Trees in the City:
Spatiotemporal dynamics, benefits, and management of urban tree canopy cover in Massachusetts Gateway Cities

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A DISSERTATION

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ABSTRACT

Urban tree canopy loss in the United States is leading to decreases in ecosystem services that trees provide: reduced air pollution, stormwater runoff regulation, reduced cooling energy demand, urban heat island mitigation, and other benefits to the health and quality of human of life. Across the US, and in Massachusetts, recent urban tree canopy loss has been due to urban development and densification which can remove canopy cover and other greenspace, and replace it with impervious cover (e.g., roads, buildings, and parking lots). As a way to counter this trend, in 2014 the Commonwealth of Massachusetts started a tree planting initiative (TPI), Greening the Gateway Cities (GGC), that targets municipalities facing a range of social and environmental challenges due to post-industrial legacy effects. Between 2014 and 2021, the GGC tree planting initiative planted over 20,000 trees across an initial 14 cities and towns in Massachusetts, and since 2020 has expanded to include eight more municipalities. Like all TPIs, the GGC trees must be able to reach maturity in order to fully provide ecosystem services, but little research has explored the benefits of juvenile trees, or how municipalities respond to large scale TPIs. This dissertation aims to understand the efficacy of the GGC TPI through three strands of research: 1) legacy effects (human and biophysical) on the urban tree canopy in two GGCs; 2) measurable urban heat island mitigation of GGC trees in Holyoke, MA; and 3) urban forest management motivations and practices at the municipal level after the influx of trees in seven GGCs. Chapter 2 delineates the historical urban tree canopy of Chelsea and Holyoke, MA (USA) between 1952-2014 and demonstrates how biophysical factors and cycles of governance and urban development and decay have influenced the spatiotemporal dynamics of urban tree canopy. While urban tree canopy gains occurred in depressed economic periods, canopy losses

occurred in strong economic periods, in contrast to previous research. The urban tree canopy cover metrics used in this chapter can help municipalities with the creation of targeted, feasible urban tree canopy goals. Chapter 3 explores the temperature variation between Landsat-derived surface temperature and in-situ sensor-derived air temperature in the GGC tree planting zone in Holyoke, MA (USA). Significant air temperature cooling effects (up to -0.087 ℃ per GGC tree) were found from the juvenile trees on the hottest days of the year between 2017-2021. Using land surface temperature to monitor the cooling effects of juvenile trees was not successful but may be more appropriate in the long-term as the GGC trees mature. Chapter 4 investigates municipal tree maintenance practices and departmental structure in seven GGCs through interviews with tree wardens (municipal urban forest managers). The majority of tree wardens found the burden of public trees planted by the GGC to be minimal which may affect their future outcomes because not all the cities employ a high number of proactive maintenance practices. Improper or insufficient maintenance could result in higher costs for future maintenance and may lead to negative experiences and perceptions of trees by residents. Municipal departmental structure was found to influence both the number of proactive tree management practices and the size of the tree activity budget. This dissertation shows the effects of a TPI through different levels of analysis, demonstrating how biophysical and human legacies interact with tree professionals to produce the spatiotemporal dynamics of the urban forest in Massachusetts' post-industrial cities.

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DEDICATION

I dedicate this dissertation to my family, Jacque, James, Meg and Francis. To Jacque whose support, love, patience (especially through all those late nights!) and editing skills made all the difference. To James, Meg and Francis who always keep me grounded (or should I say rooted) and remind me to play more and work less.

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1.1 Urban forest canopy cover decline: globally and in the US

Urban forests—the trees on our streets, parks, and properties—can provide a broad range of ecosystem services: stormwater runoff regulation (Xiao and McPherson, 2002; Berland et al., 2017), reduced cooling energy demand (Akbari, 2002; Nowak et al., 2014; Ko, 2018), and mitigated urban heat island effect (Arnfield, 2003; Ziter et al., 2019). Urban forests also provide aesthetic benefits (Price, 2003; Tyrväinen et al., 2003), increase mental health (Dadvand et al., 2016), reduce crime (Troy et al., 2012; Schusler et al., 2018), and decrease human stress (De Vries et al., 2013). Despite overwhelming evidence of the human-environmental benefits urban trees provide, tree canopy in urban areas is declining globally by as much as 40,000 ha per year, and this dire situation is compounded by the fact that impervious surfaces—roads, buildings, and parking lots—are increasing by as much as 326,000 ha per year (Nowak and Greenfield, 2020). A survey of canopy cover in urban and community areas across the United States (US) using aerial imagery between 2009 and 2014 found no gain nationally in canopy cover and that average canopy cover (42.9%) declined 1% in urban areas and 0.7% in community areas, and gained 1% and 0.6% in impervious cover, respectively (i.e., equivalent to losing 36 million trees per year) (Nowak and Greenfield, 2018). Urban tree canopy (UTC) cover was shown to be lost to impervious surface or because of tree removal it uncovered impervious surface, indicating that future changes in tree canopy and impervious surface need to be monitored because understanding the variation of change can shape urban forest and sustainability policies (Wang et al., 2016; Nguyen et al., 2017; Eisenmen et al., 2021), characterize the effects of urbanization

(Walton et al., 2008; Medley et al. 1995), and is a simple metric for public awareness and comparison across multiple scales (i.e., neighborhood, city, region) (McPherson et al., 2011).

While global and US-nationwide observations of UTC loss started in 2003 (Nowak and Greenfield, 2012), they do not address UTC before this time period, or the legacies of previous land-use and human interactions that have shaped the patterns and processes of UTC loss, witnessed today. Current UTC is the result of natural regeneration and/or tree planting (or lack thereof) previously done, and other biophysical and historical factors, therefore more research is needed to understand the historical trajectory of the modern urban forest (Roman et al, 2018).

In order to address the decline of UTC and the impact of historical legacies, and to better equitably share the human-environmental benefits of urban trees, it has been suggested that focus is needed on supporting and expanding UTC (Nowak and Greenfield, 2018; Roman et al, 2018). Supporting and expanding UTC can be accomplished through tree planting initiatives (TPIs) that plant trees across various urban land types and that monitor and manage tree health and survivorship (Eisenman et al., 2021). Long-term, continual tree planting by TPIs that focus on tree health and survivorship is critical because the benefits of urban trees are fully realized when they can reach maturity, which is species-dependent and can take up to 30 years (Ko et al., 2015).

1.2 Greening the Gateway Cities – A Massachusetts Case Study

The Greening the Gateway Cities (GGC) TPI, is focused on increasing UTC cover to lower utility bills in underserved, post-industrial communities across the Commonwealth of Massachusetts, was started in 2014. The GGC TPI is part of a broader statewide initiative for aiding Massachusetts Gateway Cities. A "Gateway City" is defined as a mid-sized, regional economic center that historically provided manufacturing jobs, but which today face a range of social and economic challenges in the post-industrial economy in Massachusetts, while retaining many assets with unrealized potential (MassINC, 2015). Gateway cities receive extra funding and assistance from state programs that focus on community and economic development. Eligibility for state assistance is based on three metrics: 1) population between 35,000 and 250,000; 2) median household income below the state average [$70,954]; and 3) average educational attainment level (Bachelor's or above) below the state average [41.3%] (US Census Bureau, 2016). The GGC TPI is administered by the Massachusetts Department of Conservation and Recreation (DCR) which is unique as most TPIs are municipally or non-profit managed (Eisenman et al., 2021). The GGC is similar to other TPIs across the US as it involves tree planting on a variety of landscapes (e.g., residential yards, sidewalks, and institutional grounds spanning both public and private land) with a focus on providing ecosystem services and other benefits to residents (Breger et al., 2019; Nguyen et al., 2017; Eisenman et al., 2021).

The GGC differs from other TPIs in the US that focus on tree giveaways for residential yards (i.e., residents collect free trees from a depot and are responsible for planting and maintenance, Nguyen et al., 2017) or that have arborists from non-profits or municipal government train volunteers to carry community and street-tree planting (Roman et al., 2015; Fisher et al., 2015; Hauer et al., 2018). In contrast, the DCR funds and manages the planting of all GGC trees using trained arborists and seasonal staff (Breger et al., 2019). The GGC TPI uses an 80:20 planting ratio of residential versus public sites, and works with landowners, city public works departments and local non-profits for watering and maintenance (Breger et al., 2019).

GGC aims to “increase the UTC by 5-10 percentage points in select neighborhoods,” in order to reduce residential winter-heating and summer-cooling costs (Commonwealth of Massachusetts, 2015). The GGC is also expected to provide co-benefits such as reduced storm water runoff, improved air quality, increased property values and tax receipts, and a safer, healthier environment for residents (Commonwealth of Massachusetts, 2015).

The GGC TPI also provides an opportunity to study small to mid-size US municipalities which are underrepresented in the literature. Many TPI studies focus on larger cities (e.g., Young, 2011; Pincetl et al., 2013, Campbell, 2014; Locke and Grove, 2014), as they account for most tree planting, but in the context of the northeast US, smaller municipalities collectively comprise the larger population (Doroski et al., 2020). Between 2014 and 2021, the GGC TPI has planted over 20,000 trees across an initial 14 cities and towns in Massachusetts, and since 2020 has expanded to include eight more cities. My dissertation focuses on the original 14 cities and towns in the GGC TPI (Figure 1-1).

The long-term success of the GGC TPI, and indeed any TPI, hinges on establishment, survival and growth to maturity, to fully realize the benefits of increased UTC cover (Ko et al., 2015; Widney et al., 2016; Breger et al., 2019). To date, tree health assessments by the Clark University Human Environment Regional Observatory (HERO) program report an average survivorship of 87.5% (range of 78%-92%) across six surveyed GGCs (HERO, 2019). This survivorship metric is similar to other studies analyzing urban tree survivorship in the establishment phase (see Hilbert et al., 2019; Koeser et al., 2014). With the GGC TPI seeing early survivorship success, there are other strands of research that will be pursued to understand its efficacy: 1) legacy effects (human and biophysical) on the UTC in two GGCs; 2) measurable urban heat island mitigation as trees establish and grow; and 3) urban forest management motivations and practices due to the large influx of TPI trees.

1.3 Multi-decadal urban canopy cover mapping

Mapping UTC over several decades has been used to study canopy gains and losses attributed to urban development (Berland, 2012; Bonney and He, 2019) and to assess fragmentation of greenspace in suburbanizing landscapes (Zhou et al., 2011). Multi-decadal canopy cover assessment can be accomplished using various types of remotely sensed data, such as aerial photographs and airborne/satellite imagery (Walton et al., 2008). Over the past decade, the most common approaches for estimating canopy cover use fine resolution (< 2 m) imagery and include the dot method to visually interpret percent canopy cover based on randomly laid points across a landscape (Berland, 2012; Roman et al, 2017) or a fused combination of LiDAR and either aerial or satellite imagery to delineate canopy cover (O'Neill-Dunne, 2013; Parmehr et al., 2016). The dot method does not create spatially explicit layers; therefore, using it to identify patterns in UTC change is difficult (Locke et al., 2017; Roman et al., 2021). Unfortunately, fine resolution satellite imagery pre-1990s and LiDAR pre-2000s is nonexistent. However, aerial photographs have long been used in landscape ecology (Morgan et al., 2017; Brook and Bowman, 2006), and to delineate urban forested patches and tree density in a spatially explicit manner (Zhou et al., 2011; Gillespie et al., 2012). Aerial photos need to be interpreted and manually delineated to assess the spatial patterns and extent of UTC over time, and if the aerial photos are old enough, they may require digital scanning and georeferencing before analysis (Berland, 2012).

Conducting UTC cover assessments only provides a snapshot into the change taking place and miss the patterns and processes of how current urban forests formed, and what might affect them in the future. Understanding past drivers is critical to inform future urban environmental management approaches. Furthermore, trees are slow-growing organisms requiring decades to reach maturity, with a temporal lag between planting actions and substantial tree cover gains (Roman et al., 2018). Therefore, long-term, multi-decadal assessments and analysis of UTC and their associated historical drivers are needed to inform reasonable UTC goals (Roman et al., 2021), and to better understand how to maintain tree canopy and greenspace over time.

1.4 Urban heat island

The urban heat island (UHI) effect causes urban areas to be warmer than surrounding rural areas, with larger differences observed at night (Voogt and Oke, 2003). This is due to large amounts of impervious surface (brick, concrete and asphalt) and buildings that absorb heat during the day and remit it throughout the evening as the surfaces cool (Oke, 1982). The intensity of the UHI can be attributed to land-cover composition/change, UTC cover, and greenspace configuration in urbanizing/urbanized areas (Lo and Quattrochi, 2003; Wang et al., 2016; Elmes et al., 2017). Expansion of urban areas globally is projected to increase by almost 1 million km^2 by 2050, resulting in average summer daytime and nighttime warming of air temperature by 0.5–0.7°C, with up to 3°C in some locations (Huang et al., 2019). Increases in air temperature have led UHI reduction and mitigation to become an important goal for US cities (O’Neill et al., 2010; Norton et al., 2015; Wang et al., 2016; ACEEE, 2021), especially with the stress of regional climate warming possibly compounding the UHI effect. Recent models project that the northeastern US will warm by 3°C when global warming reaches 2°C (Karmalkar and Bradley, 2017). This places the northeastern US as potentially the fastest warming region in the contiguous US, over the next 30 years.

The UHI impacts human health and comfort (Milošević et al., 2017; Taleghani et al., 2015) while also exacerbating energy and water usage during the hottest months (Akbari et al., 1997; Akbari et al., 2001), which can severely impact low-income communities that lack access to cooling resources and increase heat-related health risks (Harlan et al., 2007; Tsilini et al., 2015). Increasing UTC cover and greenspace can help mitigate the UHI through increased shading, evaporative cooling, and reduced impervious surface (Akbari et al., 2001; Nowak and Dwyer, 2007; Ko, 2018; Ziter et al., 2019). TPIs are an essential part of increasing UTC to mitigate the UHI, and studies typically evaluate the projected, future benefits of recently planted trees (Solecki et al., 2005; Wang et al., 2016; Moody et al., 2021) because the benefits of juvenile trees are overlooked as being too little to matter or not significant. As a result, there is little research that has tracked the benefits, especially that of cooling, of the trees planted by TPIs over time.

The UHI can be tracked through the use of air temperature sensors which can record a diurnal range of responses, but the data are not continuous and can be difficult to map across a large city area (Smoliak et al., 2015). Remotely sensed thermal infrared data can be used to study the Surface Urban Heat Island (SUHI) effect through Land Surface Temperature (LST) products. SUHI, which is part of the UHI, can then be mapped, quantified, and modeled across entire cities or regions showing spatial temperature trends (Huang et al., 2011; Zhang et al., 2017; Elmes et al., 2017). Pairing the accuracy of urban air temperature sensors with LST has not been well researched but has the ability to demonstrate spatial trends that can identify locations that experience prolonged heat exposure with detailed temperature profiles (Elmes et al., 2020).

