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Construction and Proactive Management Led to Tree Removals on an Urban College Campus

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Abstract: Urban trees in lawns and along streets are anthropogenically constructed systems, in that these tree communities are formed by human planting and removal actions. Tree mortality studies are essential to understanding the temporal dynamism of urban forests, and in particular, it is critical to incorporate institutional records and human decision-making regarding tree removals. In this study, we investigated tree removals on a highly urbanized college campus in Philadelphia, Pennsylvania (US) by analyzing field inventories and institutional records, and by considering firsthand accounts of the University Landscape Architect. The annual mortality rate was 4.3%, higher than typical for comparable studies, which we attribute to construction pressure and proactive management to promptly remove unhealthy trees and manage risk. Capital projects and other construction caused 48.5% of all removals, other human land use decisions caused 2.0%, and tree health decline and risk management collectively accounted for 48.7%. The number of removed trees exactly equaled the number of new trees, and the campus has high taxonomic diversity, reflecting the extensive oversight by university tree and landscape professionals regarding tree removal and planting decisions. This study demonstrates the value of mixed-methods and transdisciplinary research to understand how urban forests change over time.

Keywords: urban densification; redevelopment; building renovation; urban forest; tree monitoring; urban greening; urban greenspace; urban tree mortality; tree demography; sustainable campus

Citation: Roman, L.A.; Fristensky, J.P.; Lundgren, R.E.; Cerwinka, C.E.; Lubar, J.E. Construction and Proactive Management Led to Tree Removals on an Urban College Campus. *Forests* **2022**, *13*, 871. https://doi.org/10.3390/f13060871

Academic Editors: Michael Andreu, Basil V. Iannone and Ryan W. Klein

Received: 1 April 2022
Accepted: 27 May 2022
Published: 1 June 2022

1. Introduction

Urban forests—the trees in cities and urbanized areas [1]—are dynamic socio-ecological systems co-produced by past and present human and biophysical forces [2,3]. Over the past few decades, municipal leaders have sought to increase tree canopy levels and launched massive planting campaigns [4,5], often rooted in conceptualizations of trees as green infrastructure that delivers ecosystem services [6,7]. Growing urban canopy involves human management of urban forest population cycles and taxonomic composition, including species selection at planting and patterns of mortality and replacement [8,9]. As Clark et al. [10] explained, "sustainable urban forests require human intervention". In particular, trees in lawns, along streets, and in plazas are anthropogenically constructed portions of the urban forest, in that these tree communities are formed primarily by human planting, rather than natural regeneration from seedlings [8,11]. Urban forests in hardscape and lawn environments are comprised of numerous tree planting cohorts of varying age classes, leading to complex size structures and taxonomic composition reflecting legacies of past human actions [12–14]. To achieve a net gain in tree counts, the number of new trees must outnumber the losses of existing trees, all in the context of limited plantable space [11,15]. The design of future urban forests, therefore, depends not only on planting campaigns but also on the pace and patterns of removals.

Tree mortality studies are essential to understanding fluxes in urban forest systems, including the impacts of management decisions. Notably, the mortality of street and lawn trees is a considerably different process compared to the mortality of trees in natural, wildland forests (and natural forested areas within cities). In wildland forests, tree mortality results from a slow accumulation of various stressors, often over many decades [16], a concept articulated by Manion [17] as the disease-decline spiral and later adapted by Franklin et al. [18] as the mortality spiral. Dead trees then serve ecological functions, including wildlife habitat and decomposition [18,19]. In contrast, when street and lawn trees die in-place, they can become dangerous to people and property [20] or signify a lack of care [21,22]. Arboricultural best practices in tree risk management generally call for dead urban trees to be removed [23]. Urban tree mortality can also result from the removal of living trees: both pre-emptive removal due to health and safety concerns (e.g., tree risk management) and removal due to aesthetic considerations or human land use preferences [24]. The pre-emptive removals are carried out to reduce risk of injury to people and property and curtail liability [20,25]. In other words, humans intervene to fell urban trees before the mortality spiral completes. Kirkpatrick et al. [26] referred to the removal of trees that are diseased or of advanced age as 'tree euthanasia', language that evokes the fact that humans are deliberately cutting short the mortality spiral.

Unfortunately, many urban tree mortality studies do not capture the human-directed nature of removal, as the reasons for removal are not often discernable from field observations alone. In a recent literature review, the factors most often statistically associated with urban tree mortality were taxa or species characteristics (e.g., drought-tolerant species), tree size or age, site characteristics (e.g., planting space, site type), stewardship, and local sociodemographic metrics [24]. Those studies used variables derived from field observations, planting records, and geospatial socioeconomic data to build statistical models for mortality outcomes (e.g., [2,9,11,27]). Mortality is generally higher for trees with smaller trunks, indicating greater vulnerability for younger and/or smaller trees to death and removal [8,24]. Urban tree mortality studies typically use data that is relatively easy to obtain during field work, or, alternatively, use a combination of field observations and sociodemographic data readily available from public sources, but there is a need for other mixed-methods approaches [24], particularly those utilizing data pertaining to removal decisions. Notably, even massive mortality events in urban forests, such as invasive pests and pathogens, have human removal decisions as a driving force, with pre-emptive removals sometimes carried out to quarantine outbreaks or avert the need for ongoing treatments [28–30]. Therefore, understanding the causes of or justifications for removal is critical to understanding mortality rates and drivers in urban forests.

With urban forests as fundamentally socio-ecological systems [2,31] and many city trees located in engineered landscapes managed by and for humans [32], there are alternative ways of generating knowledge about tree mortality. It is important to assess direct causal mechanisms for mortality, including why humans remove trees that are not yet dead. Studies that have investigated administrative records, local expert knowledge, and resident experiences have shown that that development activities, tree risk and health perceptions, and aesthetic preferences were impetuses for tree removal and canopy loss [26,28,33–35]. Local institutional records, such as municipal renovation permits and demolition records, have rarely been included in mortality analyses, but when they were, such construction activities were found to be associated with tree loss [36–39]. Indeed, tree removal, and its flip side, tree preservation, are central to urban forest management and governance [38,40,41]. For instance, many cities have enacted public policies, such as tree preservation ordinances and by-laws, to curtail tree loss associated with development [35,41]. Additionally, compositional changes in urban forest systems are tied not only to rates of loss but also to changing planting decisions over time, as certain species gain or lose favor [3,14]. Incorporating perspectives from land managers and administrative records can promote transdisciplinary understandings of urban tree population and community dynamics, and the central role of human decision-making in orchestrating change in sylvan urban landscapes.

To investigate mortality and compositional shifts in an urban forest, we used a unique data set of two linked inventories, 11 years apart, from an urban college campus. By connecting tree removals with institutional records and land manager knowledge, we were able to attribute direct causal mechanisms for tree loss. Our research questions were: (1) Based on institutional records and land manager knowledge, what caused tree removal? (2) How well do tree size and site type predict mortality? (3) How did the urban forest system change, in terms of removals and planting, and the species composition? We then discuss implications for urban forest management.

