelphia, Pennsylvania (US), a city with a large area (369.9 km^2; Fig. 1) and human population (1.6 million residents as of 2020, 6th largest in the US). The city's people are racially diverse (40.8% Black or African-American, 38.5% White, 7.4% Asian, and 15.4% Hispanic or Latino), with relatively high poverty (22.8%, almost twice the national average; U.S. Census Bureau 2021). Prior to European colonization, Philadelphia was predominantly a mesophytic forest (Dyer 2006). The city is situated on the northern edge of the humid subtropical climate zone (Köppen-Geiger Cfa), characterized by cold winters, humid hot summers, and mild fall and spring, with year-round precipitation (Peel et al., 2007). Street tree management falls under the jurisdiction of Philadelphia Parks and Recreation (PPR), which is responsible for permitting plantings and removals and carrying out maintenance, although there is not currently a proactive street tree inspection program, and PPR is under-resourced due to decades of disinvestment (Roman et al., 2021). Property owners formally own adjacent street trees, and nonprofit stewardship programs have carried out extensive plantings in collaboration with residents and community leaders for several decades (Riedman et al., 2022). Many of the city's older neighborhoods have long stretches of attached housing (called rowhomes) with little private yard space, or sometimes none at all, making street trees critical to these residential areas.

In early 2023, PPR released its first-ever urban forest strategic plan, called the *Philly Tree Plan*. This citywide collaborative effort was focused on environmental justice and involved deep community engagement (City of Philadelphia, 2023), addressing current and historical inequities and disinvestments. As of 2018, overall urban tree canopy cover in Philadelphia equated to 20%, with substantial variation across neighborhoods (O'Neil-Dunne 2018), and the plan proposes to increase tree cover to 30% in every neighborhood over a 30-year period (City of Philadelphia, 2023), which would require extensive planting and long-term survival of street trees. Street tree management issues were raised repeatedly in this plan, which we relate to our findings in the discussion.

2.3. Study design and field data collection

The study spanned the entire city and was designed to be geographically representative of diverse neighborhoods and street types. The plot sampling scheme was adapted from a grid-based sampling frame used in the Forest Inventory and Analysis program of the USDA Forest Service (Reams et al., 2005). In 2015, plot locations were determined by selecting random points within cells of a 1.83 km square grid, which were used as starting locations for 103 fixed-length plots each traversing 0.8 km of total road length, randomly choosing a direction at each intersection, such that each plot was comprised of 1–12 city blocks (Fig. 1).

Longitudinal tracking of individual trees followed Roman et al. (2020) and van Doorn et al. (2020). In the summer 2015 baseline inventory, we recorded the characteristics (species, diameter at breast height (1.37 m; DBH), crown vigor) and location of all street trees in each plot. For purposes of this study, to enable consistency across space and time, street trees were strictly defined as woody vegetation at least 1.37 m in height and located within the sidewalk space (including sidewalk cut-outs and sidewalk planting strips) or medians. Front yard trees within the right-of-way where no sidewalk was present were not considered street trees for this study. During summer 2021, we

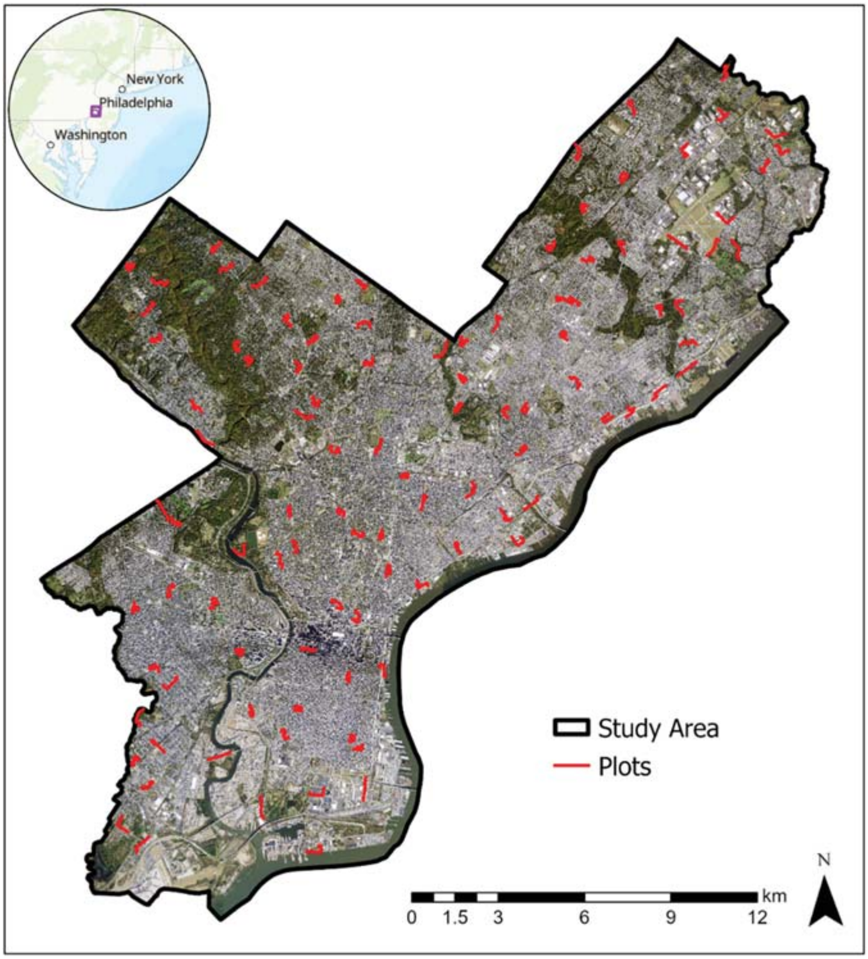

Fig. 1. Map illustrating locations of 103 fixed-length (0.8 km), permanent street tree monitoring plots in Philadelphia, PA displayed over 2019 NAIP satellite imagery.

remeasured each of the previously established plots, noting the mortality status of trees which existed in 2015 (alive, standing dead, or removed) and recording newly planted trees. To ensure longitudinal tracking of the same individual trees over time, location was carefully recorded using the block edge distance method (Roman et al., 2020), in which each tree's precise location is recorded using a measuring wheel along a given city block (Silva et al., 2013).