1.5 Urban forest management

In order for TPIs to deliver on promised urban tree benefits, trees must be able to reach maturity amid a complex web of governance and stewardship including residents, municipal staff, staff at non-profit organizations, and other stakeholders. When ownership, stewardship and maintenance responsibilities are clearly defined, well-funded, and supported by arboricultural expertise, juvenile trees may be expected to have high rates of survival (Roman et al., 2015; Geron et al., in review). In US cities, municipal agencies and staff are critical for providing stewardship and maintenance for newly planted trees, including street trees and other trees in the public right-of-way and in municipal parks (Braverman, 2008; Eisenman, 2016; Roman et al., 2018). As trees age they require different kinds of maintenance, that if not completed may lead to problems such as nuisance complaints (e.g., messy fruit droppings), public safety risks (e.g., falling limbs or trees), and fostering negative opinions towards trees (Shatz et al., 2013; Conway and Yip, 2016; Carmichael and McDonough, 2018). Thus, the long-term success of TPIs that plant on public property remains challenged by existing frameworks of municipal government and tree management (Pincetl et al., 2013; Hauer and Peterson, 2016; Geron et al., in review).

Urban forest management varies by municipality based on factors including community size (Grado et al., 2013; Treiman and Gartner, 2004), resource availability (Stobbart and Johnston, 2012; Miller and Bates, 1978) and residents' priorities (Treiman and Gartner, 2005).

These factors individually or collectively impact municipal tree care practices and residential stewardship. Furthermore, urban forest managers are employed across many different municipal departments such as parks and recreation, public works, planning, transportation, and forestry (Hauer and Peterson, 2016). Larger municipalities (>50,000) have been found to have more forestry-focused departments (e.g., parks and recreation, forestry) than smaller communities (Hauer and Peterson, 2016). This mismatch in departmental focus may lead to different priorities across departments, may vary their access to resources, and may impact tree care practices and management (Breger et al., 2019).

1.6 Dissertation chapters

The following lays out the research objectives and brief synopsis for chapters 2-4 of the dissertation. These chapters describe three interrelated research projects, centered on the Greening the Gateway Cities tree planting initiative in the Commonwealth of Massachusetts.

Chapter 2: Historical urban tree canopy cover change in two post-industrial cities

Research Objectives

1. Quantify urban tree canopy cover change (1952 to 2014) in Massachusetts Gateway cities Holyoke and Chelsea.
    a. Delineate urban tree canopy for years 1952, 1971, 2003, and 2014 and then define and calculate urban tree canopy metrics.
    b. Create persistence, gain, and loss maps for each time period to identify stable and changing tree canopy.

2. Describe how biophysical factors and cycles of governance and urban development and decay have influenced urban tree canopy using historical archival data.
    a. Use archival records (published articles, histories, public budgetary and planning documents, and newspaper articles) to uncover major local events and processes that impacted urban tree canopy spatiotemporal patterns.

Synopsis

This chapter shows how biophysical factors and cycles of governance and urban development and decay have influenced the spatiotemporal dynamics of UTC. An inverse relationship was found between UTC and economic prosperity: while canopy gains occurred in depressed economic periods, canopy losses occurred in strong economic periods. UTC cover metrics used in this chapter can help cities and towns with the creation of targeted, feasible UTC goals at both the neighborhood and city scales.

Chapter 3: How do juvenile trees planted through a tree planting initiative effect air and land surface temperature? A case study of Holyoke, MA

Research Objectives

1. Measure the cooling effects of recent Greening the Gateway Cities tree plantings have on air temperature in residential areas.
    a. Model the effects of percent canopy cover, percent impervious cover, and number of trees planted on hottest day temperatures
    b. Measure significance for hottest day temperature differences between Control and Planted residential areas

2. Evaluate the variation of land surface temperature across Holyoke residential sensor sites and measure the cooling effects recent Greening the Gateway Cities tree plantings have on land surface temperature.
    a. Compare land surface temperature point-based pixel values and averaged 100 m buffer-based pixel values to residential air temperature sensor values.
    b. Model the effects of percent canopy cover, percent impervious cover, and number of trees planted on hottest day land surface temperature temperatures for both point-based and buffer-based approaches.

Synopsis

This chapter explores the temperature variation between Landsat-derived surface temperature and in-situ sensor-derived air temperature in the context of a tree planting initiative. Significant air temperature cooling effects were found from the trees planted by the GGC TPI on the hottest days of the year between 2017-2021. Inconclusive results were found using a point-based or a 100 m buffer-based approach to analyze cooling effects of GGC trees on LST. Linear regression models were used to identify the relationship between the buffer-based LST measurements and landcover types and the number of GGC trees. The models produced high adjusted R^2 values and significant p-values, but none of the coefficients were significant. Using LST to monitor the cooling effects of juvenile trees is not currently recommended, but may be more appropriate as the trees mature, while in-situ air temperature measurements can readily demonstrate the cooling effects of juvenile trees.

Chapter 4: Urban forest management motivations and practices in relation to a large-scale tree planting initiative

Research Objectives

1. Evaluate urban forest management motivations and practices in relation to the Greening the Gateway Cities tree planting initiative.
    a. Conduct structured interviews with municipal tree wardens to understand the various impacts that current and future maintenance practices may have on the recently planted trees.
    b. Analyze department structure of interviewed tree wardens to assess its impacts on municipal funding and management practices of public trees.

Synopsis

Interviews were conducted with tree wardens (municipal urban forest managers) in seven GGCs to understand the various impacts that maintenance practices, municipal support and funding, and department structure may have on the recently planted trees. The future burden of maintenance for public trees planted by the TPI is perceived to be minimal by a majority of the interviewed municipalities. This is concerning for the municipalities that have small tree budgets and low numbers of proactive practices because of the possibility that GGC trees will be insufficiently cared for once the TPI ends. Insufficient care can result in higher costs for future maintenance and may lead to negative experiences and perceptions of trees by residents. Municipal departmental structure was found to influence both the number of proactive tree management practices and the size of the tree activity budget.

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Figures

Figure 1-1. Fourteen original Greening the Gateway Cities, 2014-2020, Massachusetts, US.

Chapter 2: Historical urban tree canopy cover change in two post-industrial cities[1]

Abstract

Present-day spatial patterns of urban tree canopy (UTC) are created by complex interactions between various human and biophysical drivers; thus, urban forests represent legacies of past processes. Understanding these legacies can inform municipal tree planting and canopy cover goals while also addressing urban sustainability and inequity. We examined historical UTC cover patterns and the processes that formed them in the cities of Chelsea and Holyoke, Massachusetts using a mixed methods approach. Combining assessments of delineated UTC from aerial photos with historical archival data, we show how biophysical factors and cycles of governance and urban development and decay have influenced the spatiotemporal dynamics of UTC. The spatially explicit UTC layers generated from this research track historical geographic tree distribution and dynamic change over a 62-year period (1952-2014). An inverse relationship was found between UTC and economic prosperity: while canopy gains occurred in depressed economic periods, canopy losses occurred in strong economic periods. A sustainable increase of UTC is needed to offset ongoing losses and overcome historical legacies that have suppressed UTC across decades. These findings will inform future research on residential canopy formation and stability, but most importantly, they reveal how historical drivers can be used to inform multi-decadal UTC assessments and the creation of targeted, feasible UTC goals at

[1] Healy, M., Rogan, J., Roman, L.A., Nix, S., Martin, D.G. and Geron, N., 2022. Historical Urban Tree Canopy Cover Change in Two Post-Industrial Cities. *Environmental Management*, pp.1-19. https://doi.org/ 10.1007/s00267-022-01614-x.

neighborhood and city scales. Such analyses can help urban natural resource managers to better understand how to protect and expand their cities' UTC over time for the benefit of all who live in and among the shade of urban forests.

1 Introduction

Urban tree canopy (UTC) is the portion of urban land covered by trees, and is comprised of trees along streets, in parks, and on public or private properties. UTC is shaped by the complex relationships between various human and biophysical drivers, which produce, over time, landscapes that represent legacies of past physical and social processes (Roman et al., 2018). These legacies explain species composition and distribution of trees planted across different time periods as well as social inequities (Roman et al., 2018; Watkins and Gerrish, 2018). In the United States (US), racial segregation and redlining (a race-based US policy that promoted segregation tactics in real estate, see Rothstein, 2017) from the early 20th century is associated with large environmental inequalities within cities, such as exposure to extreme heat due to high levels of impervious surface and low levels of UTC, compared to non-redlined neighborhoods (Hoffman et al., 2020; Locke et al., 2021, Nowak et al., 2022). In post-industrial cities, mid- and late-20th century depopulation and urban renewal also had lasting impacts on UTC in terms of building demolition, tree growth on abandoned lands, and interconnected suburbanization phenomena (Roman et al., 2021; Haase et al., 2014). The present-day spatial patterns of UTC within a given city result from the intricate ways these legacies have played out on the landscape.

Recent studies have statistically associated UTC distribution with sociodemographic variables: Schwarz et al. (2015) found a positive correlation between UTC cover and median household income across several US cities, while Zhou et al. (2021) found that across 38 US cities, people who are lower income, have low educational attainment, and are a minority live in hotter neighborhoods with less UTC. In response, local governments, nonprofit organizations and tree advocacy groups have committed to substantial tree planting and canopy cover goals to address urban sustainability and inequity (Nguyen et al., 2017; Eisenmen et al., 2021), but the legacies that produced current UTC do not typically inform how these programs establish and meet their goals. Much of the motivation for tree planting relates to the ability of trees to provide ecosystem services, which has taken more prominence in the scholarly literature (Roy et al., 2012) and urban environmental management (Young, 2013). This has led municipalities to gather information on the UTC within their boundaries (O'Neil-Dunne et al., 2013; Parmehr et al., 2016), and set goals for future UTC levels (Nguyen et al., 2017; Locke et al., 2016). But these analyses miss the patterns and processes of how current urban forests formed, and what might affect them in the future. Understanding past drivers is critical to inform future urban environmental management approaches. Furthermore, trees are slow-growing organisms requiring decades to reach maturity, with a temporal lag between planting actions and substantial tree cover gains (Roman et al., 2018). Therefore, long-term, multi-decadal assessments and analysis of UTC and their associated historical drivers is needed to inform reasonable UTC goals (Roman et al., 2021), and to better understand how to protect trees and greenspaces over time.

The mapping of UTC over several decades has been used to study canopy gains and losses attributed to urban development (Berland 2012; Bonney and He 2019) and to assess forest fragmentation in suburbanizing landscapes (Zhou et al., 2011). Multi-decadal canopy cover assessment can be accomplished using various types of remotely sensed data, such as aerial photographs and airborne/satellite imagery (Walton et al., 2008). Over the past decade, the most common approaches for estimating canopy cover use fine resolution (< 2 m) imagery and include the dot method to visually interpret percent canopy cover based on randomly laid points across a landscape (Berland, 2012; Roman et al, 2017) or a fused combination of LiDAR and either aerial or satellite imagery to delineate canopy cover (O'Neill-Dunne, 2013; Parmehr et al., 2016). The dot method does not create spatially explicit layers; therefore, using it to identify patterns in UTC change is difficult (Locke et al., 2017; Roman et al., 2021). Unfortunately, fine resolution satellite imagery pre-1990s and LiDAR pre-2000s is nonexistent. However, aerial photographs have long been used in landscape ecology (Morgan et al., 2017; Brook and Bowman, 2006), and to delineate urban forested patches and tree density in a spatially explicit manner (Zhou et al., 2011, Gillespie et al., 2012).

Currently, mixed-method, interdisciplinary research has had little representation in the field of urban forestry but there are calls for this to change (Roman et al., 2018; Vogt, 2018). Qualitative historical analysis can help interpret past UTC change processes and spatial patterns (Roman et al., 2017; Roman et al., 2021; Nix et al., 2022), leading to a better understanding of human and biophysical impacts on UTC. Incorporating local urban history into UTC assessments can provide a more holistic understanding of the causes and consequences of UTC change, revealing how legacy effects impact the present-day UTC (Roman et al., 2021), while better informing future planning and policy.

In this study, we investigate historical UTC cover patterns and processes in Chelsea and Holyoke, Massachusetts (US; Figure 2-1) from 1952-2014. During this time period, both cities experienced post-industrial downturns, radical shifts in population racial makeup, urban renewal programs, and economic transformation. Similar change processes occurred throughout many post-industrial towns and cities in the midwestern and northeastern US, southeastern Canada, and western Europe (Wachter and Zeuli, 2013; Neumann, 2016). The UTC cover changes that played out in Chelsea and Holyoke may therefore be indicative of broader changes that have taken place in other old industrial centers. UTC metrics were assessed and then contextualized using local history from each city, generating insights into the drivers of change that will be critical for informing urban tree planting and management.

2 Methods

2.1 Study Area

Massachusetts is in the New England region of the US and was forested prior to European colonization. During the 18th-19th centuries, forests were cleared around much of New England as land was converted to agriculture. Chelsea is situated in eastern Massachusetts, 4 km northeast of Boston, in a humid subtropical climate, while Holyoke is located in western Massachusetts in a humid, continental climate. This study covers the entirety of Chelsea (5.7 km2), which has the second-highest population density in the state. Holyoke is geographically much larger (59.0 km2), but a forested state park constitutes nearly half the city. Our analysis focuses on the heavily urbanized city core (10.2 km2), which we selected based on 1952 imagery outlining neighborhoods surrounding the city center.

2.1.1 Chelsea Historical Context

Originally a resort for Boston's elite (Gillespie, 1898), by the mid-19th century Chelsea had developed into an industrial center with extensive ship construction and manufacturing (Lake, 2011), including the establishment in 1836 of the Chelsea Naval Hospital. The city quickly became a hub for new immigrants, especially Russian Jews (Lake, 2011). Chelsea suffered a catastrophic fire in 1908 that burned down more than one-quarter of the city. This area was rebuilt with a focus on manufacturing, which created new jobs and led Jewish migrants to settle in Chelsea. The mid-20th century saw residential tax increases, increasingly dilapidated aging housing, and the construction of the Mystic River (Tobin) Bridge and US Route 1 highway, displacing residents and leading many families to leave to surrounding suburbs (Lake, 2011). Chelsea's population peaked in 1930 at 45,816 (US Census, 1930), which declined due to the suburbanization process just described, with more outmigration following another massive fire in 1973 that destroyed approximately one-fifth of the city. The population bottomed out at 25,431 in 1980 (US Census, 1980). New immigrants from Puerto Rico and Central America began to settle in Chelsea in the late 20th century due to the location in the Boston metropolitan area and low cost of housing, despite racial tensions with local government authorities (Lake, 2011). Following years of reduced tax revenue, misgovernance and corruption, Chelsea was managed by the Commonwealth of Massachusetts from 1991-1995, after which local governance was restored. (The Boston Globe, 1991a; The Boston Globe, 1994). See Table 2-1 for 2020 US Census data for Chelsea.