2. Materials and Methods

2.1. Study System

Our study took place at the University of Pennsylvania (Penn), a private university located in Philadelphia, Pennsylvania, United States (US). Philadelphia is currently the sixth-largest city in the US, with a population of 1.6 million as of 2020 [42]. Philadelphia is at the northern edge of the humid subtropical climate zone [43], with steady year-round precipitation, hot humid summers, cold winters with variable snowfall, and mild spring and fall. The mean January temperature is 0.9 °C, mean July temperature is 25.9 °C, and mean annual precipitation is 1120 mm [44].

The Penn campus is west of the Schuylkill River in the University City neighborhood. Philadelphia's natural terrestrial ecosystems are largely mesophytic forests [45], with wetland areas along many creeks and rivers [46]. The campus and the city include the fall line between the Piedmont (generally hilly and forested) and Atlantic Coastal Plain (generally flat and swampy) physiographic regions [47].

During our study period (2003–2014), campus tree canopy cover ranged from 19.1–21.6%, a substantial rise from only 8.7% tree cover in 1970 [48]. Following the physical expansion of the campus through urban renewal policies in the mid-20th century, a Landscape Development Plan in 1977 ushered in renewed attention to vegetation management and landscape beautification [49]. The Penn campus has been designated as a Tree Campus USA by the Arbor Day Foundation since 2009 and was recognized as an arboretum in 2017 [48]. The campus is embedded within a highly urbanized neighborhood, surrounded and bisected by high-traffic municipal roads, including streets with frequent public transit buses and trucks, and streets designated as snow emergency routes. Many of the internal campus walkways are former streets that were closed to public vehicular traffic decades ago [48] but are still regularly used by university vehicles.

We focused on the core campus (63 ha, Figure 1): trees in lawns, plazas, and along streets. We did not include the Botanic Garden—the largest wooded space on campus—which had incomplete inventory records. In the core campus, the University Landscape Architect, Landscape Planner, and Urban Park Manager are responsible for vegetation management, including trees. These staff are under the Penn Facilities and Real Estate Services (FRES) office, which manages the physical assets of the university, with the University Landscape Architect and Landscape Planner being under the Office of the University Architect. These personnel work with other Penn ground crews, arborists, designers, gardeners, and external contractors to plant, maintain, and remove trees. Campus tree management is also informed by the Morris Arboretum's Urban Forestry Consultants, who are part of the Penn system, with offices located in the Chestnut Hill neighborhood of Philadelphia. The consultant team includes a Board-Certified Master Arborist. While Morris Arboretum's tree professionals have been involved in campus landscape management since at least the late 1970s [48], the Morris–FRES partnership became more formal and intensive in the early 2000s. This partnership included the Morris Urban Forestry

Consultants carrying out the tree inventories described below and tree inspections to provide pruning, preservation, and removal recommendations. These inspection cycles became more frequent around 2005, covering one-quarter of the campus every year, and shifted to annual inspections of all trees in 2014. Inspections and other tree management tasks by Morris consultants are currently formalized through a FRES-Morris Memorandum of Understanding. In short, since the early 2000s, the campus urban forest has been proactively managed by landscape and arboricultural professionals, in contrast to the situation in the 1970s, when the landscape was neglected and numerous trees were diseased or dying, and professional tree care was lacking [48,49]. Campus tree management has been steadily growing more proactive since the 1977 Landscape Development Plan.

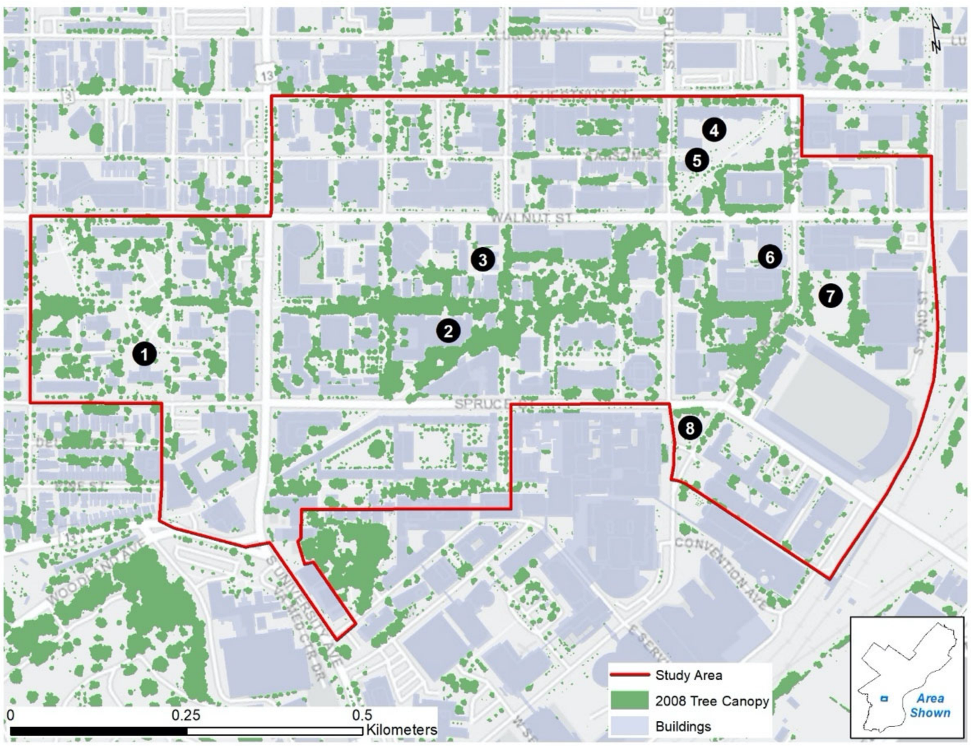

Figure 1. The University of Pennsylvania campus, displaying the core campus area used in this study. Numbers indicate places discussed in the text: (**1**) Harrison College House, (**2**) Steinberg-Dietrich Hall, (**3**) Annenberg Public Policy Center, (**4**) New College House, also known as Lauder College House, (**5**) McNeil Center for Early American Studies, (**6**) Skirkanich Hall, (**7**) Shoemaker Green, and (**8**) Edward W. Kane Park.

2.2. Case Study Approach

Using a case study approach, we investigated the phenomena of urban tree mortality in-depth in a real-world context [50]. As a group of co-authors, we are all current or former students or staff at Penn—a mix of scholars and landscape professionals, representing transdisciplinary research that has been called for in urban forestry [51,52].

Furthermore, we treat the campus as an experimental landscape and living laboratory for the study of urban tree temporal dynamics [48,53], following in the tradition of other urban forestry studies that have used college campuses to investigate ice damage [54] and dendrochronology [55]. Additionally, some college campuses have recently set tree cover goals and used ecosystem services discourse in ways similar to municipalities [48], making campus case studies relevant to broader urban forestry research. At Penn, a student report estimated the ecosystem services of campus trees [56], and a campus climate action plan mentioned the importance of maintaining the integrity of the campus urban forest [57]. These reports align Penn with broader trends in urban forestry, whereby professionals view trees through the lens of ecosystem services [6]. Additionally, the tree cover on this campus has been shaped over many decades by the same historical forces that influenced urban forests more broadly, such as the City Beautiful movement, urban park creation, and urban renewal [3,48,58].