2.4. Predictors of street tree mortality

We investigated relative effect sizes of predictors of individual tree survival in three categories: 1) individual tree traits and attributes, 2) site conditions, and 3) local socioeconomic context (Table 1). Tree traits and attributes that have been statistically associated with street or urban tree survival include: tree size (Koeser et al., 2013; Roman et al., 2014a), initial foliage condition (Roman et al., 2014a), taxonomic group identity, and species' water use demand and expected stature at maturity (Hilbert et al., 2019; Roman et al., 2014b). Local biophysical factors that have been related to street tree survival include site type (Roman et al., 2014a), percent impervious surface (Vogt et al., 2015), and presence of overhead utility wires (van Doorn & McPherson, 2018). Local social factors that have been associated with street and urban tree survival include land use (Lu et al., 2010; Nowak et al., 1990), stewardship activity (Breger et al., 2019; Vogt et al., 2015; Lu et al., 2010), economic status (Vogt et al., 2015), measures of homeowner stability and educational attainment (Ko et al., 2015), and construction activity (Hauer et al., 1994; Morgenroth et al., 2017; Steenberg et al., 2018).

Individual tree attribute data were derived from a combination of field data (2015 DBH and crown vigor) and literature-based species characteristics (a full list of species included in the data set is included in Appendix Table A1). Tree diameter (DBH) was used to delineate individuals into "Small" (<7.6 cm) and "Large" (>7.6 cm) categories; this delineation aligns with previously reported variation in street tree survival rates associated with newly-planted trees during the establishment, when trees generally have the highest mortality rates (Hilbert et al., 2019). We used a four-class visual assessment of crown vigor for live trees that incorporates defoliation, discoloration, fine twig dieback, and branch mortality (Pontius & Hallett, 2014; Roman et al., 2020). Species attributes were derived from urban forestry fact sheets and guides (full citations and details in Appendix Table A1) including: expected stature (height) at maturity (small < 7.6 m, medium 7.6 m $\leq$ 15.2 m, large ≥ 15.2 m) and drought tolerance (low/medium or high).

Local site conditions were derived from a combination of field-collected data, remotely sensed data, and institutional records. Field crews recorded local site conditions including site type, land use, and presence of overhead wires. Site type describes a tree's immediate location (e.g., "sidewalk cut-out", "planting strip", and "median"), while land use describes how the property around the tree is used (categorized here into "residential", "commercial/industrial", and "park/other", Roman et al., 2020). To characterize local impervious ground cover we used 2013 1-meter resolution derived land cover data (University of Vermont Spatial Analysis Laboratory, 2016). Construction events were determined based on data posted to the city's open data portal, specifically, demolition permits that involved major structural and/or site

Table 1
Potential explanatory variables for predicting street tree survival used in iterative logistic regression model building process. "Local Site Conditions" refers to variables collected at the tree location scale either from field sampling or highly specific geospatial data.

Category	Variable	Variable Type (units or coding)	Data Source
Tree Trait & Attributes	Small Tree (DBH < 7.6 cm)	Binary (1 = Small, 0 = Not)	Field Monitoring, 2015
	Crown Vigor	Ordinal (1 = poor/ dying, 2 = fair, 3 = good, 4 = excellent)	Field Monitoring, 2015
	Drought Tolerance	Binary (0 = Low/ Med, 1 = High)	Tree Fact Sheets
	Expected Stature at Maturity	Categorical (Small, Medium, Large)	Tree Fact Sheets
Local Site Conditions	Site Type	Categorical	Field Monitoring, 2015
	Crown Light Exposure	Ordinal (0–5 Sides)	Field Monitoring, 2015
	Overhead Wires	Binary (1 = Present, 0 = Absent)	Field Monitoring, 2015
	% Impervious Surface (15 m buffer)	Continuous (100–0%)	UVM's 2013 HLRC, PASDA
	Demolition Permit	Binary (1 = Demolition Permit, 0 = No Permit)	PHL DOR Property Parcels, PHL Building and Zoning
	In Stewardship Group "Turf"	Binary (1 = Present, 0 = Absent)	Pennsylvania Horticultural Society
	Land Use Type	Categorical	Field Monitoring, 2015
Socioeconomic Context	Median Household Income	Continuous ($)	ACS, 2015–2019
	Median Household Value	Continuous ($)	ACS, 2015–2019
	% Unemployed	Continuous (%)	ACS, 2015–2019
	% Poverty	Continuous (%)	ACS, 2015–2019
	% Unoccupied Housing Units	Continuous (%)	ACS, 2015–2019
	% Renter-occupied Housing Units	Continuous (%)	ACS, 2015–2019
	% Moved in Last 5 Years	Continuous (%)	ACS, 2015–2019
	% Bachelor's or Higher	Continuous (%)	ACS, 2015–2019
	% Nonwhite Population	Continuous (%)	ACS, 2015–2019

work issued and completed after summer 2015 and prior to summer 2021 by the Department of Licenses and Inspections' Building and Zoning Permit and the Department of Records' Property Parcels (City of Philadelphia, 2022a, 2022b). To incorporate the potential effect of stewardship, we overlayed tree locations with a shapefile delineating the boundaries of the Pennsylvania Horticultural Society's Tree Tenders program, a network of neighborhood volunteer community groups that focus on planting and establishment maintenance, especially for street trees (Riedman et al., 2022), with each group having defined geographic boundaries spanning one or more neighborhoods.

To retrieve local socioeconomic and human demographic data from the US Census Bureau's American Community Survey (ACS), we used the 'tidycensus' package in R (Walker & Herman, 2022). Specifically, we used US Census tract-level measures of race, economic status, educational attainment, housing vacancies, owner versus tenant occupancy, and resident tenure. Individual trees received unique values for each sociodemographic variable using the inverse-distance-weighted calculation tool in ArcPro 2.9.2 (Esri Inc., 2022).

2.5. Data Analysis

Data were analyzed in R 4.0.2 (R Core Team, 2022), with much of the analysis using the 'tidyverse' package (Wickham et al., 2019), as well as ArcGIS Pro Version 2.9.2 (Esri Inc., 2022).