2.1.2 Holyoke Historical Context

In contrast to Chelsea, Holyoke was a company town that sprung up quickly as a planned industrial community next to the Connecticut River. Specifically, a group of Boston financiers founded the Hadley Falls Company, which funded the construction of the Hadley Falls Dam and canal system starting in 1847 (The Hadley Falls Company, 1853). The city plan, created by the company, called for dwellings 3-4 stories high, located adjacent to the industrial area, because the company believed Holyoke would eventually grow to a population of 200,000 and wanted to preserve land around the canals for mills and factories (Curran, 1960). This land use pattern shaped the present city core, while surrounding neighborhoods developed later with mostly single-family homes. By 1920, the population of Holyoke had grown to over 60,000, with dozens of mills and machine shops. The closure of some of these businesses resulted in outmigration from Holyoke, slowing down only in the 1940s due to World War II spending. Holyoke was once a major global center of papermaking, but more than half of the paper plants closed by the 1990s (Jacobson-Hardy and Weir, 1992). Due to mechanization of agriculture in Puerto Rico and the resulting mass unemployment, between 1945-1965 there was a large migration of Puerto Ricans to Holyoke, and other eastern US cities (Borges-Méndez, 2007). The magnitude of Puerto Rican settlement in Holyoke has given the city the highest percentage of Puerto Ricans of the total population of any municipality in the US, outside of Puerto Rico (Hardy-Fanta, 2002). See Table 2-1 for 2020 US Census data for Holyoke.

2.2 Aerial photo preparation

To assess the spatial patterns and extent of UTC in Chelsea and Holyoke over time, we delineated tree canopy from digitally scanned and georeferenced aerial photos, following Nix et al. (2022). Whereas Nix et al. (2022) focused on urban parks, we expanded upon that study by applying similar methods to an entire city and an urbanized city core. Following this precedence and those of other studies on multi-decadal urban tree and forest cover (Zhou et al., 2011; Merry et al., 2014) we manually delineated tree cover polygons to create spatially illustrative datasets. We used leaf-on imagery across 62 years: 1952, 1971, 2003, and 2014 (Table 2-2).

Aerial photos from 1952 were stored as 9×9-inch (22.86×22.86 cm) prints at the Clark University Map Library while the 1971 aerial photos were stored as negatives at the US Department of Agriculture Aerial Photography Field Office located in Salt Lake City, Utah. Both sets of images had 1:20,000 resolution, and individual images had an approximate extent of 20 km2. We scanned the image sets into either Tagged Image File Format (TIFF) or GeoTIFF and prepared for georeferencing. We cropped the images that covered the study area (Chelsea – two images per year; Holyoke – 4 images per year) to remove registration marks and edge distortion. We used United States Geological Survey (USGS) imagery from 2014 with 30 cm resolution (MassGIS, 2019) for the georeferencing process. We selected this USGS imagery because it has higher resolution than the NAIP imagery (1 m) used in this project and because it shares the same projection as the NAIP imagery: NAD83/UTM Zone 18N (for western Massachusetts) or 19N (for eastern Massachusetts).

We selected ground control points (GCP) to connect the 2014 USGS imagery to the 1952 and 1971 aerial photos, favoring hard-edged GCPs (i.e., building/roof corners, street and sidewalk intersections, monuments) as they are locations that normally do not change position over time and usually lead to lower Root Mean Square Error (RMSE) (Hughes et al., 2006). On average, 14.3 GCPs were used per image—removing or replacing some that had abnormally high RMSE values (>15 m). By keeping first order RMSE within an acceptable level (Berland, 2012; Baily and Inkpen 2015), we applied second or third order polynomial transformations to the images based on levels of alignment and distortion, which led to an average RMSE of 2.97 m. Post georeferencing, we mosaicked the images together to create a final composite of each study area.

2.3 Manual delineation of UTC cover

We created a set of rules (Table 2-3) to accurately and consistently digitize polygons of canopy cover, adapting rules used by Zhou et al. (2011) and Nix et al. (2022). We used a minimum polygon size of 7 m^2 as the threshold for a minimum mapping unit, similar to Gillespie et al. (2012). This mapping unit was the smallest unit at which a juvenile tree could be consistently recognized by an interpreter based on its shape and corresponding shadow. Displacement can cause small amounts of error between each year of UTC cover polygons (primarily in georeferenced datasets). When placed over one another, the polygons will look offset from each other, but such errors can be reduced by utilizing the stated minimum mapping unit. As suggested by Zhou et al. (2011), we employed a minimum area of change across these metrics (7 m^2, the same as the minimum mapping unit), to reduce the effects of misregistration and small amounts of edge change that may not be considered as real change. Some shrub cover was also included in the delineation of tree cover as there is no way to entirely differentiate the two in a two-dimensional image (Nowak and Greenfield, 2020; Roman et al., 2017).

All imagery used in this study was leaf-on, therefore some edges of tree canopy were obscured by shadow. Table 2-3 describes the rules that were used for reducing delineation error due to shadow effects and for identifying canopy cover versus other types of cover (i.e., shrub cover, grass cover, impervious surface). The other dates of imagery used in this study assisted in the interpretation and delineation of UTC. We used one interpreter for the study to promote consistency, with validation areas being delineated by a second interpreter (Berland, 2012; Roman et al., 2017; Nowak and Greenfield, 2018; Roman et al. 2021).

2.4 Canopy cover and error assessment

We assessed UTC metrics according to the computational definitions in Table 2-4. Then we made gain/loss/persistence maps for each time interval of the study period: 1952-1971, 1971-2003, 2003-2014 (Pontius et al., 2004). Notably, persistence in UTC can be computed relative to a given study area (e.g., the entire city) or relative to the initial canopy from the earliest time period. For instance, considering the pairs of images just mentioned (1952-1971, 1971-2003, 2003-2014), we calculated persistence both as a portion of the entire study area, and as a portion of the canopy area from the first image (1952, 1971, and 2003, respectively).

However, while persistence is often discussed in relation to only two years of imagery (Pontius et al., 2004; Locke et al. 2017), the notion of where canopy persists can be extended beyond this to span more than two years of imagery, and the intersection of persistence across multiple time periods. When more than two images are compared, we considered stable canopy as that which was tree cover across multiple images, adapting Roman et al. (2021). More specifically, as with persistence, stable canopy can be calculated in relation to a given study area or relative to initial canopy from the earliest time period.

We also propose a new UTC metric, maximum saturation, which is defined as the aggregated area of tree cover across the entire study period (i.e., spaces that had tree cover in any year), divided by the area of the study area. This metric is useful for understanding what the potential maximum of UTC could be across the study area, based on past and current land cover. The metrics we used in this study are reported on the scale of the total study area for each city (i.e., the entirety of Chelsea and the urbanized core for Holyoke), and then at the neighborhood scale.

Discrepancies between image interpreters can introduce error into the UTC product (Richardson and Moskal, 2014). To reduce this error, one interpreter carried out manual delineation across the entire study area while another interpreter delineated approximately 10% of the area to validate the replicability of the dataset. We calculated the percent agreement between interpreters (Table 2-5) and both cities had over 93% agreement between interpreters, similar to other studies of this nature (Roman et al., 2021; Roman et al., 2017; Zhou et al., 2011). Both interpreters' tree cover estimates were close, with the average difference across all years in Chelsea being 0.54 percentage points and 1.80 percentage points in Holyoke. We calculated detectable change thresholds for each city following Roman et al. (2021). For the entirety of the 62-year study period, UTC changes larger than 1.07 percentage points in Chelsea and 4.76 in Holyoke will represent detectable change.

2.5 Historical Research

This study follows other research that uses qualitative historical narrative to explain quantitative UTC change (Roman et al., 2017; Roman et al., 2021; Ogden et al., 2019). In order to link UTC change with local history, we consulted local archivists at the Chelsea Public Library Historical Archives, Holyoke Public Library History Room and Wistariahurst Museum about published histories and articles regarding Chelsea and Holyoke and concerning potential drivers of UTC change during the study period. They suggested various sources, such as published histories and articles (e.g., Curran, 1960; Harper, 1973; Lake, 2011), and public budgetary and planning documents (e.g., Presley Associates Inc., 1999; City of Chelsea, 2006; City of Holyoke, 2017), which led to backward and forward chaining to identify pertinent articles. This information provided a contextual backdrop for the underlying issues affecting each city, such as large infrastructure developments, significant racial change, and municipal budget cuts. We examined these topical leads further through newspaper articles using the database of newspapers.com (Ancestry, 2021). We investigated the processes behind large UTC changes detected in the imagery using the resources mentioned above; site investigations across the imagery verified the types of change taking place (i.e., vacant site in one time period has apartments in the next). Collectively, all this information was used to uncover major local events and processes that impacted UTC spatiotemporal patterns.

3 Results

3.1 Chelsea: Canopy Cover

Between 1952 and 2014, UTC cover more than doubled in Chelsea (+6.41 percentage points), far higher than the detectable change threshold (1.07), indicating that UTC change was meaningful. The largest UTC increase took place between 1952 and 1971 (Table 2-6). Both persistence metrics increased over the study period (Figures 2-4) with the largest amounts of persistence from 2003-2014 (Table 2-7; Figure 2-4). This is because the 2003-2014 time interval is shorter than the others, which inflates the percentage. However, the growth of persistence UTC also suggests that areas consistently covered by tree cover are growing. The low portion of Chelsea that has been consistently covered by UTC (0.87%) in all images (i.e., stable canopy metrics, Table 2-8) reflects the already low canopy cover found in Chelsea (Table 2-6), but also the high level of UTC turnover taking place—that is, the readily visible gain and loss in each time interval (Figures 2-4). The spatiotemporal dynamism of UTC in Chelsea is also reflected by the stable canopy relative to initial canopy area, which is 14.08%. In other words, only 14.08% of the UTC present in 1952 remained tree covered in all subsequent images. The maximum saturation of UTC for Chelsea was 24.71%, with a breakdown by neighborhood in Table 2-9. Figure 2-5 shows the neighborhoods of Chelsea and the UTC levels in each year of the study. Every neighborhood except for Everett Avenue, Mill Hill, and Addison Orange had net gains of approximately 5 percentage points of UTC. Everett Avenue, Mill Hill and City Center had the lowest maximum saturation but also appear to have had the highest levels of impervious surface in the city and the least greenspace. The residential neighborhoods of Shurtleff Bellingham, Soldier's Home, and Prattville all increased in canopy cover, while Addison Orange declined.

3.2 Chelsea: UTC Historical Change & Context

3.2.1 1952-1971

Construction of the double decker Tobin Bridge (1948-1950) and the Route 1 highway (1956-1958) in Chelsea split the city in two. Many residential properties were demolished (Lake, 2011) and canopied thoroughfares were cleared so Boston could have better connections to neighboring towns. Streets were also widened, leaving little to no space for street trees. In other parts of Chelsea, residential areas lost canopy to densification as apartment buildings were built and some vacant and other properties were built up for multi-family housing. The Everett Avenue neighborhood had industrial properties built on aggregated smaller properties that previously had tree canopy on their property. The Prattville neighborhood had new housing built on vacant lots which led to some tree loss, but most of the change there is from UTC turnover.

Properties with steep inclines around the Soldiers Home neighborhood were never built on and have filled in with vegetation over time. Similar growth occurred in the Admiral's Hill neighborhood around the US Naval Hospital. Edges along railroad corridors saw increased canopy cover as well as vacant and under-developed properties. Patches of canopy cover growth were seen in residential areas across the city and were especially strong in Prattville, where the properties are slightly bigger with more greenspace.

3.2.2 1971-2003

In 1973, a large fire destroyed most of the Everett Avenue neighborhood, including the canopy. The burned area was redeveloped into an industrial park and large retail mall (Lake, 2011) and the site appears to have more impervious surface than the previous land use, while trees were planted along the periphery and streets. The US Naval Hospital, in the Admiral's Hill neighborhood, was closed in 1974 and donated to the Commonwealth of Massachusetts and the

City of Chelsea (The Boston Globe, 1974b). The waterfront area around the Naval Hospital became a state park, locking in the greenspace for public use and UTC growth. The remaining area of the Naval Hospital was developed into apartments and condos, reusing some existing Naval Hospital buildings and constructing several new ones (The Boston Globe, 1983), which resulted in canopy loss. Canopy growth expanded most in the state park and in steeper incline areas where development was more difficult. New housing and commercial developments were built in the Soldier's Home, Mill Hill, and Shurtleff Bellingham neighborhoods and were responsible for UTC loss, which was visible from the aerial photos. Most of the new housing that was built were apartments and townhomes with only one small section where single family homes were built in Mill Hill. Further densification seems to have taken place as properties were developed/redeveloped.

Many of the larger visual UTC gains came from vacant, undeveloped, and edge areas across the city, for example, properties bordering Route 1 (Figure 2-3). A commercial area in the Soldier's Home neighborhood has highly visible UTC growth but across the city most UTC growth was constrained to smaller properties. Public parks saw an increase in UTC, as well as some streets in the Shurtleff Bellingham neighborhood, which were recipients of a 1994 Massachusetts Department of Conservation (DCR) street tree planting grant (Massachusetts DCR, 2008). In 2001, the city started its own tree planting campaign (The Boston Globe, 2001), but details on the number of trees planted are unavailable. The city received another DCR tree planting grant in 2002 (Massachusetts DCR, 2008).

Other highly visible changes were the high amounts of turnover taking place in the residential neighborhoods. Street tree canopy loss seems to be taking place at a higher level in

Prattville, when compared to other residential neighborhoods of Chelsea. UTC may also have been impacted during this period due to a budget crisis and storm damage. As noted earlier, the city of Chelsea went into receivership in 1991, with budgets cut back to essential services only (The Boston Globe, 1992), but there was no specific mention about tree care. Also in 1991, Hurricane Bob hit several New England states causing widespread damage to infrastructure and trees (The Boston Globe, 1991b). Other large storms of note also passed through in 1993 and 1997.

3.2.3 2003-2014

Post-2000, Chelsea experienced economic renewal. When Chelsea left receivership in 1995, it had solid fiscal footing and was able to invigorate several areas of the city with new development, which was then associated with UTC loss. New apartment and assisted living developments in the Shurtleff Bellingham, Mill Hill, Addison Orange, and Admiral's Hill neighborhoods visibly reduced UTC. Commercial areas were developed/redeveloped in the Prattville, Soldier's Home, and Everett Avenue neighborhoods, with the latter being one of the most important, as it replaced a failing mall with a new transit-oriented development. Parts of the mall were demolished (The Boston Globe, 2007), and the property was relandscaped with new trees planted. Other commercial sites in Chelsea cleared emergent or weedy tree growth from their property. Some vacant properties were also cleared, for instance, a brownfield site situated in the Mill Hill neighborhood. Trees had been cut and a windmill installed, but further development halted as funding evaporated in 2009 (The Boston Globe, 2013).