2.3. Data Collection

To analyze change over time in the campus urban forest, we used three data sources: (1) inventories from 2003 and 2014, (2) institutional records of trees removed due to capital projects, and (3) firsthand accounts of the University Landscape Architect (a co-author on this manuscript), who has managed campus trees since 1994.

2.3.1. Tree Inventories

The campus inventories included plants that are considered either small trees or large shrubs (e.g., *Amelanchier* spp., *Hamamelis* spp., *Lagerstroemia* spp.) as there is no standard botanical definition of 'tree' [59] and all these woody plants fall under the purview of FRES staff. We refer to all plants in the campus tree inventories as 'trees' throughout this paper because these plants are managed as part of the campus urban forest.

The 2003 and 2014 inventories were conducted by Morris Arboretum's Urban Forestry Consultants. In 2003, tree species and circumference at breast height (CBH, to the nearest 2.54 cm) were collected on paper and transcribed to a spreadsheet. We converted CBH to diameter at breast height (DBH), taking the quadratic sum for multi-stemmed trees [60]. Tree locations were recorded based on a grid system for the campus that mirrors the city street grid (e.g., grid cell M38 for the area bordered by Walnut St. and Locust Walk, 38th and 39th Sts.). Within each grid cell, trees were located within quadrants and assigned a number (e.g., grid cell M38, quadrant 1, tree 20). The 2003 inventory was later paired with a spatially explicit map, based on a supplemental inventory collected in 2005, and entered into AutoCAD [61], which is a mapping software widely used in landscape design. The grid cell and tree number connected the AutoCAD map to the inventory spreadsheet. Tree planting site type (i.e., a description of the tree's immediate location, [62]) was categorized as street tree, other hardscape, or lawn, based on the AutoCAD map and the current BG-BASE map (see description below). Street trees are in sidewalk cutouts or trenches while other hardscape trees are in plazas or courtyards.

For the 2014 inventory, tree data was collected and uploaded into BG-BASE, a software system for vegetation records, which is used in botanic gardens, arboreta, and herbaria [63]. Data was collected on tablets, through which field crews confirmed tree locations while looking at BG-MAPS, which are CAD-based maps. All trees were given an accession number; trees that were new in the database as of 2014 only had this accession number, whereas trees that were also present in the 2003 inventory retained the grid cell and tree number while also receiving an accession number. The mortality of trees that were alive in 2003 was determined based on removals noted in the 2014 inventory, and removed and standing dead trees noted in intervening inventories in 2005 and 2012. Neither the 2005 nor the 2012 inventory covered the entire core campus area that was of interest to this study; therefore, we only analyzed the 11-year mortality from 2003 to 2014 because these inventories were the most spatially complete. We did not assess trees that were planted post-2003 and died pre-2014, also known as "ghost mortalities", as they take place in between monitoring end points [11]. Incomplete DBH records from 2014 precluded an assessment of DBH growth or basal area changes among campus trees between the 2003 and 2014 inventories.

We linked the inventories using a compound key based on the grid cell and tree number (e.g., M38.1_20). We screened the records linked in this manner for mismatches in species and decreases in DBH, which could indicate that a tree number was re-used for a new planting (i.e., the number did not retire following tree removal). For trees that did not link using the compound key, we connected inventory records by manually examining the AutoCAD and BG-BASE maps in conjunction with species and DBH records, and discussions with the University Landscape Architect.

2.3.2. Categorizing Causes of Removal from Institutional Records and Firsthand Accounts

We documented causes of tree removal between the 2003 and 2014 inventories. In the context of this managed campus landscape, we refer to 'causes' as the rationale for removal, as reported by the campus tree managers and determined through institutional records. Crucially, for trees that were removed due to declining health or risk management, Penn landscape staff generally removed trees while they were still alive to maintain an attractive campus with low tree-related risk. In general, during our study period, campus trees were not allowed to complete the mortality spiral and die in-place [17,18,24]. Furthermore, because of proactive management, in the rare instances of trees that may have died in-place, it is unlikely that standing dead trees would persist in this campus landscape, in contrast to neighborhoods where standing dead trees may linger for years due to lack of stewardship [11,21]. Therefore, we refer to mortality cases in terms of removal rationales, with the assumption that nearly all removals were of living (although sometimes unhealthy) trees.

The causes of removal were grouped by:

(1) *Removals due to tree health decline or risk management*, including storm and wind damage, pests and diseases, shading from trees or structures, tree risk management (i.e., removal of hazardous trees that could damage built infrastructure or injure people), street tree stresses (e.g., de-icing road salt), vehicular accidents, and trunk girdling (e.g., from signs or decorative lights);
(2) *Human land use choices*, including capital projects, other construction (e.g., building façade and walkway upgrades, utility repairs), and aesthetic and functional landscape preferences (e.g., trees interfere with social landscape uses or visual site lines); and
(3) *Unknown*.

At Penn, capital projects are currently defined as major renovation or construction activities costing US$100,000 or more, such as the construction of new buildings or landscapes, or major renovations of existing built structures [64]. Capital project maps at the campus archives include records of tree removals. Collectively, we consider capital projects and other construction activities to be akin to renovation and redevelopment activities that occur throughout urbanized landscapes. For tree losses that could not be directly linked to capital project records, causes of removal were categorized based on the recollections of the University Landscape Architect. Specifically, campus maps, removal data, and capital projects records were discussed at-length during in-person meetings to categorize the causes of removal.

Although there were undoubtedly intersecting impacts of human and biophysical factors on the health and survival of trees in removal groups (1) and (2) that may have predisposed them to decline and removal [24], the removal causes we analyzed were the final contributing factors [17] from the perspective of the University Landscape Architect, i.e., the justification or impetus for tree removal. We use the terms contributing and predisposing following Manion's seminal disease decline spiral [17]. For removals due to tree health decline or risk management (group (1)), these removals were considered necessary by arboricultural professionals, such as trees suffering from abiotic or biotic stressors, structural stability concerns, or safety concerns for pedestrians and infrastructure. The motivations for removal often blended together tree health and safety concerns, particularly for large trees, because declining health can make such trees hazardous. Some trees in group (1) may have been predisposed to poor health due to prior human decisions, but our cause of removal classification indicates the final contributing factor to tree removal, and not the predisposing or inciting factors that made the tree more vulnerable. For instance, a tree stressed from excessive winds, and subsequently removed due to this health decline, was deemed part of removal group (1), even though species selection and site placement set the stage for those wind problems, as predisposing factors. For removals due to human land use choices (group (2)), the removals were not directly caused by judgements about tree health and safety, and instead reflect human priorities for landscape functions.

2.3.3. New Trees

New trees in 2014 consisted of trees that were planted between 2003 and 2014, and also survived to 2014. We specifically noted the number of trees that were planted as part of capital projects. However, we were unable to obtain comprehensive records about tree preservation during capital projects for our study period.