To calculate the annual mortality rate for trees in this study, we followed aforementioned demographic conventions (Roman et al., 2016) as follows:

$$Annual \quad Mortality \quad Rate[2015, 2021] = 1 - \left(\frac{Survived[2015, 2021]}{Alive[2015]}\right)^{(1/6)}$$

(Eq. 1, after Sheil et al., (1995) Eq. 6)

where *Alive*[2015] represents the total number of trees alive in 2015, and *Survived*[2015,2021] represents the number of trees that were observed alive in both 2015 and 2021. The above equation was used for all trees alive in 2015 (n=2657), and then separated by DBH size classes to construct a size class mortality curve (n=2654; with 3 trees excluded due to lack of DBH measurements).

To analyze potential factors associated with street tree survival (and thus, mortality), we used a model building approach to multiple logistic regression with 6-year tree survival (2015–2021) as the response variable. For the regression analysis we used a subset of the original 2015 inventory (n = 2606), excluding trees identified only to genus and where species level variability in traits and attributes within genera varied enough to preclude an acceptable degree of certainty of individual tree characteristics. Multiple logistic regression is frequently used in epidemiology to study binary outcomes in human populations, including death and disease (Hosmer et al., 2013; Jewell, 2004) as well as urban tree mortality studies (e.g., Breger et al., 2019). The model building approach has the advantage of incorporating comparative effect size analyses and multiplicative interactions. To build the logistic regression models in R, we used the 'glm' function (family = "binomial"). To compare constructed models via likelihood ratio tests, we used the 'anova' function (test = "LRT"). Both functions are in the 'stats' package (R Core Team, 2022).

The multivariate logistic regression model is expressed as follows:

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

(Eq. 2, after Jewell (2004) Eq. 14.2)

where X and Y represent independent risk factors, and $p_{x,y}$ represents the probability of survival given particular values of the independent risk factors. For ordinal risk factors (e.g., crown vigor), the coefficient b can be interpreted as the odds ratio (OR) — a measure of effect size in relation to the outcome variable — of a unit increase in x, holding the variable y fixed (Jewell, 2004). ORs greater than one indicate increased relative probability of survival; ORs less than one indicate decreased relative probability of survival. For nominal variables (e.g., land use), we used indicator ("dummy") variables, with one level selected as baseline for within-factor comparison. In this case, the OR is interpreted as the comparison of the given category to the baseline in relation to the outcome variable. Higher ORs in this context mean a higher probability of survival.

Regression models were built using an iterative model-building process to compare nested models using likelihood ratio tests. The final model had the highest likelihood, corrected for degrees of freedom (Hosmer et al., 2013; Jewell, 2004). We evaluated the final model fit using the Hosmer-Lemeshow goodness-of-fit (GOF) test (a small p-value indicates a lack of fit) and the receiving operator characteristic (ROC) curve (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 et al., 2013). For the GOF test, we used the 'hoslem.test' function in the 'ResourceSelection' package (Lele et al., 2019). For the area under the ROC curve, we used the 'lrm' function in the 'rms' package (Harrell, 2022).

In addition to each of the individual predictors indicated in Table 1, we tested for an interactive effect between DBH and crown condition, because small (and presumably young) trees are more susceptible to stress, injury, vandalism and ultimately, mortality (Miller & Miller, 1991; Richards, 1979; Roman et al., 2014a).

3. Results

The total number of live trees in our study decreased from 2657 trees in 2015 to 2542 trees in 2021. This represents a net decrease of 115 trees, or 4.33%. Of the 2657 live trees from 2015, 2122 survived to 2021, yielding an overall 6-year survival rate of 79.86% and an annual mortality rate of 3.68%. The size-class mortality relationship followed a Type III mortality curve (Fig. 2), meaning that the highest mortality was for the smallest size classes. Among the trees that experienced mortality, 443 were removed, 54 were left as stumps, and 38 were standing dead in 2021. We recorded 420 new live trees in 2021.

The final model (Table 2) predicting individual street tree survival included the following explanatory variables: crown vigor, expected stature at maturity, drought tolerance, tree size category, percent impervious surface, site type, land use, percent unoccupied housing units, percent renter-occupied housing units, construction (demolition), percent residents in poverty, and a multiplicative interaction between the variables denoting tree size and crown vigor (categories for each variable specified in Table 1). A model without the interactive effect between tree size and vigor was evaluated as well for comparison (Table A2). A likelihood ratio test and comparison of area under the ROC curve demonstrated improved model performance after including the multiplicative interaction. Measures of model evaluation for the final model with the interaction convey acceptable model discrimination (area under the ROC curve = 0.743) and no evidence of lack of fit (Hosmer-Lemeshow GOF p-value = 0.5568).

In this study, trees with the following traits and attributes were more likely to survive: better initial crown vigor, small or medium expected mature stature (compared to the baseline large stature), and high drought tolerance (compared to the baseline of low/medium drought tolerance; Fig. 3). Trees found in environments with the following local biophysical characteristics were more likely to survive: lower percentages of impervious surfaces in a 15 m buffer, and sidewalk planting strips or medians (compared to the baseline sidewalk cut-outs; Fig. 3). Trees found in environments with the following local site and socioeconomic and demographic characteristics were more likely to survive: residential or park land use types (compared the baseline commercial and industrial areas), no demolition construction events, higher percentages of unoccupied housing units, higher percentages of renter-occupied housing units, and lower percentages of residents in poverty (Fig. 3). The multiplicative interaction revealed that among trees with an excellent condition rating in 2015, larger trees (DBH > 7.6 cm) were more likely to survive than smaller trees (DBH < 7.6 cm). This effect was only significant for trees with "excellent" crown vigor rating (Fig. 5). Although we did not include taxa in our modeling (instead focusing on species traits), for context, annual mortality rates for the most prevalent families, genera, and species are provided in Table 3.