Other undeveloped and vacant properties in Chelsea saw UTC growth, but across fewer sites in this period. In the Admiral's Hill neighborhood UTC growth continued to expand in steeper incline areas, and increase along streets, open spaces, and in the state park. There was also an increase of UTC along several streets in the Everett Avenue and Shurtleff Bellingham neighborhoods.

During this period, UTC turnover is evident, but overall there was UTC gain across the city. Several factors played a role in this outcome such as tree planting grants, zoning regulation, and management. Capital Improvement Plans (City of Chelsea, 2006; 2010; 2014)—five-year urban planning implementation reports—consistently mentioned street tree planting, but without any specific details. A 2008 budget report detailed specific tree planting by the city's Public Works department the previous year (63 trees) as well as a DCR grant of $15,000 for planting 60 more trees (City of Chelsea, 2008). Chelsea also received tree planting grants for the years 2004 and 2005 (Massachusetts DCR, 2008). Zoning regulations in Chelsea were changed in 2005 mandating requirements for new developments that outlined specific setbacks, amounts of greenspace, and the spacing of trees in planting strips (City of Chelsea, 2005). Also, in 2005 a municipal Tree Board was formed to help advise the city and to manage their annual Arbor Day planting event. That same year Chelsea joined the Tree City USA program and has continued with that program ever since. In 2012, the city received a DCR grant for an urban forest assessment, which reported the 2012 UTC with change metrics from 2004, sociodemographic analysis of UTC patterns, and very broad, undetailed tree planting plans focused only on public spaces (City of Chelsea, 2016).

3.3 Holyoke: Canopy Cover

Between 1952 and 2014, UTC in the Holyoke study area increased by 2.27 percentage points (Table 2-6), which is lower than the detectable change threshold (4.76), meaning that the total level of UTC largely remained the same across Holyoke during the study period. However, our change maps show UTC turnover occurring throughout the study area. For example, the persistence map of 1952-1971 (Figure 2-6) shows many small areas of loss and gain. UTC cover did have a more than a 4 percentage point gain in 2003 and is visually affirmed in the persistence map of 1971-2003 (Figure 2-7) with the overwhelming yellow areas signifying the areas of gain, while there was an UTC loss from 2003-2014 (Figure 2-8). Additionally, Figure 2-8 shows large areas of persistence, and as with Chelsea, this is due to the time period of 2003-2014 being much shorter than the others in this study. However, the overall change in persistence signifies that areas consistently covered by tree cover are growing. The low levels of stable canopy in Holyoke (i.e., the portion of land covered by tree in all images, Table 2-8) are higher than in Chelsea, reflecting the generally higher UTC found in Holyoke, but, like Chelsea, denotes that a high amount of UTC turnover has taken place. The high amount canopy turnover in Holyoke is further supported by the stable canopy relative to initial canopy area (Table 2-8) which is 27.30%. Therefore, just above one-quarter of the original area covered by canopy in 1952 remains covered in all subsequent images. The maximum saturation of UTC for Holyoke was 40.92%.

Figure 2-9 shows the neighborhoods of Holyoke and Table 2-10 shows UTC levels for each neighborhood and year of the study. Three neighborhoods—The Flats, South Holyoke, and Downtown—had the highest net growth, with an approximately 5 percentage point increase or more in UTC. These three neighborhoods appear to have some of the highest levels of impervious cover in the city and the least amount of greenspace, as they are mostly comprised of commercial and industrial areas. Along these lines, the maximum saturation is lower in the aforementioned three neighborhoods in comparison to the more residential neighborhoods of Highland, Oakdale and Elmwood. These three neighborhoods, made up of predominantly single-family homes, have the most consistent UTC percentage across the study period.

3.4 Holyoke: UTC Historical Change & Context

3.4.1 1952-1971

UTC losses due to construction played a large role in Holyoke's changing landscape in this period. Construction of Interstate 91 (1965) cut through the last remaining orchards and agricultural areas, as well as some forested patches along the western boundary of the study area. This construction reshaped major roadways in the city as arterial roads connecting to the highway were widened and mature street trees and plantable strips were removed. Street widening also seems to be the cause of street tree loss in the neighborhoods of Downtown, Oakdale and Churchill. Some streets completely lost all roadway planting spaces, while others were reduced in size. There was a large loss of mature street trees during this time period. The Springdale and Churchill neighborhoods had the most remaining undeveloped properties in 1952 but were subsequently built by 1971. The construction of a high school (1964) and middle school (1973) in the Elmwood neighborhood removed canopy on forested land that was previously part of a public park (Sanborn Map Company, 1956; Harper, 1973). Several other public parks lost tree canopy for an array of reasons: one factor could have been Ophiostoma sp., or Dutch Elm

Disease (DED), which afflicted Massachusetts during this period, with half a million trees lost across the state from 1941-1974 (The Boston Globe, 1974a). The full impact of DED in Holyoke is not known, but records from the Municipal Register of the City of Holyoke (an annual budget report) show expenses related to DED from 1949-1963, with 1963 being the last budget year in the archival collection (City of Holyoke, 1920-1963). These budget records also shed light on the nature of tree planting taking place in Holyoke. Prior to DED, tree planting was variable from year to year and may have completely stopped during World War II, due to funding cuts (City of Holyoke, 1920-1963). The budget records show tree planting starting again in 1954 and continuing every year through 1963 (when the record ends), seemingly to replace trees lost to DED (City of Holyoke, 1920-1963). Another factor that may be responsible for UTC loss is weather: three hurricanes hit western Massachusetts between 1954-1955, which downed trees and caused flooding (The North Adams Transcript, 1954; The Berkshire Eagle, 1954). Two of the hurricanes struck in 1954 and the Holyoke budget records show a special expense in that year for storm damage (City of Holyoke, 1920-1963). These factors may also explain the high amount of tree canopy turnover taking place in residential areas, mainly in the neighborhoods of Highland, Oakdale and Elmwood.

Canopy gain in this period was mainly on residential property with other notable gains on undeveloped and vacant properties in industrial areas and along railroad lines. We also observed UTC gains along the Connecticut River edge but have not identified a driver of this gain from the archival records. In the 1952 imagery, the shoreline of the river was mostly clear of brush and trees, presuming that it had been mowed, but in the 1971 imagery forest emergence is seen.

3.4.2 1971-2003

Visible UTC losses during this period came from the expansion of the Holyoke Medical Center and water treatment plant (Valley Health & Life, 2018; MassLive, 2014). Road construction for Interstate 391 (completed 1982) on the border of the Springdale and South Holyoke neighborhoods led to canopy loss as streets were widened and lanes added to develop the ramps needed for the interstate. Wistariahurst, a public historical center and garden, was a heavily canopied greenspace in the Churchill neighborhood in 1952. By 2003 the majority of UTC had been lost, with the property-surrounding street trees almost all removed without replacement, while the grounds maintenance of Wistariahurst primarily consisted of "vegetation removal and keeping the lawn areas mown" (Presley Associates Inc., 1999). Public parks also saw canopy loss during this period. Another notable loss of UTC can be seen along the canals in thin strips. Hundreds of flowering trees had originally been planted along the canals between the late 1940s and early 1970s (Harper, 1973). By 2003, many of these trees appeared to have died or been removed.

This period saw the largest increase of UTC overall. The imagery details canopy growth along the river edge, as well as emergent forest on undeveloped properties and abandoned vacant industrial properties (Jacobson-Hardy and Weir, 1992). However, many of the vacant lots in the Churchill neighborhood [products of urban renewal projects that demolished many buildings in the older parts of the city (City of Holyoke, 2017)], and rebuilt housing in the Downtown and The Flats neighborhoods, were not areas of tree canopy growth. The rebuilt housing replaced former three-to-four story, apartment-style, stacked tenement housing with duplexes or multi-family townhomes (City of Holyoke, 2017). This new housing had larger yards and expanded plantable space for trees, but UTC growth in these areas is much smaller than the emergent forest seen on vacant industrial properties. Public spaces such as parks and schools in the Downtown and Churchill neighborhoods saw an increase of tree canopy, especially as trees matured and increased their canopy size. Other city parks saw expansion of their canopy as greenspace areas filled in, although it is unclear whether this park canopy growth was due to planting, unintentional forest emergence, or both. Properties abutting Interstate 91 saw an increase of canopy directly next to the interstate. Some of the canopy gain during this period may have been due to tree plantings that took place in 1995, 2001, and 2003, made possible by grant awards by the DCR (Massachusetts DCR, 2008).

3.4.3 2003-2014

Major areas of UTC loss for this period are mostly tied to development of new or previously underused properties. For example, commercial development led to visible UTC loss in the Springdale neighborhood while industrial properties in The Flats reclaimed overgrown land. Downtown lost UTC around the canals as new investments for an innovation area led to a publicly accessible canal walkway and the building of the Massachusetts Green High Performance Computing Center (The Boston Globe, 2012; 2016a). As in previous time periods, public spaces faced UTC loss as parks were remodeled and expanded. For example, part of a forested city park was cut down for a new dog park (MassLive, 2012). The city library was renovated, and several large trees were removed to accommodate construction (The Boston Globe, 2016b; MassLive, 2011).

The major areas of UTC growth were visibly constrained to unintentional forest emergence along a rail line and vacant properties in South Holyoke. Much smaller gains scattered throughout the study area may be the result of crown expansion of surviving trees, and resident and/or city tree planting. In 2004, the city received a grant from the DCR for tree planting in its Downtown neighborhood (Massachusetts DCR, 2008).

4 Discussion

Our analyses illustrate the spatiotemporal variation of UTC across two cities. The UTC level in Holyoke was mostly steady across the study period, but saw large changes to its geographic distribution, similar to the findings from Merry et al. (2014) for Detroit and Atlanta between 1951-2010. In contrast, Chelsea saw a large increase in UTC during the study period, and this city also witnessed large changes in UTC distribution. Overall, both cities had low levels of stable canopy, with some differences across neighborhoods, and high amounts of turnover taking place in residential areas. Urban development, disease, storms and other factors can partly explain the low stability and high turnover, but more needs to be understood about UTC in residential neighborhoods, as they typically have higher levels of UTC, stable canopy, and greenspace for planting. Residential neighborhoods contain large portions of tree canopy in many cities (Nguyen et al. 2017), so future research should investigate the drivers of tree cover gains and losses on residential properties.

Urban form constrains or enables the trajectory of each city and neighborhood to gain UTC, particularly in terms of the amount of non-impervious plantable space (Roman et al. 2018; Ossola et al., 2019). Holyoke’s industrial neighborhoods exemplify this phenomenon, as such neighborhoods have the lowest UTC, with residential neighborhoods having the highest. While other studies have found spatial relationships among land use, urban form, and UTC change (Ossola et al., 2019; Pham et al., 2017), more research is needed regarding long-term UTC dynamics in industrial and post-industrial urban landscapes, particularly planned industrial areas like Holyoke. Notably, land use and urban form are far from permanent (Kane et al., 2014), and land use changes can impact the availability of UTC. Urban renewal had little direct effect on increasing UTC in Chelsea and Holyoke. Chelsea's urban renewal funding led to the creation of an industrial park (Lake, 2011) while Holyoke's produced vacant lots, but they were seemingly maintained to suppress vegetation growth. Evidence from other post-industrial cities like Baltimore and Philadelphia demonstrate that urban renewal can result in increased UTC in a variety of ways: urban renewal funding sponsored the planting of street trees, and buildings that were demolished and abandoned through urban renewal policies became sites of unintentional forest emergence (Merse et al., 2009; Roman et al., 2021). Other post-industrial and suburbanization processes such as highway construction and shifting economies had substantial impacts on UTC change in our study cities. Currently, both Holyoke and Chelsea are struggling to overcome the legacies of highway building. Both cities lost greenspace and UTC to highway construction, but also, surrounding streets were widened, resulting in the shrinking or removal of street tree planting strips. Similar impacts have been recorded in other US cities due to highway construction and street widening (McPherson and Luttinger, 1998; Merse et al., 2009), but more research is needed to fully understand the historical impacts of the US highway system on UTC. Holyoke also saw forest emergence around abandoned factories, schools, and an industrial park. In Chelsea, areas rebuilt from the 1973 fire appear to have more impervious cover than previous land uses, and more recently, urban density has increased with the construction of high-density residential buildings. Roman et al. (2021) found similar connections as redevelopment construction led to lower UTC as some neighborhoods densified and/or gentrified in Philadelphia. Increasing urban density has been seen as a positive development since it can counteract the negative effects of urban sprawl (Haaland & Konijnendijk, 2015). Yet, it can also reduce the amount of available greenspace in a city (Brunner & Cozens, 2013). Preserving trees and greenspace is critical with new developments, requiring zoning and ordinance regulations that include conserving vegetation and/or planting it as part of building and landscape design (Jim, 2004; Chojnacky et al., 2020).

UTC gains from unintentional forest emergence during years of declining municipal budgets came mostly from edge spaces and steep slope areas that are difficult to develop, similar to findings of Berland et al. (2015) where hilly terrain in Cincinnati was a primary factor for high UTC. Holyoke's shoreline became forested after these areas were seemingly abandoned. Steep slopes limited development in the hills of Chelsea, leaving some areas to be maintained greenspaces and others small forest patches. In both Chelsea and Holyoke, spaces bordering the interstate or highway were canopied consistently over time. We characterize such UTC increases as unintentional forest emergence, similar to observations by Roman et al. (2021) and Nix et al. (2022) for vacant properties and unmaintained parks in post-industrial Philadelphia.

UTC losses due to biophysical factors play a large role in UTC change. Hurricanes caused large swaths of damage across Holyoke and Chelsea. DED impacts varied, with Holyoke budgetary documents providing evidence of substantial tree loss—like other cities in New England and the midwestern US (Roman et al. 2018)—while in Chelsea it only played a small role, if at all. An elderly archivist who lived in Chelsea all his life does not remember U. americana trees being discussed in the city, nor has he has come upon any archival information talking about DED (B. Collins, personal communication, March 16, 2021). Therefore, past species choices and resulting loss from an invasive pathogen can explain some differences in UTC loss between Holyoke and Chelsea.