2.4. Overview of Campus Tree Preservation and Governance Context

Because tree preservation and planting policies can restrict tree removals in cities and/or promote new tree planting [35,41,65], we briefly explain the governance context about tree preservation in Philadelphia and on the Penn campus. There is extensive oversight about campus tree removal and planting decisions by the University Landscape Architect and Landscape Planner. These professionals have control over removals, plantings, and replacements in areas of the campus that are not active capital projects, and they also have influence regarding tree removals and additions spurred by capital projects. With capital projects, the trees intended for removal and planting are first proposed by a contracted landscape architect associated with a given project, then subsequently reviewed by the Office of the University Architect, and sometimes adjusted by the University Landscape Architect and Landscape Planner. These campus staff attempt to retain large healthy trees when it is practical to do so (e.g., when a tree is not directly located within the footprint of a new building). However, there is not a comprehensive campus tree policy, notwithstanding the campus plans mentioning the importance of trees [49,57] and the existence of tree management plans for particular taxa (e.g., regarding diseases and pests of *Ulmus* spp. or *Fraxinus* spp.). In other words, during our study period, campus tree removals and plantings were not tightly regulated through a formal codified process but were closely managed by landscape and arboricultural professionals.

Furthermore, campus capital projects whose parcel's land area was greater than 464.5 m^2 fall under Philadelphia's zoning code (Philadelphia Municipal Code §14-705) concerning development standards for landscapes and trees. In 2012, new municipal policies were established to promote tree preservation and planting during development [66,67]. Specifically, the policies were intended to curtail the removal of heritage trees, defined in the code as trees that are both over 61 cm DBH and on a select species list (mostly natives). However, tree preservation is not strictly required by this municipal policy. Rather, tree removal is allowed when the tree is located in the intended building footprint, if "a certified arborist has determined that the tree is dead, damaged, diseased, or a threat to public health or safety", or if the tree is considered problematic for traffic safety (Philadelphia Municipal Code §14-705). Removed trees are required to be replaced, based on a system of credits in relation to the size of preserved and removed trees on the parcel. These policies would not have impacted capital projects on the Penn campus that were planned and approved prior to 2012 (even if trees were removed post-2012) but did impact some projects that took place in late 2013 and early 2014, towards the end of our study period.

Notably, street trees in Philadelphia fall under the purview of Philadelphia Parks & Recreation (PPR), which gives permits for street tree planting and removal actions [68]. PPR also has a list of permitted street tree species. Penn FRES staff have assumed management responsibility for street trees within the core campus, including funding and overseeing planting, maintenance, and removal. Species planting choices for private landscapes (i.e., most trees on the Penn campus) are not tightly regulated by the municipality.

2.5. Data Analysis

2.5.1. Tree Demography

Our analysis of campus tree population changes followed a previous street tree mortality study [11] and urban tree demography primer [8], which in turn adapted classic demographic methods [69] to urban forests. We defined survival as trees observed to be alive in 2014 that were also alive in 2003, and mortality as trees that were standing dead or removed by 2014 that were alive in 2003. We calculated the annual mortality rate, q_{annual}, as:

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

where x is the time interval (11 years), K_0 is the number of trees alive for the baseline inventory (2003), and K_x is the number of trees alive at the end of x years (2014). We report annual mortality rates for all living trees from 2003, and by DBH size class groups from that baseline inventory (<15.3, 15.3–30.5, 30.6–45.7, 45.8–61.0, and >61.0 cm) and site type (street tree, other hardscape, and lawn). Analyses using size class considered only trees with known DBH from the 2003 inventory.

We also report the overall population size in 2003 and 2014, and the raw counts of trees that died or were removed (mortality) and newly planted trees (recruitment). Notably, while ecologists use 'population' to refer to individuals from the same species, urban foresters have used 'population' more generally to refer to all trees within a geographic area or landscape type [8].

2.5.2. Association between Size Class, Site Type, and Mortality

Because size class and site type have been found to be statistically associated with mortality outcomes in other urban forestry studies [24], we assessed the association between DBH size class and mortality, and between site type and mortality, using χ^2 tests and odds ratios. We used the 'epitools' package in R [70].

2.5.3. Compositional Changes and DBH of Removed Trees

We report the species richness and proportions of common species and genera. The styling and format of species records varied, as sometimes only common names were used, so data processing included the assignment of Latin names to each record. Nearly all trees were identified to at least the genus level (99.97% in 2003 and 99.81% in 2014), but around one-fifth of trees did not have species-level identification (19.2% of genus-level identifications lacked species identifications in 2003). Our assessment of compositional change focused on genera and species constituting over 2% of all trees in either 2003 or 2014. When we report species richness counts, we include only individual trees identified to species or hybrids (e.g., *Prunus* × *incam* and *Prunus serrulata* would count as identified species, but *Prunus* sp. would not). We also report basic summary statistics regarding the size of removed trees (median and mean DBH of removed trees).

3. Results

3.1. Annual Mortality and Net Population Change

Within the core campus, the total number of live trees remained the same: 3694 in both 2003 and 2014. There were 1427 trees that were removed during this period, meaning that over one-third (38.5%) of the live trees from 2003 were removed over the full 11-year study period. The annual mortality rate was 4.3%. The number of newly planted trees (1427) exactly equaled the number of trees lost.

3.2. Causes of Removal

Of the 1427 trees lost, 50.4% were caused by human land use choices, 48.7% were caused by tree health decline and risk management, and 0.9% of removals had unknown causes (Table 1). The single largest causal subgroup of tree loss was construction from capital projects (29.9% of all removals). Another 18.6% of removals were due to construction unrelated to a capital project; so, capital projects and other construction combined accounted for 48.5% of all removals. Examples of capital projects and construction activities include major renovations to dormitories and academic buildings (e.g., Harrison College House, Steinberg-Dietrich Hall; see Figure 1), new academic buildings and dormitories (e.g., Skirkanich Hall, McNeil Center for Early American Studies, New College House), the installation of manicured parks (e.g., Shoemaker Green, Edward W. Kane Park), improvements to walkways and building façades, and sculpture installation. The largest loss of trees from a single capital project was New College House (now known as Lauder College House) on Hill Field (85 removals). The second-largest loss of trees from a capital project was the Annenberg Public Policy Center (51 removals). Removals due to aesthetic and functional landscape preferences (2.0% of all removals) included trees that conflicted with a high-traffic taxi stop, interfered with visual sight lines for security cameras, were deemed to be in an overly dense cluster of other trees, and misaligned with landscape uses and preferences of university administrators.

Table 1. Causes of tree removal (n = 1427) on the Penn campus, 2003–2014, and the median DBH of those removals (for the subset of 1407 removed trees that had DBH values).