4. Discussion

The annual mortality rate for street trees in this study (3.68%) is relatively high compared to that of other repeat inventory studies of urban trees (i.e., not specifically street trees). A literature review by Hilbert et al. (2019) assessed results from 18 repeat inventory studies and found median tree mortality of 2.3%-2.6% (75th percentile of 3.0%-3.3%) and studies specifically evaluating street trees (but not necessarily repeat inventory studies), had a median annual mortality range of 2.5%-2.6% (75th percentile of 4.6%-5.5%). Prior street tree mortality studies that specifically focused on repeat inventories of mixed age and size classes (i.e., broadly comparable to our study design) had remarkably similar mortality rates. For example, a neighborhood in Oakland, California had an annual mortality rate of 3.7% with high levels of loss for young and small trees (Roman et al., 2014a). Annual street tree mortality rate for Cambridge, Massachusetts, was also very close to our Philadelphia findings and the Oakland results, at 3.6% (Boukili et al., 2017). City-scale analyses such as that presented here can be particularly useful for managers and ecosystem services projection models, as such studies reflect street tree mortality across a range of

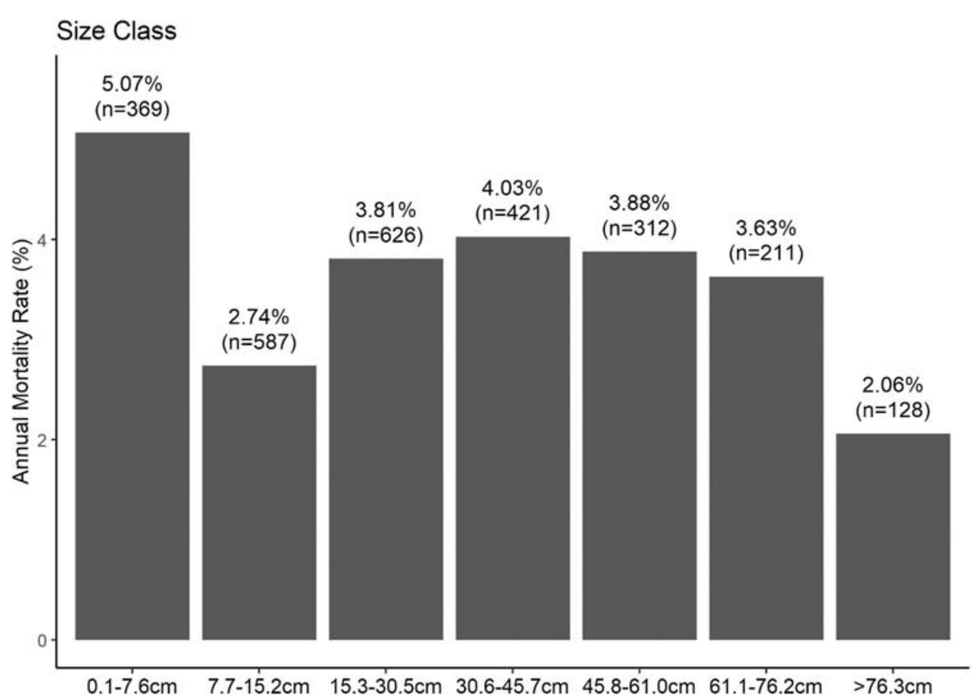

Fig. 2. Size-class mortality curve for the Philadelphia street trees (n=2654), using annual mortality calculated from the 2015–2021 observation period (Eq. 1). Bins are not all equal in size, see the *n* above each bin for tree counts. Note that the regression model used a threshold effect of trees in the smallest bin vs. all other trees.

Table 2
Logistic regression model results describing tree survival between 2015 and 2021 (n=2606). Included are OR estimates, 95% Confidence Intervals (CI), and p-values for each factor. ORs measure variable association with tree survival, comparing factor levels to a baseline (if categorical or binary). ORs > 1 indicate increased relative probability of survival. See Fig. 3 for the complementary forest plot.

Parameter	OR estimate	95% CI	*p*-value	Baseline
Crown Vigor				Poor/Dying
Fair	2.811	1.863, 4.263	< 0.001	
Good	5.430	3.671, 8.068	< 0.001	
Excellent	8.653	5.807, 12.964	< 0.001	
Stature				Large
Small	1.430	1.068, 1.926	0.017	
Medium	1.551	1.035, 2.382	0.039	
% Impervious Surface (15 m)	0.352	0.174, 0.707	0.004	
Drought Tolerance				Low/Medium
High	2.594	2.051, 3.290	< 0.001	
Small Tree (DBH < 7.6 cm)				No (DBH > 3in)
Yes	1.042	0.499, 2.169	0.912	
Site Type				Sidewalk Cut-out
Median	2.163	1.373, 3.474	0.001	
Sidewalk Planting Strip	1.402	1.084, 1.817	0.010	
Land Use				Commercial / Industrial
Family/Residential	1.715	1.310, 2.245	< 0.001	
Park/Other	1.680	1.028, 2.813	0.043	
% Unoccupied Housing Units				Low (< 7%)
Med (7% - 12%)	1.573	1.194, 2.075	0.001	
High (> 12%)	1.750	1.264, 2.433	0.001	
% Renter-Occupied Housing Units	3.272	1.348, 8.030	0.009	
Construction (Demolition)	0.275	0.110, 0.723	0.007	
% Residents in Poverty	0.259	0.075, 0.888	0.032	
Interaction between Crown Vigor and Tree Size				Poor/dying
Fair	0.895	0.329, 2.494	0.829	
Good	1.346	0.549, 3.369	0.52	
Excellent	0.348	0.150, 0.809	0.014	
Measures of Model Fit				
AIC	2294.8			
AUC	0.741	Acceptable		
Hosmer-Lemeshow	0.3951	No evidence of poor fit		

biophysical and socioeconomic conditions throughout the city and allow inferences at the scale of municipal forest management.

As a reflection of the importance of tree health to future mortality, trees that had poorer initial crown health were less likely to survive. On the one hand, this finding is fairly straightforward and aligns with prior studies which found associations between urban tree mortality and measures of tree condition (Koeser et al., 2013; Roman et al., 2014a). However, live trees in poor condition can also be removed to mitigate perceived safety risks (Hilbert et al., 2019; Klein et al., 2019) or due to aesthetic and land use preferences (Conway, 2016; Roman et al., 2022). Our analysis of the multiplicative interaction between crown vigor and initial tree size is particularly illuminating. Among trees with an excellent crown vigor rating, small trees (DBH < 7.6 cm) were substantially less likely to survive than larger trees (annual mortality 4.51% for DBH < 7.6 cm and 1.96% for DBH > 7.6 cm; Fig. 2). Young, newly-planted trees are more susceptible to injury and vandalism (Miller & Miller, 1991; Richards, 1979), which can increase the likelihood of tree mortality even in otherwise healthy trees (Hilbert et al., 2019; Manion, 1981). Young trees may also be considered expendable, due to relative ease of removal during construction (Roman et al., 2022). In contrast, large tree removal requires specialized equipment, trained arborists, and associated funding (Morgenroth et al., 2017).