Other UTC change, both unintentional and intentional, are more broadly linked to economics. Both cities suggest an inverse relationship between wealth and UTC, contrary to other literature tying wealth to higher UTC (Roman et al. 2017; Locke et al., 2017). Specifically, we observed unintentional forest emergence during poor economic and budgetary periods, and the loss of it during economically strong periods. Some of the largest gains in UTC came from majority industrial and commercial neighborhoods where neglected and vacant properties became sites of unintentional forest patches. This effect has been seen in other post-industrial cities (Haase et al., 2014; Berland et al., 2020) and is especially prevalent for cities in forest biomes (Roman et al., 2018; Roman et al., 2021). Years later, the neglected and vacant properties have started to be reclaimed from the forest patches, with declining UTC from tree removal, and future UTC growth in these neighborhoods is constrained due to lack of greenspace. Forest emergence in edge spaces may represent a significant portion of UTC to the local neighborhood, however, such areas may not be protected greenspace and may be lost depending on the economic prospects of Holyoke and Chelsea. For land managers and urban environmental advocates, preserving unintentional forests can achieve similar ecological goals to traditional tree planting, but with less municipal investment and a greater chance of long-term success (Del

Tredici, 2010), yet they will require some maintenance, otherwise overgrown forested areas may be perceived as socially undesirable weedy spaces (Brownlow, 2006; Roman et al. 2021).

More recently, a smaller amount of new canopy was formed through new economic and governance structures such as grant funding, non-profit initiatives, new regulations and new governing bodies. Grant funding and non-profit programs have provided small bursts of new UTC in both Chelsea and Holyoke, especially during the 2003-2014 period, but more sustainable governance structures are needed to provide a constant influx of UTC that can compensate for UTC losses each year. Chelsea has enhanced urban tree governance through zoning regulations, the addition of a tree board to the local government, and by participating in national tree programs, all which have recently helped increase UTC in the city. However, without dedicated governing and funding structures (i.e., tree-focused municipal departments, city budget allocations for tree planting) sustainable canopy growth and retention cannot be achieved. Simply funding tree planting is not enough to increase the UTC, as newly planted trees require high levels of tree stewardship to ensure long-term survival and growth to fulfill UTC goals (Roman et al., 2015). Trees also require plantable greenspace, which is minimal in some dense neighborhoods in Chelsea and Holyoke, similar to other post-industrial cities in the northeastern US (Nguyen et al., 2017). The maximum saturation metric used in this paper uncovers the total extent of UTC across each city, which can help shape long-term UTC goals because it represents an approximate UTC maximum. By breaking it down further to the neighborhood level, feasible UTC goals that can be tailored to individual neighborhoods. This can impact how, where, and with whom tree planting programs focus their time and energy.

The availability of imagery for this research exposed how the scaling of time intervals effects affects visible UTC change. The first two time periods of this study cover 19 and 32 years of change respectively, while the last time period covers 11 years. These different time intervals display varying levels of UTC change and persistence. Measuring UTC over longer periods shows broad levels of change and low levels of persistence while shorter periods show specific UTC changes and high levels of persistence. These differences in temporal scale can be used to aid specific research goals, like understanding the spatial and historical factors of UTC, or small amounts of change that occur after a small number of years in response to new tree regulations or planting programs. As UTC mapping progresses in more cities, temporal scaling and other metrics can be used to analyze specific or broad levels of UTC change.

This study broadly concurred with drivers of UTC change identified in other post-industrial cities located within forested biomes, like Philadelphia, Cincinnati and Baltimore. Therefore, the UTC cover changes that played out in Chelsea and Holyoke may be indicative of broader changes that have taken place in post-industrial cities located within forested biomes. Our results cannot necessarily be generalized to cities with different socioeconomic histories or ecological conditions and further research is needed to compare long-term UTC spatiotemporal patterns across cities with divergent socioeconomic and ecological contexts.

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6 Tables

Table 2-1. 2020 sociodemographic characteristics of Chelsea, Holyoke, and the Commonwealth of Massachusetts (US Census, 2021; US Census, 2019).

Selected Demographic Variables	**Chelsea**	**Holyoke**	**Massachusetts**
Total Population	40,787	38,238	7,029,917
Race			
White	48.4%	87.5%	80.6%
Black	6.4%	4.5%	9.0%
Other	45.2%	8.0%	10.4%
Hispanic or Latino	67.0%	53.9%	12.4%
Educational Attainment (Age 25+)			
4+ Years College	18.5%	21.5%	43.7%
Household Median Income	$56,802	$40,769	$81,215
Persons Below Poverty Level	18.1%	29.3%	9.4%

Table 2-2. Imagery Data

Year	**Resolution**	**Type**	**Source**
1952	1:20,000	Black & White Film	USDA Agriculture Stabilization and Marketing Service [1]
1971	1:20,000	Black & White Film	USDA Farm Service Agency [2]
2003	1 m	Color Film	USDA National Agriculture Imagery Program
2014	1 m	4 Band Digital	USDA National Agriculture Imagery Program

1. Clark University Map Library
2. USDA Aerial Photography Field Office

Table 2-3. Rules for delineation of tree canopy polygons

1. Tree canopy polygon area must be greater than or equal to 7 m^2. Polygons smaller than this can be removed post delineation.
2. Remove treeless gaps within a canopy cover polygon if groundcover is identifiable and the gaps are not dominated by shadow. E.g., open space in a forested patch.
3. Use shadow to aid in discernment of tree canopy versus shrub (i.e., a juvenile tree would produce a visible, angled shadow denoting its height, while low-to-ground shrubs would not have this trait).
4. Use visual obstruction of objects (i.e., sidewalks, rooflines, roads) by tree cover to aid in identification.
5. Edges of delineated tree canopy that extend beyond the study area boundary should be clipped to said boundary post delineation.
6. A variable viewing scale of 1:400 to 1:1000 was used across all imagery during delineation.

Table 2-4. Definitions for urban tree canopy (UTC) metrics used in this chapter

UTC Metric	Computational Definition
UTC	The area of delineated tree canopy for a given year, divided by the area of the study area.
UTC net change	For two years of UTC, the percentage point increase or decrease in tree canopy. O'Neil-Dunne (2017) refers to this as 'absolute change' in UTC.
Persistence relative to initial canopy area	For two years of UTC, the overlapping tree canopy area between those two years is divided by the tree canopy area of the earliest year of the pair. In other words, the portion of initial canopy from the first date that remained canopy in the second date.
Persistence relative to study area	For two years of UTC, the overlapping tree canopy area between those two years is divided by the study area. In other words, the portion of the entire study area that was canopy in both years.
Stable canopy relative to initial canopy area	For more than two years of UTC, the intersected area of tree canopy from all available years is divided by the tree canopy area of the earliest year. In other words, the portion of initial canopy from the first date that remained canopy in all subsequent years of available UTC. Roman et al. (2021) referred to this as simply *stable canopy*.
Stable canopy relative to total study area	For more than two years of UTC, the intersected area of tree canopy from all available years is divided by the study area. In other words, the portion of the entire study area that was canopy throughout all available years of UTC.
Maximum saturation	The aggregated area of tree cover across the entire study period (i.e., spaces that had tree in any year) is divided by the area of the study area.

Table 2-5. Interpreter agreement between validation areas for Chelsea and Holyoke

Interpreter Agreement				
	1952	**1971**	**2003**	**2014**
Chelsea	96.92%	97.77%	96.46%	96.48%
Holyoke	93.66%	95.09%	93.34%	95.66%

Table 2-6. Urban tree canopy (UTC) cover metrics for Chelsea and Holyoke for each year in the study as well as the overall net change of urban tree canopy from 1952-2014.

	Chelsea		*Holyoke*	
Year	**UTC (m^2)**	**UTC (%)**	**UTC (m^2)**	**UTC (%)**
1952	355,439.5	6.20	1,924,237.1	18.79
1971	553,315.6	9.65	1,900,202.8	18.56
2003	623,431.4	10.87	2,328,049.2	22.73
2014	723,109.4	12.61	2,157,104.4	21.06
1952-2014 Net Change	+367,669.9	+6.41	+232,867.3	+2.27

Table 2-7. Persistence relative to initial canopy area and persistence relative to study area urban tree canopy metrics for Chelsea and Holyoke for each time interval in the study.

	Persistence relative to initial canopy area (%)		*Persistence relative to study area* (%)	
Year	**Chelsea**	**Holyoke**	**Chelsea**	**Holyoke**
1952-1971	37.07	48.47	2.29	9.11
1971-2003	37.02	57.61	3.57	10.69
2003-2014	59.68	67.98	6.49	15.46

Table 2-8. Stable canopy relative to initial canopy area, stable canopy relative to study area, and maximum saturation urban tree canopy (UTC) metrics for Chelsea and Holyoke from 1952-2014.

UTC Metric	Chelsea	Holyoke
Stable canopy relative to initial canopy area (%)	14.08	27.30
Stable canopy relative to study area (%)	0.87	5.13
Maximum Saturation (%)	24.71	40.92

Table 2-9. Chelsea neighborhood breakdown of urban tree canopy (UTC), stable canopy relative to initial canopy area and maximum saturation. Asterisk represents neighborhoods that exceeded the detectable change threshold.

Neighborhood	1952 UTC (%)	1971 UTC (%)	2003 UTC (%)	2014 UTC (%)	Stable canopy relative to initial canopy area (%)	Maximum Saturation (%)
Admiral's Hill-Waterfront*	3.9	7.4	10.4	14.6	20.8	22.9
Everett Avenue*	2.1	4.9	4.5	5.1	2.2	13.1
Mill Hill*	3.0	4.5	5.7	7.5	5.8	14.7
City Center*	3.5	5.2	7.6	10.1	6.7	18.6
Shurtleff Bellingham*	5.2	10.1	11.8	15.2	12.8	27.3
Addison Orange*	14.1	14.5	12.9	12.8	8.1	35.0
Soldier's Home*	10.1	16.4	19.6	21.9	29.6	36.6
Prattville*	12.9	18.9	20.0	19.9	12.4	42.1

Table 2-10. Holyoke neighborhood breakdown of urban tree canopy (UTC), stable canopy relative to initial canopy area and maximum saturation. Asterisk represents neighborhoods that exceeded the detectable change threshold.

Neighborhood	1952 UTC (%)	1971 UTC (%)	2003 UTC (%)	2014 UTC (%)	Stable canopy relative to initial canopy area (%)	Maximum Saturation (%)
The Flats*	2.9	6.0	13.2	12.3	24.1	20.6
South Holyoke*	2.9	4.4	9.4	11.5	21.9	17.4
Downtown*	8.7	8.2	14.7	13.6	6.8	27.3
Churchill	7.1	5.2	7.6	9.6	8.3	18.9
Highlands	30.6	30.8	33.0	32.4	29.4	59.5
Oakdale	25.0	24.7	28.2	25.4	24.2	51.7
Elmwood	25.8	25.4	28.6	25.0	32.6	50.9
Springdale	22.5	18.9	23.9	18.8	24.1	43.1

7 Figures

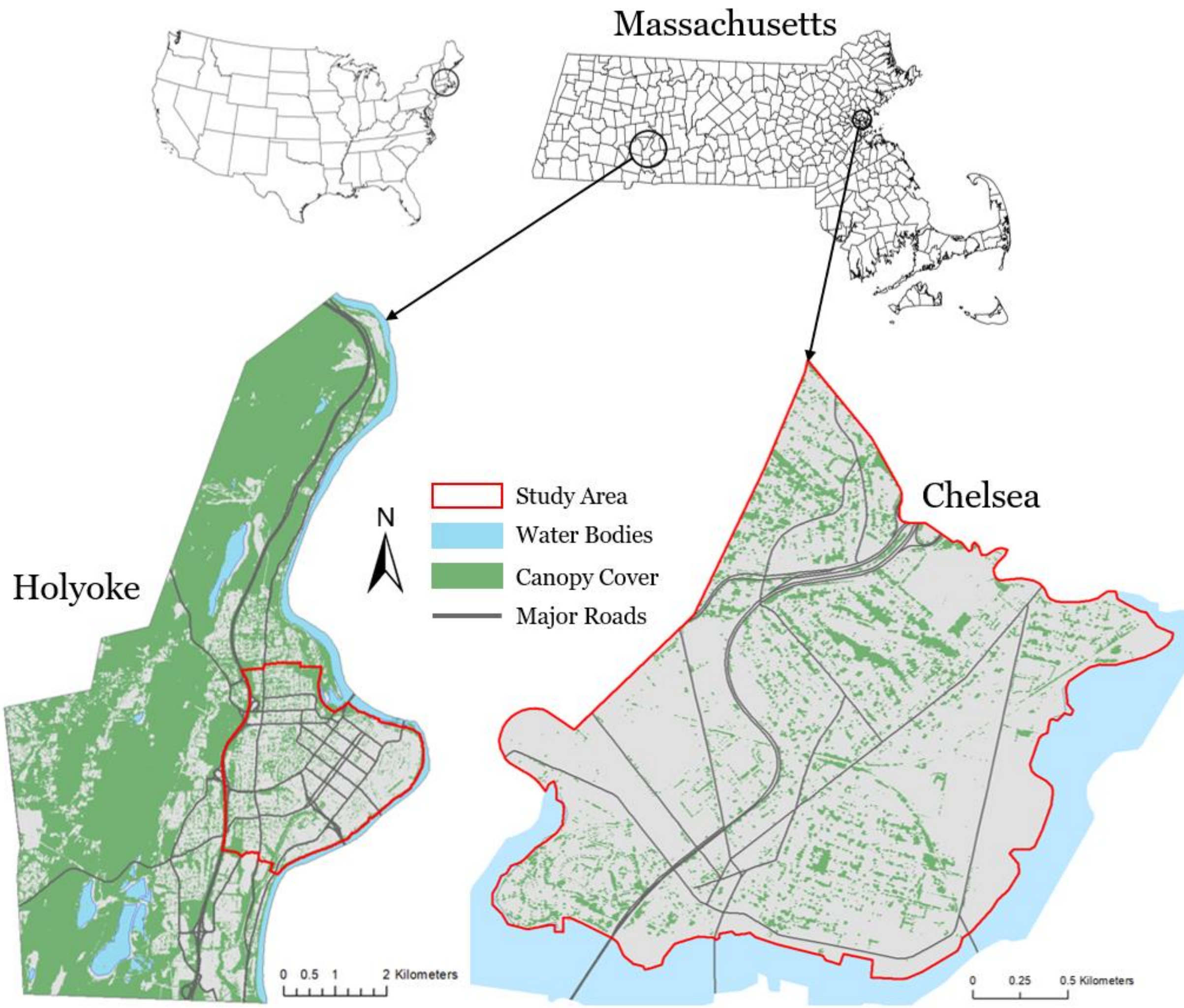

Figure 2-1. Holyoke and Chelsea, Massachusetts city boundaries displaying the study area, water bodies, 2014 urban tree canopy cover and major roads.