Cause	n	% of Total	Median DBH of Removals (cm)
human land use choices	719	50.4	14.6
capital projects	426	29.9	12.9
other construction	265	18.6	16.1
aesthetic and landscape preferences	28	2.0	12.5
tree health decline and risk management	695	48.7	10.5
unspecified general health decline	413	28.9	12.1
shading	115	8.1	4.9
street tree stresses	73	5.1	12.1
pests and diseases	37	2.6	8.9
storm damage	24	1.7	10.9
wind damage	20	1.4	8.9
tree risk management	8	0.6	30.3
accidents	4	0.3	8.9
girdling	1	0.1	26.7
unknown	13	0.9	13.7
all	1427		12.1

Within removals due to tree health decline and risk management, most were unspecified general health declines (Table 1), such as newly planted trees that failed to establish, or declining old trees. In terms of specific causes of tree health problems that the University Landscape Architect could identify, shading was the most commonly identified stressor (8.1% of all removals). Examples of trees that were unhealthy and removed due to light limitation include 43 *Magnolia* spp., 24 *Cornus* spp., and 18 *Prunus* spp. (out of 115 shade-related removals). For street trees, stresses common to that site type (including road salt) resulted in 5.1% of all removals, although street trees were also removed due to other stressors discussed here (e.g., disease). Trees that were determined to be unhealthy due to wind stress, and subsequently removed (1.4% of all removals), were mostly located in the "superblock" of high-rise dormitories near 40th St. and Locust Walk, which is an area well-known to Penn students and staff for excessive winds [71]. Removals due to pests and disease (2.6% of all removals) were mostly *Fraxinus* spp. (23 out of 37 removals due to pests and diseases), which often had white peach scale (*Pseudaulacaspis pentagona*). Notably, our study took place before the confirmed arrival of the emerald ash borer (*Agrilus planipennis*) in Philadelphia in 2016, which is fatal to *Fraxinus* spp. if left untreated. Very few trees (0.6%) were removed primarily due to safety and infrastructure concerns related to tree risk management. However, as mentioned previously, risk management factored into removal decisions for any large trees that had declining health. Rarely, vehicular accidents damaged trees and caused removals along streets and walkways (0.3% of all removals). In one instance, a tree suffered health decline due to girdling decorative lights (0.1% of all removals). A summary of the mortality rates and common causes of removal for common genera is provided in Table S1.

3.3. Plantings Associated with Capital Projects

Of the 1427 new trees present in 2014, 402 were from capital projects (28.2%). The largest number of new trees from a capital project was 96 trees planted at Shoemaker Green, at which a new greenspace was installed to replace aging tennis courts and parking areas [72]; 38 trees were also removed due to the installation of Shoemaker Green. These removals at Shoemaker Green included two relatively large, old *Platanus* × *hispanica* (59.8 cm in 2003), which will be explained further in the discussion section.

3.4. Association between Size Class, Site Type, and Mortality

Considering the 3657 trees with known DBH from the 2003 inventory, smaller trees (<15.3 cm) had the highest annual mortality, 4.7%, while the trees in the largest size class (>61.0 cm) had the lowest annual mortality, 2.7% (Figure 2, Table 2). The smallest and second-smallest size classes both had a significantly different mortality from the three largest size class groups (Figure 2). The estimated odds ratios indicate a greater probability of mortality for the two smallest size classes (Table S2). For example, the estimated odds ratio for mortality, comparing trees < 15.3 cm vs. trees > 61.0 cm, was 2.0 (95% CI: 1.4–3.1). Notably, capital projects and construction were relatively less frequent causes of removal for the largest size class (35.3% of removals for trees >61.0 cm) but a very common cause of removal for the second-highest size class (70.0% of removals for trees 45.8-60.9 cm) (Table 2).

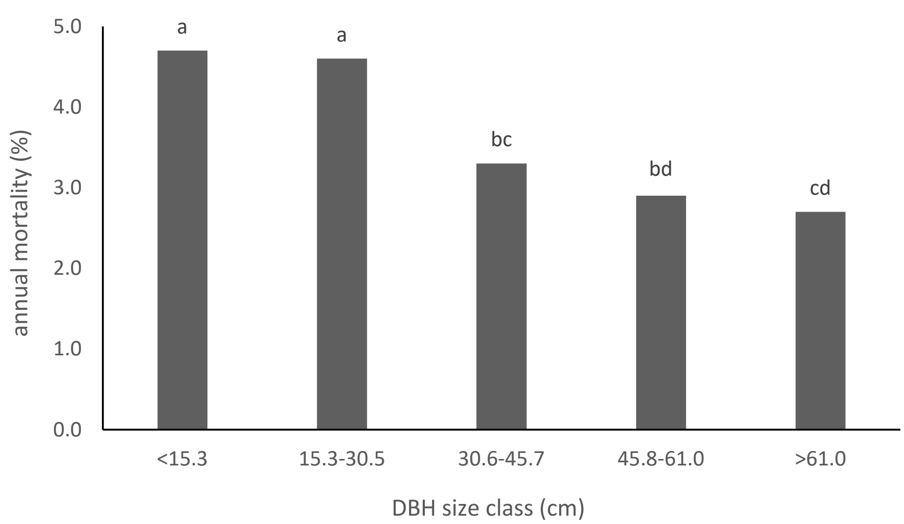

Figure 2. Annual mortality rates by DBH size class. Classes with the same letter are not significantly different.

Table 2. Annual mortality rates by DBH size class (for the subset of 3657 trees in 2003 that had DBH values).

DBH Range (cm)	Number Alive in 2003 (K_0)	% of 2003 Inventory	% Annual Mortality Rate (q_{annual})	% Removals Due to Capital Projects and Construction
<15.3	1981	54.2	4.7	42.9
15.3–30.5	879	24.0	4.6	54.8
30.6–45.7	449	12.3	3.3	65.2
45.8–61.0	216	5.9	2.9	70.0
>61.0	132	3.6	2.7	35.3

In terms of site type, annual tree mortality was highest for street trees, at 5.4%, which was significantly higher than mortality for lawn trees, at 4.2%, and both street and lawn trees had significantly higher mortality than other hardscape trees, at 1.3% (Table 3). The estimated odds ratio comparing street and lawn trees was 1.4 (95% CI: 1.2–1.7) (Table S3). Notably, capital projects and construction were the cause of removal for 53.6% of lawn trees but only 25.4% of street trees (Table 3).

Table 3. Annual mortality rates by site type (for all 3694 trees in 2003).

Site Type	Number Alive in 2003 (K_0)	% of 2003 Inventory	% Annual Mortality Rate (q_{annual})	% Removals Due to Capital Projects and Construction
street tree	558	15.1	5.4	25.4
other hardscape	76	2.1	1.3	40.0
lawn	3060	82.9	4.2	53.6

3.5. DBH of Removed Trees

The mean DBH of removed trees (for the 1407 removed trees that had 2003 DBH data) was 17.7 cm (standard deviation 15.6 cm), with a median value of 12.1 cm. The mean DBH of the surviving trees (for the 2250 surviving trees that had 2003 DBH data) was 21.0 (standard deviation 18.4), with a median value of 14.6 cm.

Certain taxa had particularly large trees removed (Tables 4 and 5). For instance, the median DBH of removed *Tilia* spp. was 30.7 cm. Trees removed due to human land use choices tended to be larger in general, although the handful of trees removed primarily due to risk management were quite large (Table 1). Trees removed due to shading tended to be small trees located under the canopies of other trees.

Table 4. Most common species in 2003 and 2014. Only species representing 2% or more of the total in either inventory are included, and trees identified to the genus level only (e.g., *Acer* sp., *Prunus* sp.) are not included here. These 12 species account for 44.4% of all trees in 2003 and 41.9% in 2014.