Demolition was associated with tree mortality in our analysis, matching prior results on urban tree survival across public and private landscapes (Koeser et al., 2013; Morgenroth et al., 2017; Roman et al., 2022; Steenberg et al., 2018). In Christchurch, New Zealand, demolition-related removals primarily involved smaller trees in close proximity to buildings (which could be related to ease of removal) and were more common with increasing land value, suggesting that potential economic gains from redevelopment provided an incentive for tree removals (Morgenroth et al., 2017). On a college campus in Philadelphia, capital projects and other construction activities were directly implicated in tree removals (Roman et al., 2022). Our findings align with these studies through the use of parcel-level construction and development data, and we recommend that future urban tree mortality studies take advantage of municipal open data to facilitate such analyses. Tree preservation and protection ordinances can minimize street tree damage during construction and/or curtail removals (Hauer et al., 2020b). The *Philly Tree Plan* recommends several ways to strengthen tree preservation, planting, and replacement requirements for development projects, including updates to the zoning code and improved coordination across municipal departments (City of Philadelphia, 2023)

The relationship between species characteristics (drought tolerance and expected mature stature) and tree survival was substantial and may work in conjunction with local site characteristics (i.e., as tree-site compatibility). Trees classified as having higher levels of drought tolerance had a higher likelihood of survival, which matches expectations, as water stress is a limiting factor for urban tree health, growth, and survival due to lack of sufficient irrigation (Roman et al., 2014b; Breger et al., 2019), urban heat island effects, and soil compaction, which limits water infiltration (Day & Bassuk, 1994). Expected mature stature also appeared to influence urban tree survival, with trees expected to reach large-stature experiencing the highest mortality rates (Fig. 4C). This pattern may be related to generally higher demands for soil availability in large-statured trees, or human-driven management decisions focused on limiting conflicts with overhead utility wires. However, neither presence of utility wires, nor a multiplicative interaction between expected mature stature and utility wires, were significant predictors of survival in this study. Notably, the *Philly Tree Plan* emphasizes the need to increase staffing to assist with tree-infrastructure conflicts, including a city forester, additional arborists, street tree inspectors, and a sidewalk coordinator (City of Philadelphia, 2023).

Additional aspects of the biophysical environment that were associated with tree survival included site type and percent impervious surfaces in a 15 m buffer. These factors can be considered predisposing factors, which create vulnerabilities to later stressors (Hilbert et al., 2019; Manion, 1981). However, findings from past literature for site type and nearby impervious surface in the literature are inconsistent (Lu et al., 2010; Roman et al., 2014a). One possible mechanism for the pattern observed here, where trees were more likely to survive in planting strips and medians than sidewalk cut-outs, is variation in soil availability (Sanders & Grabosky, 2014). Sidewalk cut-outs in our study tended to be relatively small (as small as 61 cm x 61 cm), whereas

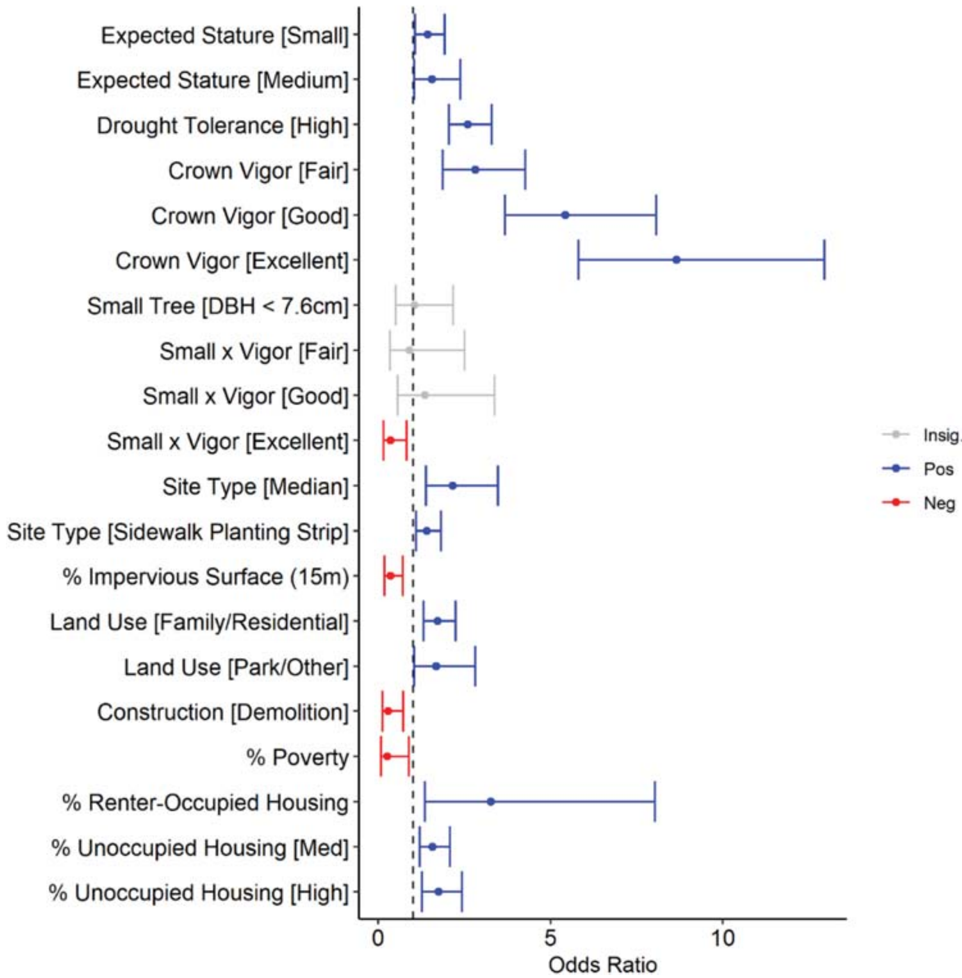

Fig. 3. Forest plot displaying variable effect sizes for significant predictors in the logistic regression analysis (model details in Table 2). Points represent Odds Ratios measuring variable association with tree survival, comparing factor levels to a baseline (if categorical or binary). Bars represent 95% confidence intervals for Odds Ratios. Blue indicates increased likelihood of survival, Red indicates decreased likelihood of survival, and gray indicates a non-significant association in this analysis for the specified factor level. Differences in effect sizes for factor levels were assessed at the Alpha = 0.05 level.