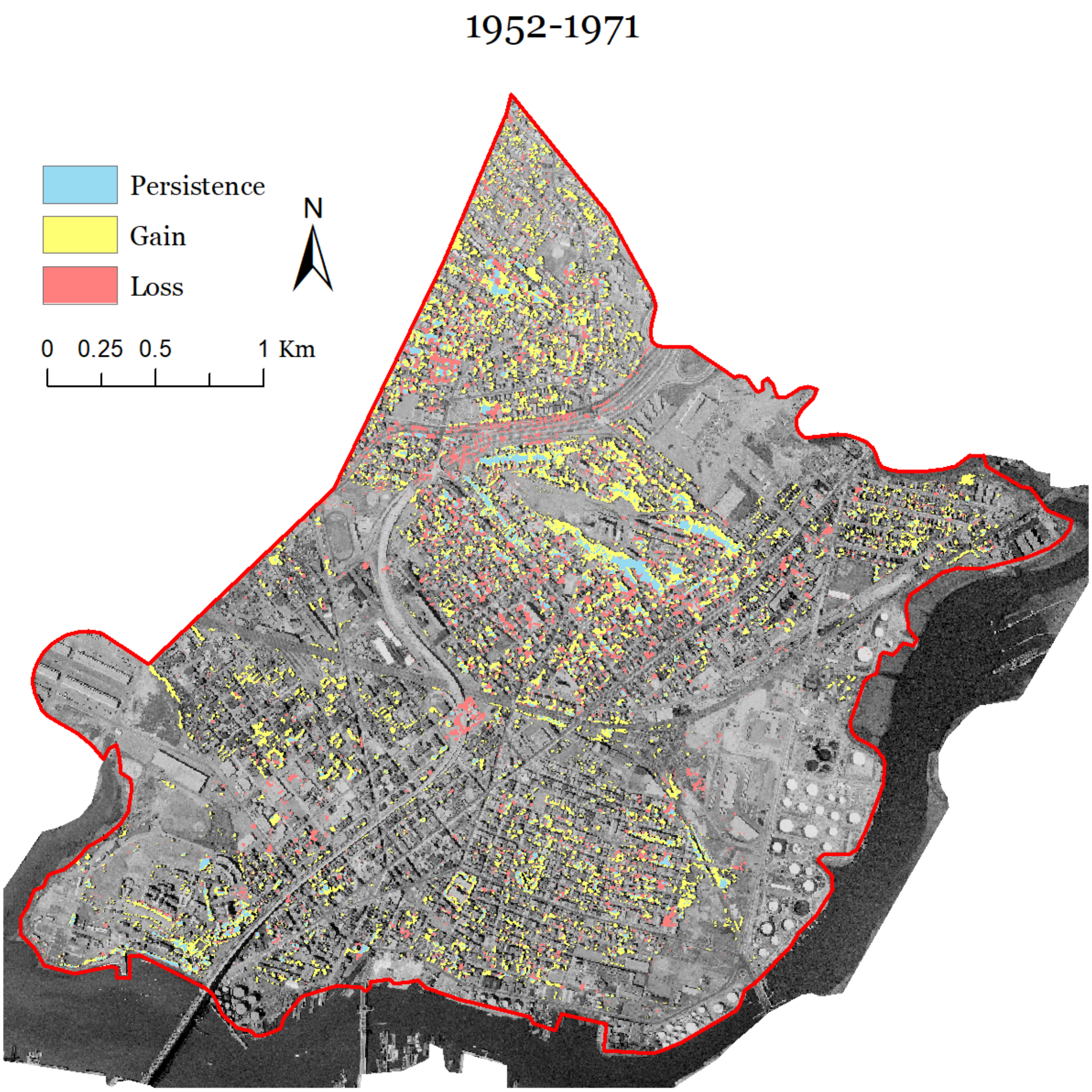

Figure 2-2. Urban tree canopy gain, loss and persistence in Chelsea from 1952-1971 with 1971 imagery as the background.

1971-2003

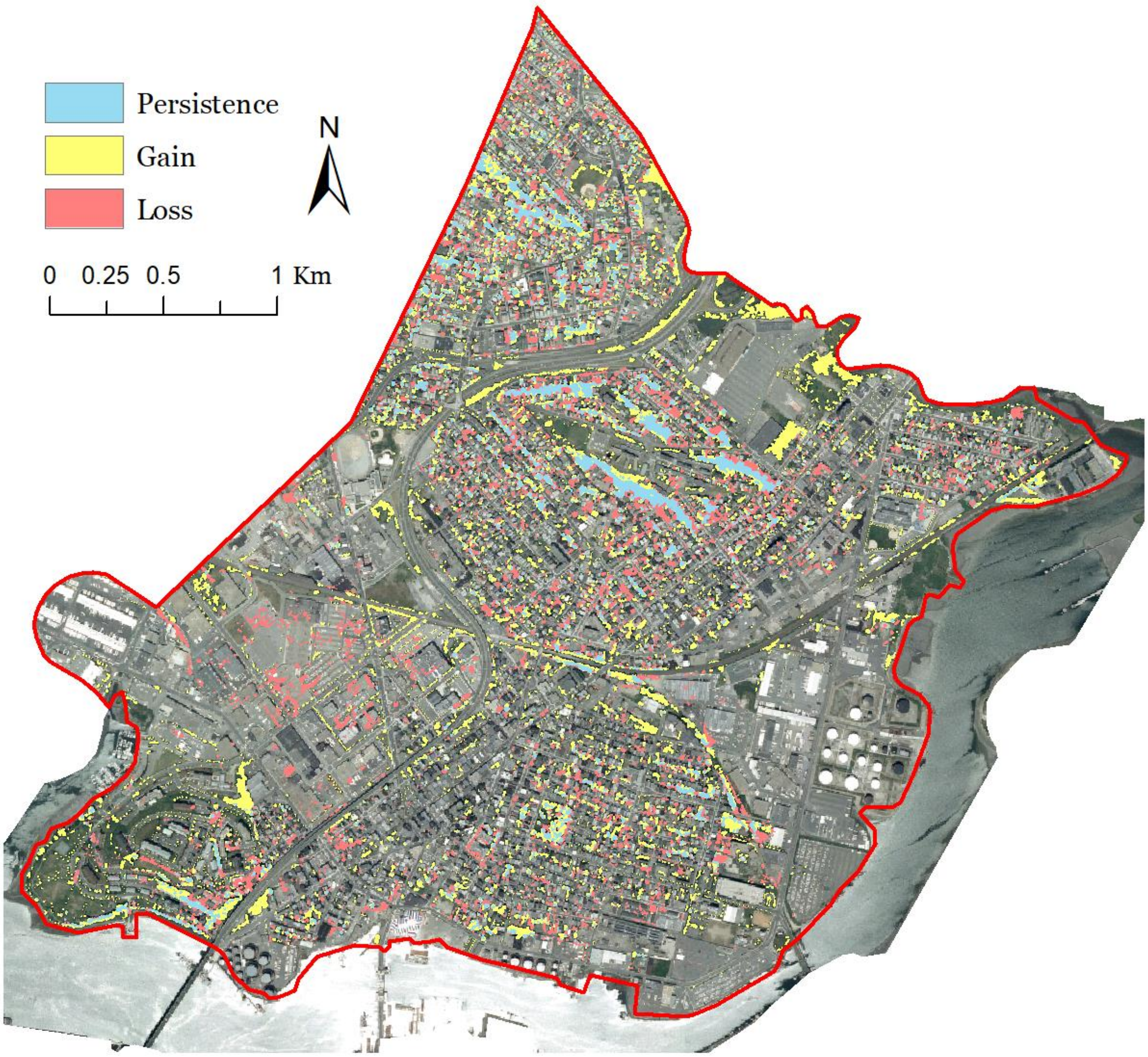

Figure 2-3. Urban tree canopy gain, loss and persistence in Chelsea from 1971-2003 with 2003 imagery as the background.

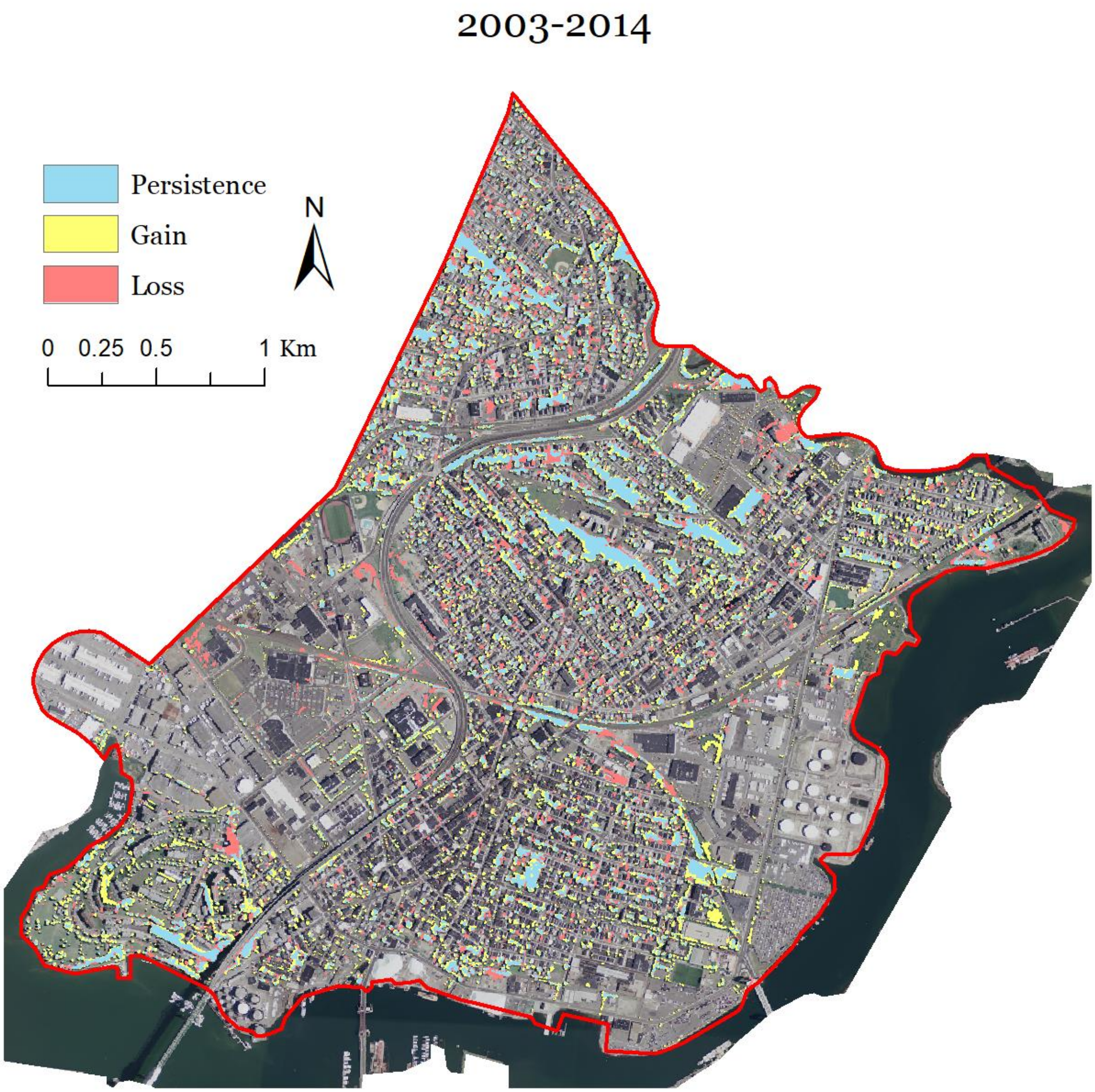

Figure 2-4. Urban tree canopy gain, loss and persistence in Chelsea from 2003-2014 with 2014 imagery as the background.

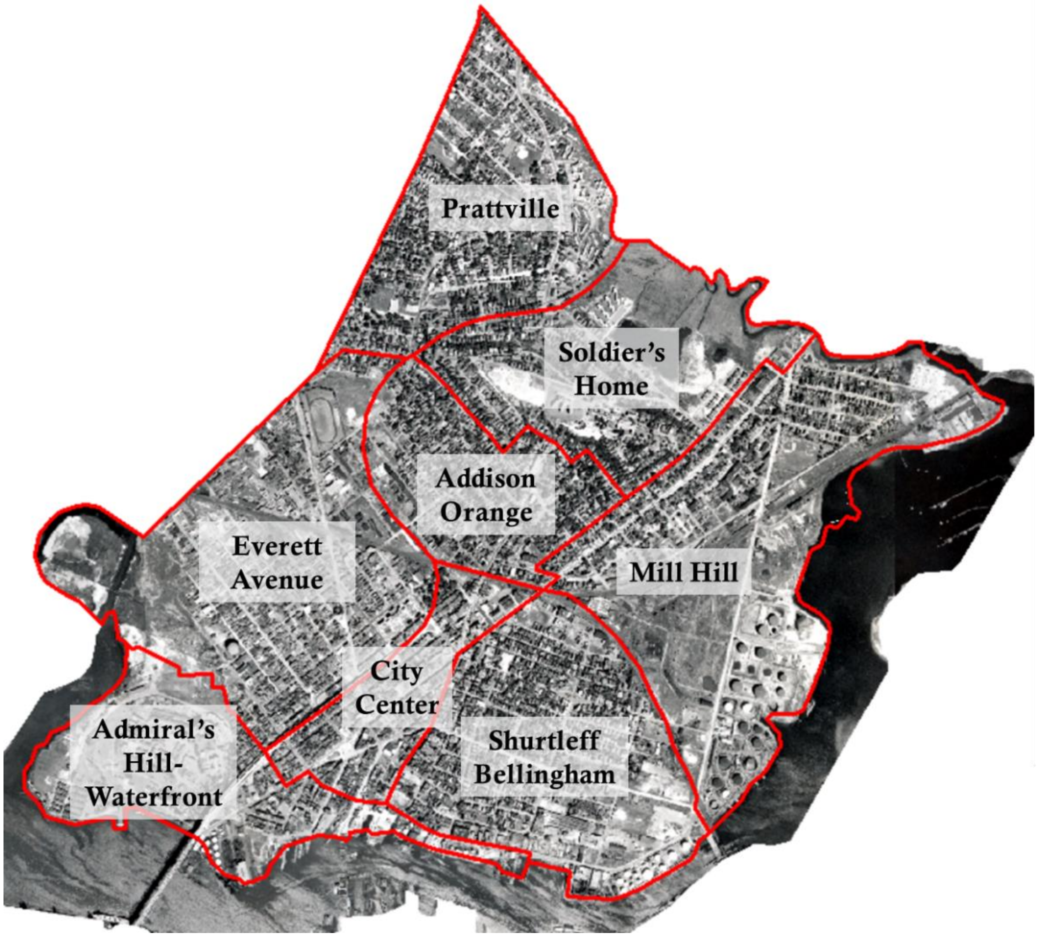

Figure 2-5. Chelsea neighborhoods with 1952 imagery as the background.