Species	% of 2003 Inventory	% of 2014 Inventory	Change	No. Removals	Median DBH of Removals (cm)	No. Plantings
Acer rubrum	3.9	3.7	−0.2	38	20.7	30
Acer saccharum	1.9	2.0	+0.1	30	12.9	34
Cornus kousa	2.5	1.8	−0.7	40	4.0	14
Gelditsia triacanthos	6.8	7.8	+1.0	81	23.4	117
Liquidambar styraciflua	2.2	2.2	0.0	26	19.8	25
Liriodendron tulipifera	2.6	2.1	−0.5	58	8.1	40
Magnolia virginiana	6.1	6.1	0.0	73	4.2	72
Platanus × hispanica	7.9	6.2	−1.7	94	19.4	29
Quercus phellos	2.8	3.3	+0.5	25	31.5	45

Quercus rubra	2.1	1.7	−0.4	23	28.3	7
Tilia cordata	2.4	1.7	−0.7	43	30.7	16
Zelkova serrata	3.2	3.3	+0.1	30	10.1	32

Table 5. Most common genera in 2003 and 2014. Only genera representing 2% or more of the total in each inventory are included. These 18 genera account for 79.8% of all trees in 2003 and 78.4% in 2014.

Genus	% of 2003 Inventory	% of 2014 Inventory	Change	No. Removals	Median DBH of Removals (cm)	No. Plantings
Acer	7.4	6.7	−0.7	103	19.4	77
Amalanchier	1.8	6.1	+4.3	27	7.7	189
Betula	2.2	2.4	+0.2	60	12.0	67
Carpinus	1.4	2.1	+0.7	14	9.7	37
Cornus	4.2	4.5	+0.3	65	6.6	77
Fraxinus	2.7	1.3	−1.4	60	11.7	6
Gleditsia	6.8	7.8	+1.0	81	23.4	117
Liquidambar	2.2	2.2	0.0	26	19.8	25
Liriodendron	2.6	2.1	−0.5	58	8.1	40
Magnolia	10.3	9.8	−0.5	133	7.3	114
Malus	3.6	2.5	−1.1	65	20.2	26
Platanus	8.2	6.4	−1.8	97	19.4	30
Prunus	4.2	4.3	+0.1	71	13.2	76
Quercus	10.2	9.5	−0.7	136	13.7	109
Tilia	3.3	2.3	−1.0	56	30.7	21
Ulmus	2.6	3.0	+0.4	20	27.1	34
Zelkova	3.9	3.6	−0.3	45	8.1	33

3.6. Compositional Changes

The campus had high taxonomic richness, with 77 identified genera and 136 identified species in 2003, and 72 genera and 143 species in 2014. A few taxa had very few individuals in 2003, all later removed (e.g., two *Abies* sp.), and no new plantings of that taxa; these are the taxonomic groups that were present in 2003 but not 2014.

Tables 4 and 5 show the species and genera that account for 2% or more of all trees in either 2003 or 2014, including the number of trees planted and removed. The most common species in 2003 was *P.* × *hispanica* while the most common species in 2014 was *Gleditsia triacanthos*. In terms of genera, the most common in both 2003 and 2014 was *Magnolia* spp. The largest increase for a genus was *Amelanchier* spp., which was only 1.8% of all trees in 2003 but 6.1% in 2014. Meanwhile, there were decreases in the proportion of campus trees for *Fraxinus* spp., *Malus* spp., and *Platanus* spp.

4. Discussion

Annual tree mortality on the Penn campus (4.3%) is relatively high in comparison to past studies. Based on a recent literature review by Hilbert et al. [24], the median mortality for 18 repeated inventories of mixed-age urban trees is 2.3–2.6%, and the third quartile is 3.0–3.3%, meaning that the mortality rate found in our study is considerably higher than over 75% of past reported mortality rates. That literature review [24] spanned a range of medium to large cities, and a variety of site types and land uses within cities. In urban tree mortality studies of mixed-age trees that reported mortality rates separately by land use, higher mortality tended to be in transportation, commercial, open park, and high-density residential areas [73,74], aligning with the landscape conditions on the Penn campus. We posit that two primary drivers were responsible for the relatively high mortality rate on this campus: construction pressure and proactive management, with both drivers leading to substantial tree removals. In Table 1, removals due to construction pressure are represented by capital projects and other construction while proactive inspection cycles are tied to most removals in the tree health decline and risk management causal group.

While construction and development activities were only cited as factors for mortality in 15% of the quantitative mortality studies in Hilbert et al. [24], construction may play a far larger role than this low percent suggests, particularly in urban cores [39]. On the Penn campus, capital projects and other construction activities consisted of new buildings and smaller-scale renovations to buildings, walkways, and utilities. The impacts of construction on this college campus in Philadelphia reflect broader processes in urban forest systems. For instance, building demolition following an earthquake was a driver of canopy loss in Christchurch, New Zealand [37], building permits were associated with tree mortality in Toronto, Canada [36], building and road construction were related to declining tree density in Mississauga, Canada [75], and major development projects caused tree removal and canopy loss in Melbourne, Australia [39]. More broadly, construction activities in urbanized landscapes are related to the phenomena of redevelopment, renovation, densification, and infill, which occur on scales from single parcels or streetscapes to entire neighborhoods, and these processes impact trees and greenspaces in a variety of ways [38,58,76–78]. Trees in urban settings are routinely removed to make way for new buildings, roads, parking lots, and other structures; to expand the physical footprint of such structures; or to upgrade critical built infrastructure systems, such as utilities [39,65,76].

Indeed, many urban forest managers recognize the conflicts between promoting tree canopy and construction pressure [79,80], and municipal tree ordinances and by-laws can restrict or compensate for tree losses during redevelopment and construction [35,41,81,82], and can require the preservation of tree rooting areas near construction zones [83]. During most of our study period, tree removals on the Penn campus were not tightly regulated by formal policies, but campus landscape personnel nonetheless enacted strong oversight, retaining trees when feasible, and regularly replacing removed trees. In fact, the number of trees removed during our study period exactly matched the number of trees planted, representing substantial turnover of the urban tree community [84]. This precise balancing of planting and removals emerged from a series of decisions about numerous capital projects and other sites on campus. Indeed, 38.5% of the 2003 inventory was removed over the entire 11-year period, and the same amount planted (albeit not necessarily at the same planting sites), representing high turnover. Our data reflects the dynamism of urban forest systems, as trees and greenspaces are frequently added to and removed from these systems due to human interests and land use changes [9,11,39,84–86]. Construction activities can lead to both tree removal and plantings, making construction-related processes and data streams critical for improved understandings of urban tree demography and shifting diversity [8,87]. Municipal or campus policies regarding tree preservation, removals, and replacements could potentially lower rates of tree loss and curtail construction impacts [41,65,88]. For instance, Washington, DC has a strict tree law, which bans the removal of healthy trees above a certain size, with developers fined for unpermitted removals [89]. Some cities also require tree replacement following permitted removal on private properties, although a recent study from Toronto demonstrated that 30% of residents did not comply with this requirement [35]. Inter-city comparisons of tree removal and preservation policies and enforcement, and associated removal and replacement data, are needed to gauge the impacts of such policies on the rates of tree loss.