sidewalk planting strips provided more available soil and infiltration capacity and medians were comparatively wide, with a greater available soil volume. Indeed, Philadelphia's built environment characteristics, including dense attached housing and narrow sidewalks, may make our street tree growing conditions more comparable to old, dense European or Asian cities, or other pre-World War II cities and towns in the New England and mid-Atlantic regions of the US, in contrast with the relatively more recently developed cities in the US mid-west, which often have wide "tree lawns" between the sidewalk and street. However, we did not systematically measure apparent plant available soil (nonpaved planting zone area, Sanders and Grabosky, 2014) which would better reflect improved capacity for root growth, nutrient availability, and water infiltration. Also, our definition of street trees excluded front yard trees in the right-of-way where no sidewalk was present, locations where impervious influence would be substantially lower. Limitations in available soil may describe the negative association found between percent of local impervious surfaces and tree survival (Vogt et al., 2015). However, further research on soil conditions or employing the rapid urban site assessment index (Scharenbroch et al., 2017) is needed to understand the relationship between street tree mortality and urban soil quantity and quality, which may be highly tree-, site-, and region-specific.

Land use at the parcel adjacent to a given street tree was also associated with survival with trees next to parks and residential properties more likely to survive than those in commercial and industrial areas. This finding aligns with prior studies focused on young street tree survival (Lu et al., 2010; Nowak et al., 1990), where the relationship between land use and survival was suggested to relate to sense of ownership and associated level of care, as well as level of pedestrian traffic, which may influence factors such as vandalism or soil compaction. However, our measure of stewardship was not significant in the final model. Stewardship and maintenance indicators at the level of individual trees have been associated with tree mortality in past studies (Boyce 2011; Breger et al., 2019). We suspect that our metric of stewardship (presence of a Tree Tenders' group) was too coarse, yet finer-scale tree care data is not available for our study area. Further research is needed to understand mechanistic relationships between land use, stewardship, biophysical stressors, and street tree survival at the scale of planting sites, parcels, and city blocks.

Complex relationships emerged between street tree survival and socioeconomic context, which can set initial conditions for future management decisions about maintenance and removal (Hilbert et al., 2019). The three socioeconomic variables which were significantly associated with street tree survival in this study were: percent of unoccupied housing units (positive), percent of renter-occupied housing units (positive), and percent of residents in poverty (negative). It should be noted, however, that these socioeconomic variables reflect the US Census tract scale, not parcel-level conditions, nor do they reflect the complexities of temporally dynamic street tree systems. The positive association of unoccupied housing units and renter-occupied housing

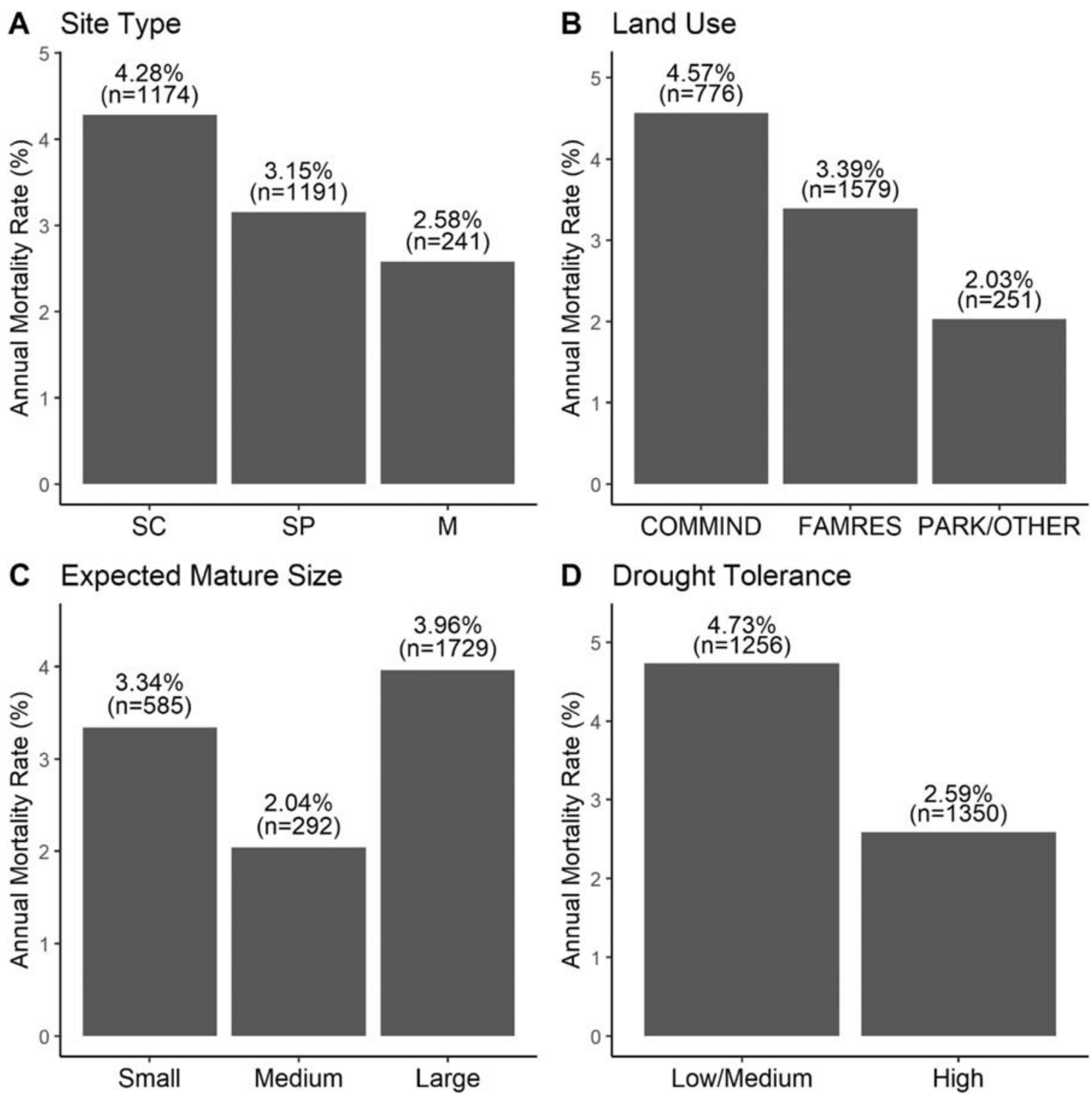

Fig. 4. Bar charts of selected variables significantly associated with tree survival in logistic regression modeling. Within-variable factor levels are displayed by annual mortality rate. A) Site Type (SC = Sidewalk Cut-out; SP = Sidewalk Planting strip; M = Median); B) Land Use (COMMIND = commercial, industrial, mixed use, institutional, vacant lots, and transportation; FAMRES = multi-family residential, single family residential - attached, and single family residential - detached; and PARK/OTHER = maintained park, natural areas, and other); C) Expected Mature Size; D) Drought Tolerance.