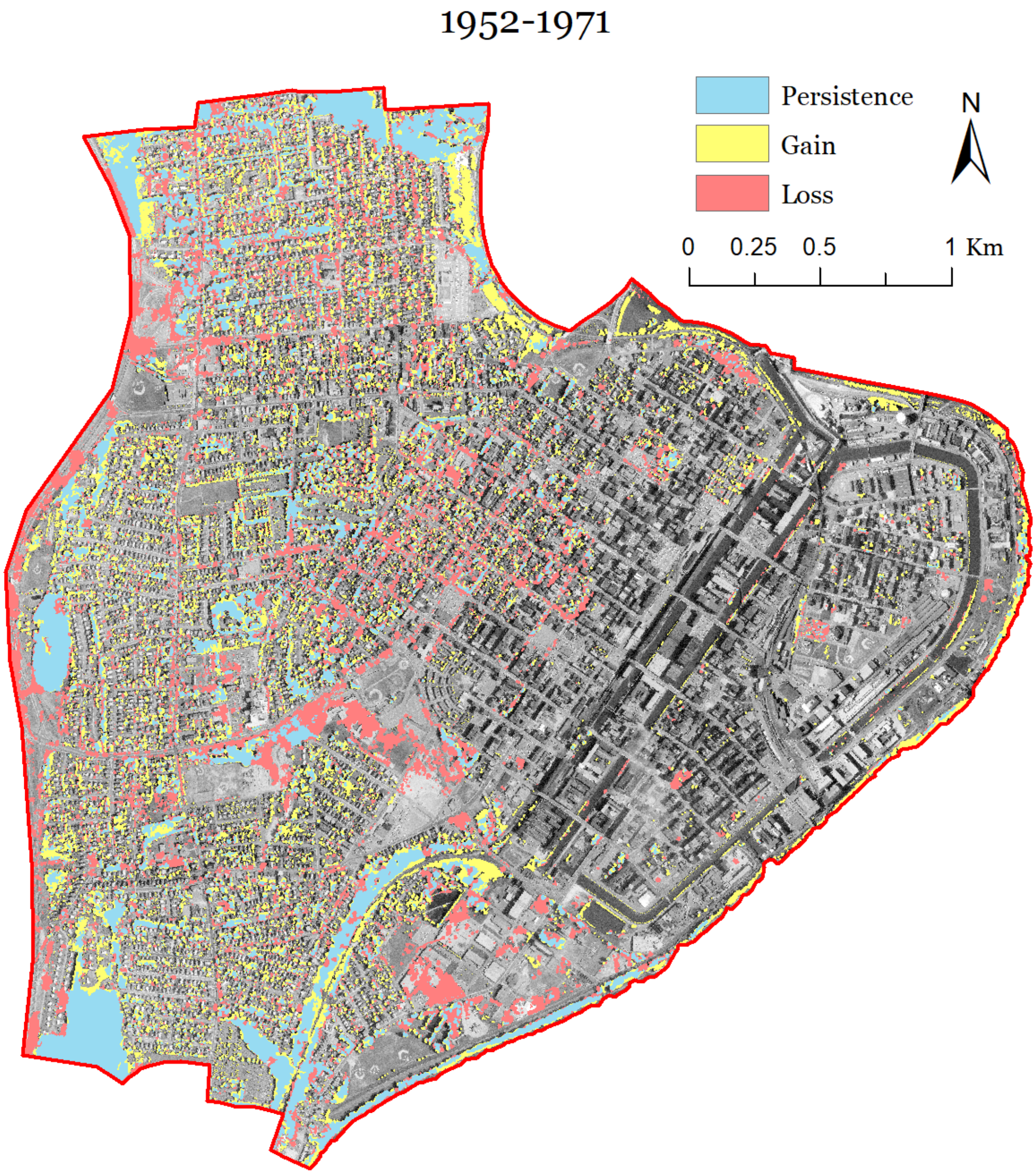

Figure 2-6. Urban tree canopy gain, loss and persistence in Holyoke from 1952-1971 with 1971 imagery as the background.

Figure 2-7. Urban tree canopy gain, loss and persistence in Holyoke from 1971-2003 with 2003 imagery as the background.

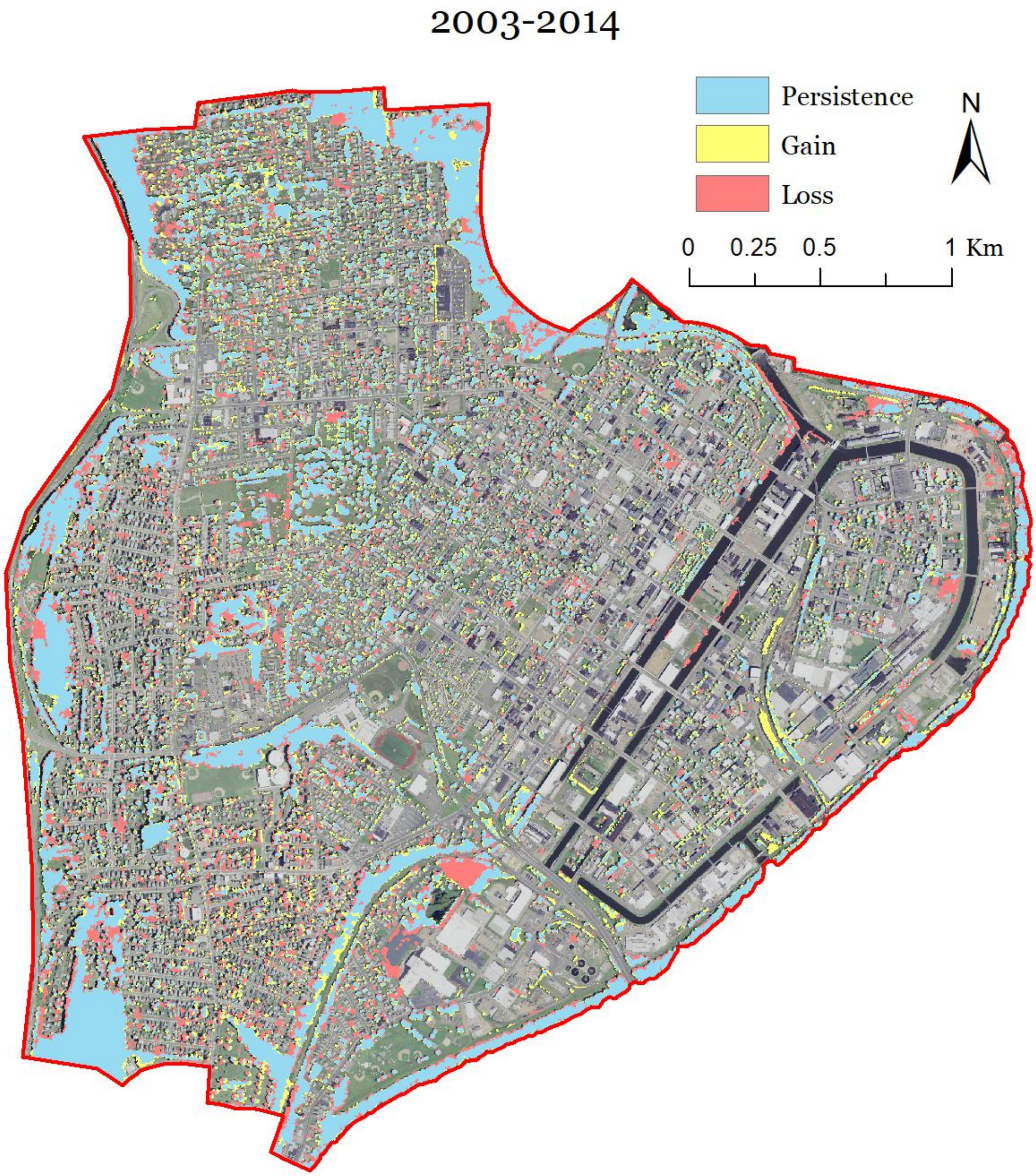

Figure 2-8. Urban tree canopy gain, loss and persistence in Holyoke from 2003-2014 with 2014 imagery as the background.

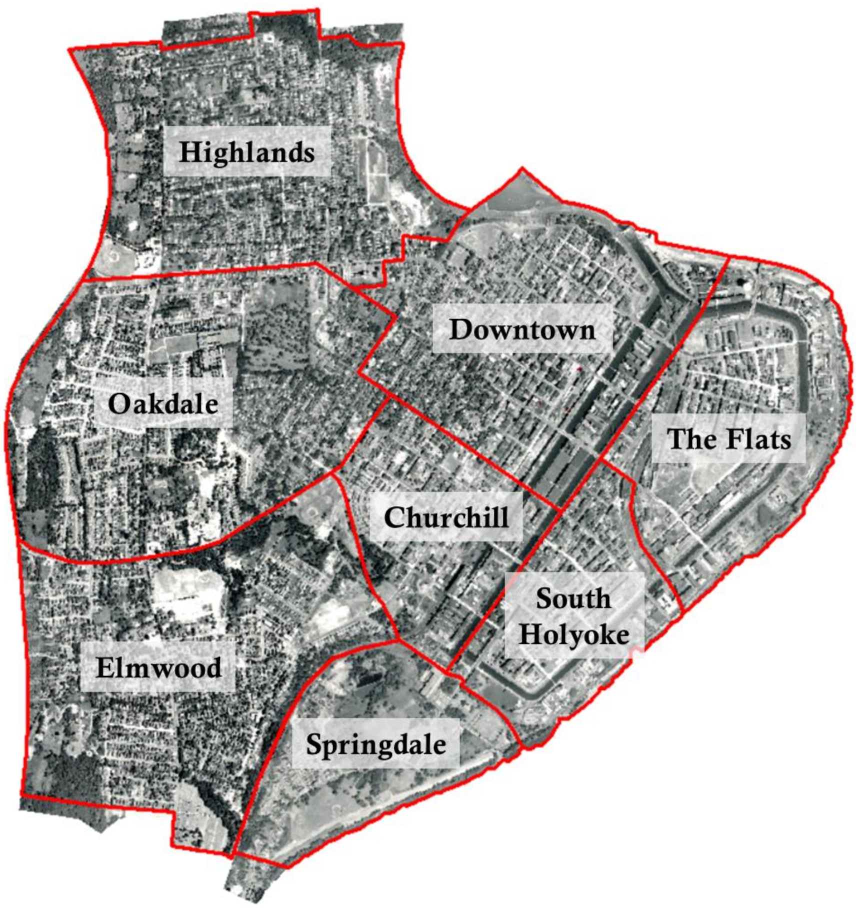

Figure 2-9. Holyoke neighborhoods with 1952 imagery as the background.

Chapter 3: How does a tree planting initiative affect air and land surface temperature? A case study in Holyoke, Massachusetts[2]

Abstract

The urban heat island (UHI) impacts human health and comfort, exacerbates energy and water usage, and can severely impact low-income communities that lack access to cooling resources. Cities have turned to urban greening, especially in the form of tree planting, to mitigate the UHI and to respond to climate change. We explore the potential cooling effects of juvenile trees planted through a tree planting initiative and explore the temperature variation between Landsat-derived surface temperature and in-situ sensor-derived air temperature in Holyoke, MA (USA). In-situ sensors showed modest, but significant, air temperature cooling effects (up to -0.087°C per tree) from the juvenile trees on the hottest days of the year between 2017-2021, with dense tree planting (3 trees per acre) offering aggregated cooling effects. Inconclusive results were found using a point-based or a 100 m buffer-based approach to analyze cooling effects of juvenile trees on land surface temperature. These two LST approaches to measuring cooling effects of juvenile trees may be more appropriate as the trees mature, but in-situ air temperature measurements can readily demonstrate the cooling effects of juvenile trees. The benefits of juvenile trees should be more readily considered by tree planting initiatives and urban forest managers when proposing tree planting projects and shared with residents who can receive and benefit from the trees earlier than previously expected.

[2] Healy, M., Rogan, J., Geron, N., Martin, D.G., and Roman, L.A. In preparation for *Urban Forestry and Urban Greening*.

1 Introduction

The urban heat island (UHI) effect causes urban areas to be warmer than surrounding rural areas, with larger differences observed at night (Voogt and Oke, 2003). This difference is due to impervious surfaces, such as brick, concrete, and asphalt, as well as buildings made of similar materials, absorbing radiation during the daylight and emitting the energy as heat throughout the evening while the surfaces cool (Oke, 1982). The intensity of the UHI is attributed to land-cover composition/change, tree canopy cover, and greenspace configuration in urbanizing/urbanized areas (Lo and Quattrochi, 2003; Wang et al., 2016; Elmes et al., 2017). Expansion of urban areas globally is projected to increase by almost 1 million km^2 by 2050, resulting in potential average summer daytime and nighttime warming of air temperature by 0.5–0.7°C, with up to 3°C in some locations, as open space is developed into various types of impervious surface (Huang et al., 2019). Increased air temperatures due to high levels of impervious surface have led to UHI reduction becoming an important goal in US cities (O'Neill et al. 2010; Silvera Seamans, 2013), especially with the stress of regional climate warming that could compound the UHI effect (Karmalkar and Bradley, 2017). Recent models project that the northeastern US will warm by 3°C when global warming reaches 2°C (Karmalkar and Bradley, 2017). The projected warming in the northeastern US places it as potentially the fastest warming region in the contiguous U.S and makes the urban areas in the northeast particularly vulnerable to increased UHI effects.

UHI impacts human health and comfort (Milošević et al., 2017; Taleghani et al., 2015) while also exacerbating energy and water usage during the hottest months (Akbari et al., 1997; Akbari et al., 2001), which can severely impact low-income communities that lack access to cooling resources and increase heat-related health risks (Harlan et al., 2007; Heaviside et al., 2017). People who cannot meet their energy needs are in energy poverty—the inability to maintain suitable housing temperatures and other electrical needs (Tsilini et al., 2015). As climate change exacerbates energy poverty, increased greenspace and green infrastructure—especially trees—provide part of a solution to mitigate the UHI, alleviate energy poverty, and provide additional health, economic and environmental benefits (Norton et al., 2015; Morzuch, 2013; Nowak and Dwyer, 2007; Moody et al., 2021). Specifically, increasing tree canopy cover can help mitigate UHI through shading, increased evaporative cooling, and less impervious surface cover (Akbari et al., 2001; Nowak and Dwyer, 2007; Ko, 2018; Ziter et al. 2019).