Not all human land use decisions that impacted tree removals on the Penn campus involved construction. A few removals occurred due to interference with road signage or security cameras or aesthetic preferences. Indeed, the conflict between trees and road visibility is recognized as an ecosystem disservice [22,90]. These removals on the Penn campus also align with residential tree removal decisions in Mississauga, Ontario, Canada, in which aesthetics were sometimes the rationale for removal [33]. However, the most commonly cited reasons for removal in that Mississauga study, and with a survey of municipal forestry programs across the US [88], were poor tree health and risk concerns for property or people. This national finding aligns with the 48.7% of removals on the Penn campus in the health declines and risk management category. Deciding whether to remove an unhealthy (but still living) tree is subjective, even for arborists [91]. In our study, it appears that a relatively low tolerance for risk, and a low willingness to accept trees that are not at peak health and appearance produced more removals. As we articulated in the introduction, the removal of unhealthy but not yet dead trees is a distinctly urban process of 'tree euthanasia' [26]. Such removals can signal high levels of maintenance, whereas an abundance of standing dead trees can indicate lack of stewardship [21,22].

Crucially, the act of tree removal is expensive, especially for large trees, and removal decision-making in urban forestry depends on routine monitoring and inspections [11,88,92]. A lack of financial resources may lead to the retention of undesired trees, even standing dead or dying trees, by some residents and municipalities [11,33]. The financial costs of removal were less of a burden on the Penn campus, as this campus represents an elite Ivy League institution, whose picturesque campus evokes the prestige of the university [48]. Campus tree management also benefits from the involvement of tree and landscape professionals, including the University Landscape Architect and Landscape Planner, Urban Park Manager, and a Board-Certified Master Arborist at the Morris Arboretum. Around one-quarter of the new trees were planted as part of capital projects, but the rest were planted at the discretion of these professionals. The fact that the number of new trees planted equaled the number of removed trees is indicative of the high levels of oversight given to the campus urban forest. Indeed, the proactive management that takes place on the Penn campus today—with routine inspections to enable prompt management decisions—is far from universal for cities. In a recent survey of municipal tree care in the US, only 55% of communities reported having proactive systematic management, with only 33% for large cities with populations over 1 million [88]. Many US towns and cities primarily carry out reactive management that responds to the latest emergency, such as pest outbreaks and storms, rather than proactive management to better prepare for such crises [88]. Municipal foresters generally recognize that frequent inspections to identify, treat, prune, or remove unhealthy, declining, or dead trees constitute best practices to sustain this natural resource and protect public safety [93,94] but are sometimes unable to do so due to limited budgets and capacity. Proactive monitoring and care require substantial investment of funds and labor [92,93].

The relatively high annual mortality rates on the Penn campus, at an institution with substantial wealth and arboricultural expertise, may seem to contradict past research that associated higher mortality and tree canopy loss with areas of lower income and educational attainment [24,95]. However, evidence linking socioeconomic status and mortality have been mixed. For example, in Sacramento, California, planted yard trees in areas of middle income had the highest survival, with trees in both lower- and higher-income areas having worse survival [96]. Further research is needed to disentangle the tree care mechanisms connecting wealth to tree removal and death in a variety of urban settings within and across cities, explicitly recognizing the financial realities that enable or constrain tree removal on residential and institutional lands. For this urban college campus, we postulate that the influx of funding for capital projects, combined with frequent tree inspections and a low tolerance for unhealthy trees, produced the relatively high mortality observed. In short, more management translated to a high mortality rate. Crucially, this relatively high mortality is not necessarily, in and of itself, a detriment to the campus urban forest. As long as replacement plantings continue apace, and the campus continues to support mature and large-stature trees, there can be sustainable numbers of trees and levels of canopy cover and basal area over many years [10]. Furthermore, the current campus tree cover and tree count may be at or close to maximal levels, in terms of supporting trees while also maintaining other landscape uses [48]. Investigations in other elite campuses and urban gardens would reveal whether the high mortality rate we observed is typical of other well-funded urban landscapes.

The sustainable management of urban forests extends beyond removals and replacements. Taxonomic diversity and composition are also critical to consider, particularly to avoid overloading urban forests with monocultures that are vulnerable to invasive pests and diseases [97]. The Penn campus had high taxonomic richness, with over 70 genera, and only 2 genera constituting more than 10% of all trees in 2003, and no species representing more than 10% in either inventory. This exceeds diversification guidelines that are commonplace in urban forest management [97].

The declining portions of *Acer* spp., *Fraxinus* spp., *Malus* spp., *Platanus* spp., *Tilia* spp., and *Quercus* spp., and the rise of others such as *Amelanchier* spp. and *G. triacanthos* show that the dominant taxa on campus are slowly shifting. The campus tree managers make a concerted effort to promote taxonomic diversity, including in negotiations with landscape architects designing capital projects, who may heavily weigh their designs towards a few species. Other studies have noted that species go in and out of favor over time due to fads among tree professionals [3,98]; for example, at present, *Amelanchier* spp., *Betula* spp., *G. triacanthos*, and *Magnolia* spp. are popular among the landscape architects hired to construct capital project designs on this campus. Caution is warranted with respect to the decline in large-stature taxa such as *Acer* spp., *Platanus* spp., and *Quercus* spp. In the northeastern region of the US, municipal streetscapes are currently being planted with more small-stature species in comparison to past practices, which prioritized large-stature shade trees [60,99]. This change has occurred, in part, due to the emphasis in modern urban forestry on planting small-stature species under power lines and in restricted growing spaces to avoid the potential for disservices from infrastructure conflicts [22]. Yet, the planting and retention of large trees is important, especially given that large trees produce the most environmental benefits, such as energy reduction through shading and carbon sequestration [100]. For instance, when Shoemaker Green was installed, six large *P.* × *hispanica*, which had been planted in the mid-20th century, were preserved through the advocacy of campus tree managers, even though the initial site plans for this capital project called for the removal of these trees [56]. Two other *P.* × *hispanica* at Shoemaker Green were removed to promote better sight lines to the stadium next to this campus greenspace. This example points to the strong influence of campus tree experts in promoting tree retention, and the ways in which aesthetic considerations factor into the removal decisions for capital projects. Additionally, the species decisions by the University Landscape Architect and Landscape Planner continue to be influenced by the 1977 Landscape Development Plan, which suggested specific taxa for various spaces on campus, such as entryways, walkways, small plazas, and larger greenspaces [49]. The present composition and richness on campus, and the slow shifts in composition we observed represent the accumulated impacts of numerous individual site decisions, as various tree and landscape professionals leave legacies on the landscape [3,48].

Currently, management plans for specific taxa threatened by pests and diseases (e.g., *Fraxinus* spp., *Ulmus* spp.), developed with the Morris Arboretum's Urban Forestry Consultants, strategize for the treatment of these trees to encourage the continued provisioning of benefits. For example, tremendous expertise and expense has been expended in maintaining one particular *U. americana* that was planted in 1896 and propagated from the *U. americana* under which William Penn, the founder of Philadelphia and Pennsylvania, purportedly signed a treaty with the Lenni Lenape tribe of Native Americans [48]. Campus tree professionals' ongoing efforts to preserve this tree include frequent inspections and regular treatments to prevent Dutch elm disease (*Ophiostoma* spp.) [101]. This management approach is rooted in an appreciation for the tree's large size and associated regulating ecosystem services, alongside its cultural significance. Future research on tree retention decisions should investigate motivations ranging from environmental to social benefits and cultural symbolism, and arboricultural expertise, financial resources, and policy structures related to those decisions.