units with street tree survival may seem counterintuitive, as stability of owner-occupied units may increase a sense of ownership and stewardship activities related to urban tree care (Ko et al., 2015a; Roman et al., 2014b; Vogt et al., 2015). However, the pattern we observed in Philadelphia may reflect a lack of proactive management, as mature tree removal requires investment of time and money (Roman et al. 2022). For example, in Detroit, Michigan, residents noted that increases in vacant properties led to a lack of street tree management and maintenance practices, such as removals of trees in poor and unsightly condition (Carmichael & McDonough, 2019). Research in Indianapolis, Indiana also found a positive association between renter-occupied housing units and street tree survival (Vogt et al., 2015). Mixed-methods investigations that combine quantitative mortality analysis with qualitative stewardship assessments could help to disentangle the mechanisms behind such relationships (Breger et al., 2019).

The negative association between street tree survival and poverty at the US Census tract scale indicates lower survival in areas with higher poverty. Limited available funding for tree maintenance and care and smaller planting spaces in lower-income neighborhoods could both indicate a harsher growing environment (Nowak et al., 1990). The *Philly Tree Plan* is prioritizing new plantings in neighborhoods of greatest need, which are defined, in part, by socioeconomic status (City of Philadelphia, 2023), yet our finding points to potential difficulties in increasing the street tree populations in such neighborhoods. If the trend we observed holds true in other cities, it could point to inequities in tree survival/mortality, in addition to inequities that sometimes manifest with new tree plantings (Locke and Grove, 2014). However, the array of socioeconomic factors potentially associated with street tree survival requires situational contextualization about governance and stewardship policies and actions, which may vary considerably across and within cities. Future research could incorporate, for example, structured interviews with residents or property owners adjacent to street trees, as well as organizations formally or informally responsible for street trees along certain city blocks (e.g., neighborhood civic associations, business districts, universities, homeowners associations). Firsthand accounts and reflections from community members and other stakeholders

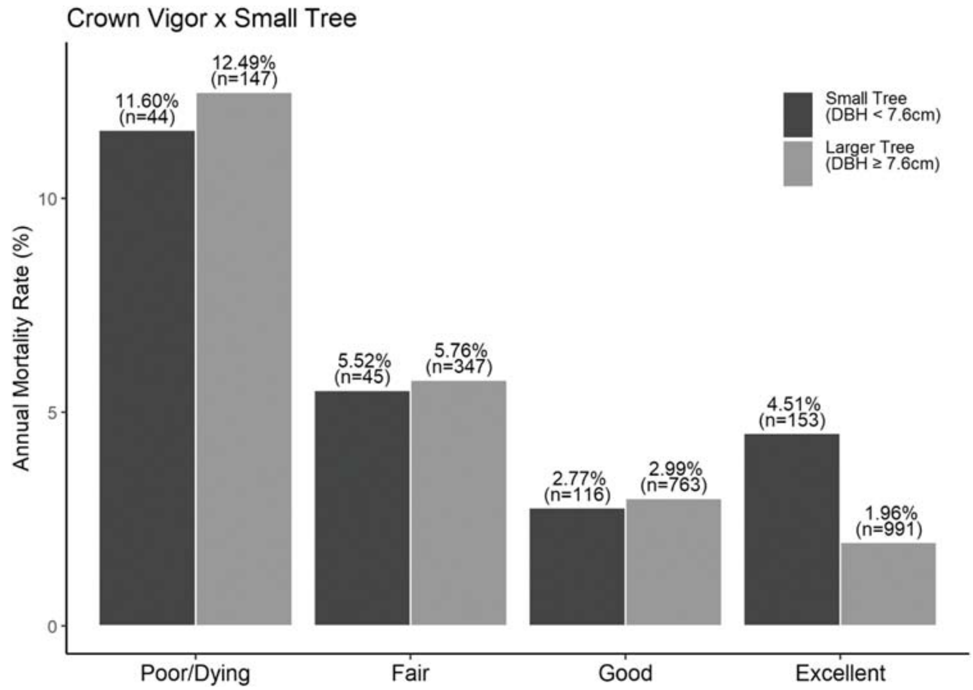

Fig. 5. Bar chart displaying interaction between crown vigor and relative tree size in relation to mortality as captured in the final logistic regression model. Small trees (DBH <7.6 cm) only had significantly greater tree survival OR than large trees when crown vigor was "Excellent".

Table 3
Tree counts and annual mortality rates (AMR, given as a percent) for the ten most common families, genera, and species in the initial (2015) survey.

Family	2015	AMR	Genus	2015	AMR	Species	2015	AMR
Sapindaceae	593	5.1	*Acer*	573	5.3	Platanus × *hispanica*	367	2.3
Rosaceae	570	2.8	*Platanus*	367	2.3	*Prunus* sp.	316	2.7
Platanaceae	367	2.3	*Prunus*	316	2.7	*Acer platanoides*	205	7.8
Fabaceae	253	3.8	*Quercus*	195	3.2	*Acer rubrum*	189	4.8
Fagaceae	195	3.2	*Gleditsia*	130	1.6	*Gleditsia triacanthos*	130	1.6
Ulmaceae	192	1.4	*Zelkova*	129	1.5	*Zelkova serrata*	129	1.5
Malvaceae	123	5.1	*Pyrus*	126	2.1	*Pyrus calleryana*	125	2.0
Oleaceae	94	8.8	*Tilia*	123	5.1	*Tilia cordata*	78	5.1
Ginkgoaceae	58	2.1	*Crataegus*	68	2.1	*Crataegus* sp.	68	2.1
Altingiaceae	39	1.8	*Ulmus*	63	1.1	*Acer campestre*	61	1.4

regarding the governance and social environment of street trees would help shed light on social processes related to mortality (Breger et al., 2019).