Like many urban forest systems [11,39,73], the Penn campus was dominated by small trees, with approximately three-quarters of the trees being under 30.5 cm DBH in 2003

(Table 2). Annual mortality was higher for these small trees and there was a Type III mortality curve based on size class (i.e., mortality is highest for the smallest trees, then dropping off substantially for mid-sized and large trees), aligning with past studies on the mortality of both urban and wildland trees [8,24,102]. Very small, recently planted trees can have high mortality as they fail to establish, with drivers such as lack of maintenance and poor site quality [21,24,103]. However, on the Penn campus, new trees are routinely irrigated by landscape contractors for 1-2 years post-planting, so insufficient young tree care was not a pronounced problem in this study system. Furthermore, the higher mortality of trees under 30.5 cm on this campus includes juvenile trees that are past the establishment years. Small and juvenile trees may be viewed as expendable, whether to make space for a capital project, or to cut down unhealthy young trees and start fresh with new plantings. When campus tree professionals strongly advocate for the preservation of particular trees, they usually focus on very large trees (e.g., the *P.* × *hispanica* previously mentioned at Shoemaker Green), just as municipal tree preservation laws generally focus only on large trees [41,89]. Indeed, for trees over 61 cm that were removed, relatively fewer (35.3%) of these removals were due to capital projects (Table 2), possibly related to the fact that Philadelphia's heritage tree policies apply to trees over this size threshold. Yet, capital projects were responsible for a much higher portion of tree removals in the next-largest size class (70.0% of removals for trees 45.8–60.9 cm). Mid-sized urban trees may, therefore, be particularly vulnerable to construction pressure. In acknowledgement of the importance of preserving mid-sized trees, there is a potential recommendation in Philadelphia's forthcoming urban forest strategic plan to lower the heritage tree threshold from 61 to 46 cm (E. Smith Fichman, pers. comm., 2 March 2022). The relatively higher portion of lawn trees that were removed due to construction on the Penn campus (Table 3) also suggests the importance of understanding heritage tree law's impacts on private properties, including institutional greenspaces.

Limitations of this research include the relatively narrow focus on a single, elite college campus in one city, and the lack of complete and comparable DBH data from 2014. Analyses of species-specific growth rates and basal area changes would be possible with repeated inventories that longitudinally track individual trees and include careful remeasurements of DBH [62]. However, since urban tree inventories, such as this campus inventory, are more typically conducted with the goal of managing tree care workflows, these data may not be conducive to comprehensive analyses of change over time (e.g., mortality, growth) [59]. For instance, in our study, linking the data tree-by-tree across two separate inventory software systems involved substantial personnel time for database management.

While the specific trends in the tree population and community dynamics on a college campus will not be precisely the same as those in an entire city or metropolitan region, this campus case study nonetheless provides insights into the drivers of urban forest change over time. The highly urbanized nature of this campus, and the ways in which campus tree management has echoed larger trends in urban forestry [48], make our investigation a compelling case study to spur further research. Our approach demonstrates the value of mixed-methods and transdisciplinary approaches, blending field data with institutional records and firsthand accounts to improve understandings of urban tree removal. Studies that rely on field observations and statistical correlations among variables predicting mortality overlook the important component of human decision-making as a direct basis of removal. For instance, with our field data, if we had only quantitative analysis of DBH and site type to make inferences about the removal, we would not have illuminated the crucial role of construction and proactive management. Given that tree removal in cities is fundamentally linked to decision-making, more research is needed to take advantage of institutional records, and to ask tree professionals and other land managers about their justifications for removals. Municipal and consulting arborists sometimes use urban tree inventory software to record when removal is needed for individual trees, as part of larger efforts for proactive management and limiting risk [39,88]. These arborists could also note their justification for each removal, following the categories used in our campus study, to quantify the portion of trees removed for different reasons. This would be akin to research in natural forest settings that documented the cause of death for trees in the Sierra Nevada area of California [104]. For urban trees, interviews and focus groups regarding the removal decision-making process would yield qualitative insights into the intersecting factors that lead to removal, and the relative influence that different stakeholder groups have on removal decisions. Additionally, geospatial analyses of urban canopy cover change in relation to redevelopment or construction records [36,39] would produce insights into the association between tree loss and construction across neighborhoods.

Supplementary Materials: The following supporting information can be downloaded at: www.mdpi.com/article/10.3390/f13060871/s1, Table S1: For genera representing 2% or more of the total inventory in either 2003 or 2014, we show here: annual mortality, most common causes of removal, the number of identified known species within the genus, and the most common species within the genus. The number of species within each genus reflects species (or hybrids) present in 2003 and/or 2014. For the two most common species within each genus, we also report the portion a given species constituted of that genus in 2003; Table S2: Association between 2003 size class and mortality (n = 3657), with results for a $\chi 2$ test and odds ratio comparison of each size class pairing. Significant outcomes ($p < 0.05$) are indicated in bold. Estimated odds ratios over 1 indicate that mortality is more likely in the first class in the pairing; Table S3: Association between 2003 site type and mortality (n= 3694), with results for a $\chi 2$ test and odds ratio comparison of each pairing. Significant outcomes ($p < 0.05$) are indicated in bold. Estimated odds ratios over 1 indicate that mortality is more likely in the first class in the pairing.

Author Contributions: Conceptualization, L.A.R. and J.P.F.; Methodology—study design, L.A.R. and J.P.F.; Methodology—linking inventories and cause of removal coding, J.P.F., R.E.L. and C.E.C.; Formal Analysis—L.A.R. and J.P.F.; Writing—original draft preparation, L.A.R.; Writing—review and editing, L.A.R., J.P.F., R.E.L., C.E.C. and J.E.L.; Visualization, L.A.R.; Supervision, L.A.R.; Project Administration, L.A.R.; Funding Acquisition, L.A.R. All authors have read and agreed to the published version of the manuscript.

Funding: Funding was provided by Penn FRES and by a cooperative agreement between the USDA Forest Service Philadelphia Field Station and the University of Pennsylvania, 14-JV-11242308.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: The data presented in this study are openly available at the USDA Forest Service Research Data Archive at: https://doi.org/10.2737/RDS-2022-0038.

Acknowledgments: We thank D. Garofalo and D. Hollenberg, formerly with Penn FRES, for supporting this research. This work was funded by Penn FRES and the USDA Forest Service Philadelphia Field Station. We thank J. Rauer for assistance with archives for recent campus capital projects, K. Heinlen for assistance with the campus map, and J. Henning for comments on an earlier version of this manuscript. We also thank two reviewers for their constructive feedback. The findings and conclusions of this publication are those of the authors and should not be construed to represent any official USDA or US Government determination or policy.

Conflicts of Interest: Researchers from the funding agencies (USDA Forest Service and Penn FRES) codesigned and coauthored the study.

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