Future research on urban tree mortality would also benefit from quantified empirical data on species and cultivar traits, such as leaf water potential and other leaf traits (Sjöman et al., 2018; Tabassum et al., 2021). Such data is largely missing from the existing urban tree mortality literature (Hilbert et al., 2019), and the lack of such data for the species in our study system was also a limitation of our research. Findings from studies focused on urban tree growth and health outcomes (Ordóñez-Barona et al., (2018); Moser-Reischl et al., 2019) indicate the need for inclusion of species trait factors in mortality investigations. However, our finding that coarse drought tolerance categories predicted survival shows the utility of such categories (from arboricultural species guides) for analyzing survival/mortality patterns.

Notably, our study system did not experience catastrophic pulse disturbances leading to massive tree loss during 2015–2021, therefore our observed survival rates can be considered as background levels of street tree mortality (Lugo and Scatena, 1996). Indeed, even though a fatal invasive non-native insect, *Agrilus planipennis*, was introduced to Philadelphia during our study period (2016), we did not observe a severe impact on overall mortality for the system because *Fraxinus* spp. are rare as a street tree in Philadelphia (constituting only 54 trees in our plots in 2015, or 2.0%) and because there were not extensive pre-emptive removals along streets (as other US cities with *Fraxinus* spp. monocultures have done, e.g., Heimlich et al., 2008). In Philadelphia, *Fraxinus* spp. street trees experienced 15.3% annual mortality rate during our study period; this is higher than many other species (Table 3) but not the highest species-level annual mortality rate (100% mortality of 20 *Gymnocladus dioicus* trees). PPR completed a comprehensive city-wide inventory of street trees in 2021, but during our study period, the locations of *Fraxnius* spp. along all city streets was unknown to municipal forest managers. Current PPR practice is to carry out ongoing pest control treatments for several dozen *Fraxinus* spp. street trees to sustain their health, while other *Fraxinus* spp. along streets are being removed (pers. comm., PPR staff, Feb. 2024). In contrast, *Fraxinus* spp. were 13.0% of trees in the city's parks in 2012 (Nowak et al., 2016). Municipal forestry staff in PPR received a grant to carry out a management plan to inventory *Fraxinus* spp. trees in neighborhood parks and along nature trails prior to the pest arriving in the city and conduct a mix of pre-emptive removals and treatments. As these details about Philadelphia's urban forest management illustrate, species prevalence from past planting legacies, available inventory data, management actions, and municipal capacity profoundly shape the extent of tree loss from invasive insects (Vannatta et al., 2012; Greene and Millward, 2016).

Additionally, Philadelphia experienced regular storm disturbances, but not a singular catastrophic event, during the study period. There can be drastic urban tree mortality and canopy loss from hurricanes, ice storms, and tornados, due to both trees felled in storms and those removed afterwards (Staudhammer et al., 2011; Conway and Yip, 2016). For cities with robust municipal tree inventory software, research on street tree mortality could also incorporate administrative removal records (Roman et al., 2022; Ordóñez et al., 2023). However, Philadelphia did not have such records during our study period. In the same vein as permanent plots in forest ecology, our fixed street tree plots with longitudinal monitoring can enable future research into system changes following major disturbance events (Staudhammer et al., 2011; Gaiser et al., 2020) and climate change (Esperon-Rodriguez et al., 2022), as well as management responses to those stressors.

Finally, the socioecological context in Philadelphia – a relatively large, post-industrial US city, with an under-resourced municipal parks department and dense housing development, located within a naturally forested biome – is broadly relevant to the interpretation of our results, and situates our findings in a particular place and time. However, given that urban forest governance and ecological conditions vary considerably within and across nations and over time, and that the US is over-represented in the urban tree mortality literature (Hilbert et al., 2019) and street tree research (Roy et al., 2012), more research is needed on street tree mortality from other geographic areas (Shackleton, 2012; Ordóñez-Barona et al., (2018)). Future research is also needed to evaluate the impacts of massive planting initiatives on street tree demography, as the US and other countries are currently experiencing unprecedented levels of investment in urban forestry (Eisenman et al., 2021; UNECE, 2024).

5. Conclusion

Our study adds to a growing body of urban tree mortality research and provides a system-specific analysis in Philadelphia at a time when municipal staff and partners are strategically aiming to grow the urban canopy. Our study is the kind of longitudinal research called for in the literature review by Hilbert et al. (2019), and we reiterate the need for interdisciplinary and transdisciplinary mixed-methods studies to understand the spatio-temporal change in street tree systems, recognizing urban forests as complex social-ecological systems (Vogt et al., 2015; Roman et al., 2022). The timing of our study also provides a unique opportunity to assess trends and patterns existing prior to the *Philly Tree Plan*, and future remeasurements of our permanent plots could assess changes related to plan implementation, including new plantings and policies that could promote tree preservation. Our findings suggest that street tree survival may benefit from targeted management interventions such as tending of small trees and those in sidewalk cut-outs, commercial or industrial areas, or near demolition activity. However, comparative mortality studies are needed across different regions and countries with distinct ecological, socioeconomic, and political contexts. Our approach – permanent plots with longitudinal tree monitoring – will enable a variety of future research avenues to deepen understandings of street tree dynamics in the face of shifting management, exogenous pressures (e.g., climate change), and pulse disturbances.

CRediT authorship contribution statement

Jason Grabosky: Writing – review & editing, Resources, Funding acquisition. **Robert Fahey:** Writing – original draft, Supervision, Resources, Project administration, Funding acquisition, Conceptualization. **Levon Bigelow:** Writing – original draft, Investigation, Formal analysis, Data curation. **Michelle Johnson:** Writing – review & editing, Resources, Methodology, Funding acquisition, Conceptualization. **Jason Henning:** Writing – review & editing, Resources, Methodology, Conceptualization. **Richard Hallett:** Writing – review & editing, Resources, Methodology, Funding acquisition, Conceptualization. **Lara Roman:** Writing – original draft, Supervision, Resources, Project administration, Methodology, Funding acquisition, Conceptualization.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Robert Fahey reports financial support was provided by Tree Research and Education Endowment Fund. Robert Fahey reports financial support was provided by USDA Forest Service Northern Research Station. Robert Fahey reports financial support was provided by National Institute of Food and Agriculture. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ufug.2024.128284.

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