UHI can be tracked through the use of air temperature sensors providing a diurnal range of responses, but these data are not spatially continuous and therefore, it is impossible to map temperature variability with precision across a large urban area (Rogan et al. 2013). Remotely sensed thermal infrared data can be used to study the Surface Urban Heat Island (SUHI) effect through Land Surface Temperature (LST) products. SUHI, which is part of the UHI, is impacted by the increase of LST which can be caused by a lack of urban tree canopy, building materials, etc. (Voogt and Oke, 2003). The SUHI is impacted more by the surface of land-cover types and the materials they are made of, while the UHI is made up of the heating or cooling effects of the different land-cover types, combined with surface air-mixing (urban air layers, humidity, and particulate matter) (Zhou et al., 2018; Arnefield, 2003). The SUHI can be mapped, quantified, and modeled across entire cities or regions showing spatial trends in LST (Huang et al., 2011; Zhang et al., 2017; Elmes et al., 2017). Pairing the precision of urban air temperature sensors with LST imagery has not been well researched, but it has the ability to demonstrate spatial trends that can identify locations that experience prolonged heat exposure with detailed temperature profiles (Elmes et al., 2020; Shandas et al., 2019). Understanding UHI temperature trends, both air and LST, are important for human health (Heaviside et al., 2017; Avashia et al., 2021) and environmental justice community impacts (Hoffman et al., 2020).

Heavily built-up areas, especially in northeastern US post-industrial cities, continue to have their past legacies exerted on the landscape (Healy et al., 2022b; Hoffman et al., 2020). Their built histories are important factors contributing to how and where the UHI is most intense, especially for racial and ethnic minorities who have historically inhabited these areas as result of racist US housing policies (Jesdale et al., 2013; Hoffman et al., 2020). In order to mitigate UHI effects and to respond to climate change, cities have turned to urban greening, especially in the form of tree planting (Larsen, 2015; McPherson et al., 2011). Large-scale urban tree planting is typically managed by tree-planting initiatives (TPIs) which have a long history of operating in the US (Eisenman et al., 2021), although there can be inequitable distribution of new trees through these programs (Watkins et al., 2017). TPIs typically focus their goals on the number of trees they can plant and the overall increase in tree canopy cover as a result (Nguyen et al., 2017; Eisenman et al., 2021), looking forward to the future benefits of recently planted trees (Solecki et al., 2005; McPherson et al., 2011; Moody et al., 2021). Shading and cooling benefits are generally expected to materialize decades after planting (Ko et al., 2015). However, little research has tracked the benefits, especially that of cooling, of trees planted by TPIs across time (Moody et al., 2021). Currently there is a gap in the research concerning when young or juvenile cohorts of trees begin to produce detectable ecosystem services, specifically cooling, as juvenile trees are overlooked for their ability to produce benefits until they have reached maturity. Identifying when juvenile trees begin to produce ecosystem services may help TPIs and other organizations justify expenditures for tree planting and better encourage residents to plant and steward trees.

In this study, we investigate the measurable effects that recent tree planting (i.e., starting in 2014) has had on residential air temperature and LST, as well as explore the relationship between air temperature and LST in state-designated environmental justice neighborhoods. A dense, temperature sensor network (n = 32) was installed by the Massachusetts Department of Conservation and Recreation (DCR) in 2014 to monitor hourly temperature in Holyoke, Massachusetts after the began its participation in the state-funded and -managed TPI, Greening the Gateway Cities (Breger et al., 2019; Healy et al., 2022a). Specifically, we address the following questions: 1) What are the cooling effects of juvenile trees on air temperature and LST; 2) how does the relationship between LST and air temperature vary and how does urban tree canopy and impervious surface affect this relationship? We propose that the juvenile trees planted by the GGC will have measurable cooling effects. Moody et al. (2021) measured ecosystem services for juvenile trees only a few years after planting and showed that the majority of benefits came from energy savings. Therefore, the cooling effects from juvenile trees should be capable of measurement as well.

2 Methods

2.1 Study Area

Located in western Massachusetts along the Connecticut River, Holyoke is a post-industrial city with a population of 38,238 (US Census, 2021). It is located in a humid, continental climate and covers 59 km² with a forested state park covering nearly half the area. Holyoke was one of the first planned industrial communities in the US, with many factories and mills located within the canaled industrial area (Curran, 1960). Due to the globalization of manufacturing, many factories and mills closed between the 1960s and 1990s (Jacobson-Hardy and Weir, 1992) causing a large outmigration from Holyoke and an influx of new immigrants, primarily from Puerto Rico (Borges-Méndez, 2007). The history of Holyoke's urban tree canopy has been well documented, with the most built-up areas consistently having the lowest tree canopy cover across several decades (Healy et al., 2022b), which is representative of other post-industrial cities in forested biomes (Healy et al., 2022b, Roman et al., 2021). As a way to remedy the low tree canopy cover and help residents save on utility bills, a state-funded and -managed tree planting initiative was started in Holyoke in 2014, as part of larger multi-city effort called Greening the Gateway Cities (GGC) (Healy et al, 2022a; Geron et al., 2022; Breger et al., 2019). The TPI identifies zones for planting based generally on median household income below the state average, average educational attainment level (Bachelor's degree or above) below the state average, high percentages of renters, low levels of tree canopy, high levels of impervious surface (Figure 3-1) and plants primarily on residential property (Figure 3-2). Between 2014 and 2021, the Greening the Gateway Cities TPI planted over 2,000 trees within Holyoke's planting zone.

2.2 Data

2.2.1 In-situ Air Temperature Sensor Data

A network of Honest Observer by Onset (HOBO) Pendant® MX air temperature data loggers were used to record in-situ, hourly temperature (C°) measurements. HOBO sensors record air temperature at a precision of 0.04°C (0.072°F) and store that data at a user defined interval (ONSET, 2021). The DCR installed 50 HOBO sensors in Holyoke in the spring of 2016 to measure the long-term temperature impacts of the GGC trees. The HOBO sensors were placed in two zones: the GGC planting zone (PZ) and areas outside the PZ, which we call the control zone (CZ) (see Figure 3-1). The PZ is the area where the GGC planted trees, primarily in residential neighborhoods (both single family and multi-family). The CZ consists of primarily residential neighborhoods that have a variable levels of tree canopy cover and impervious cover, but overall, the CZ has higher canopy cover and lower impervious cover than the PZ (Table 3-1). The HOBO sensors, with a radiation shield, were mounted on fences or street signs, 1.5 m above the ground by DCR staff, and have been serviced yearly (data downloaded, battery replaced, sensor calibrated). In the beginning, 25 HOBO sensors were placed in each zone, but over the subsequent years several HOBO sensors have been lost to theft or failure, with 32 HOBO sensors remaining in 2021 (PZ: n = 14; CZ: n = 18). HOBO sensor locations were also aligned to match up with Landsat thermal band 90 x 90 m pixel footprints.

The air temperature data were converted from a HOBO proprietary format to CSV for aggregation and analysis in R. Sensor data were stored by year and required preprocessing in preparation for aggregation and analysis. Each yearly dataset had an hourly timestamp, the temperature at that timestamp, and the sensor ID number. We added a separate date and time field for analysis purposes and subset each dataset to the summer of each of year (01 June – 31 August) as well as adding a field based on their location—inside the PZ or in the CZ. Hourly sensor datasets were then aggregated, by zone, into study period datasets from 2017-2021 based on their location and used to create daily statistics (mean, min, max, range) datasets. Air temperature at the time the Landsat satellite overpasses, approximately 10:25 am, was calculated by averaging the hourly temperatures of 10 am and 11 am for the purpose of comparing Landsat surface temperature imagery with air temperature collected by the HOBO sensors. Some HOBO sensors had data gaps due to malfunctioning batteries or other errors. Regardless of the gaps, all available temperature data in the time series was used in this analysis.

The daily temperature datasets were combined with 2016 land-use/land-cover data that was derived from the 2016 USDA National Agricultural Imagery Program multispectral imagery by MassGIS, the Massachusetts Bureau of Geographic Information (see MassGIS, 2019). From the 2016 land-use/land-cover dataset the percent canopy cover (PCC) and percent impervious surface (PIC) were calculated for a 100 m buffer around each sensor and added to the daily temperature datasets, as well as data for land-use and land-cover type at the temperature sensor point, and the number of trees planted by the GGC between 2014-2017 within the 100 m buffer. A 100 m buffer was used because prior research has shown areas of similar and larger size are representative of local scale effects (e.g., shade from nearby forested patch, impervious surface heating up in the afternoon) (Ziter et al., 2019; Alonzo et al., 2021). The buffer area can contain differing levels of urban tree canopy cover, greenspace and impervious surface cover which impact both the air and surface temperature (Ziter et al., 2019; Shandas et al., 2019). The buffers may also contain a small amount of tree canopy produced by the GGC trees since the TPI started in 2014.

2.2.2 Landsat Surface Temperature Data

A combination of twenty-one Landsat C2 US Analysis Ready Data (ARD) LST images for Landsat-8 and Landsat-7 were used for analysis, from 13 June 2017 to 27 August 2021 in tile grid horizontal 29 and tile grid vertical 6 (Table 3-2). Landsat ARD imagery are preprocessed data products from the Landsat archive that are tiled and corrected (geometric, top of atmosphere, and atmosphere) for scientific monitoring and application (see Dwyer et al., 2018). The ARD Surface Temperature product was derived using a single channel algorithm on the thermal band for Landsat-8 (Band 10) and Landsat-7 (Band 6) (see Cook et al., 2014), and produced LST estimates over land with a root mean square error (RMSE) of approximately 2.2°K (Dwyer et al., 2018; Duan et al., 2021). Each image was scaled with a multiplicative and an additive factor to convert the values to degrees Kelvin and were later converted to degrees Celsius. Images were selected based on visual inspection for cloud free coverage of the study area for the summer (01 June – 31 August) of each year (2016-2021). While the failure of the scan-line corrector on Landsat-7 causes data gaps in the imagery, the vast majority of the image data are usable. If a temperature sensor was contained within the data gap of a Landsat-7 image, then an NA value was recorded.

Two approaches were used to collect LST values from the imagery: point-based and buffer-based. The point-based approach recorded the LST value at each temperature sensor location while the buffer-based approach averaged the LST values within a 100 m buffer of each temperature sensor. Both approaches were used for analysis to understand and measure the differences between using values from only one pixel which represents the immediate area around the HOBO sensor and an averaged value of several pixels within the 100 m buffer which can represent the local land-cover and their land surface temperatures.

2.3 Analysis

2.3.1 In-Situ Air Temperature Analysis

We analyzed residential air temperature measurements from the PZ using the available HOBO sensors, the number of which varied by year due to sensor error or data collection issues (see the n row on Figure 3-5). First, we examined the cooling effects caused by juvenile trees on the daily average and daily maximum temperatures throughout the entire study period. We used multivariate regression to explore the associations between the daily average and daily maximum temperature across the entire study period and the variables PIC, the PCC, the number of trees planted by the GGC within 100 m of each PZ HOBO sensor, and the number of years since planting as an approximate estimate of the age of the trees. The PIC and PCC were used to control for urban form and existing tree canopy cover respectively. Next, we used linear regression to assess the relationship between each individual variable and the daily average or daily maximum temperature across the entire study period.

Second, we examined the cooling effects caused by juvenile trees on the hottest day of each year between 2017-2021. This approach was utilized as recent research has shown that this is when the trees would have the largest, most measurable effect (Wang et al., 2019; Rosenzweig et al., 2006). We used linear regression to analyze the relationship between the maximum daily air temperature on the hottest day of 2017 to 2021 and the number of trees planted by the GGC to investigate the cooling effects of the juvenile trees. Next, we furthered our analysis with multivariate regression to find associations between the maximum daily temperature of the hottest day of 2017 to 2021 with the PIC, the PCC, and the number of trees planted by the GGC within 100 m of each PZ HOBO sensor.

Lastly, we examined the differences in temperature between the PZ and the CZ. We used a t-test for each year (2017-2021) to determine if the maximum daily temperature of the hottest days were significantly different between the PZ and the CZ. Next, we investigated the diurnal temperature range across a number of HOBO sensor sites based on their land-cover attributes in order to reveal temperature differences between the PZ and CZ.

2.3.2 Landsat Surface Temperature Analysis

We sampled the variation of residential air temperature to LST across all of the HOBO sensors sites (Table 3-1). The land-cover composition of a sample of four of the HOBO sensors can be seen in Figure 3-3. The PZ sensor sites have a wider range of PIC than the CZ, but similar ranges of PCC. The air temperature and LST (point-based and buffer-based) observations were compared using ordinary least squares regression to find the adjusted coefficient of determination (R^2). The mean absolute error (MAE) and RMSE were also calculated, following Elmes et al. (2020). Use of MAE and RMSE are important metrics that express average model prediction error, with lower values representing higher accuracy (Chai and Draxler, 2014). MAE considers errors with equal weighting while RMSE gives a high weight to large errors. Like Elmes et al. (2020), these measurements were compared to explore the spatiotemporal thermal patterns of the CZ and PZ.

Following the same type of analysis in Section 2.3.1, we analyzed LST in the PZ at each of the 14 sensor sites using multivariate regression to find associations between the LST (point-based and buffer based) on the hottest day 2017-2021, with the percentage of PIC and PCC, and the number of GGC trees within 100 m of each PZ sensor site. The hottest days chosen for the LST analysis did not match all the days from the analysis in Section 2.3.1 because the acquisition period of Landsat did not always correspond with the hottest days. Other hot days were chosen from the LST imagery, and they typically were only different from the hottest days by 1-3C°.

3 Results

3.1 In Situ Air Temperature and Tree Cooling Effects

We used multivariate regression on the entire dataset and found that the number of trees planted, PIC and PCC were significantly influencing, albeit modestly, the average and maximum temperature during the study period and had a very low p-value (< 1.22 e-08) (Figure 3-4). The number of trees planted had a negative coefficient for both average and maximum temperature (-0.012 to -0.043), and PIC and PCC had positive coefficients. When we used linear regression for each individual variable, we found that the number of trees planted had a significant negative relationship (-0.031, p-value: 6.53e-09) on the maximum temperature while the PCC had a significant positive relationship (0.015, p-value: 0.03) (Figure 3-4). For the average temperature we found that years since planting had a significant positive relationship (0.129, p-value: 8.35e-05) (Figure 3-4).

We examined the linear relationship between the maximum daily air temperature on the hottest day of the year and the number of trees was consistently negative over the study period, though the relationship had low R^2 values (i.e., 0.11-0.47) (Figure 3-6). This finding suggested that the more planted trees within a 100 m buffer of an in-situ sensor, the greater the observable cooling effect. To further investigate the relationship between the maximum daily air temperature on the hottest day of the year and the number of trees planted, we used multivariate regression and found a significant negative coefficient between the air temperature and the number of trees planted in four of the five years (2018-2021), culminating in a negative coefficient of -0.087 on the latest date of 29 June 2021 (Figure 3-5). This means that for each tree planted, there was an observable decrease in temperature, albeit modest, by 0.087 °C. A significant positive relationship between the air temperature and PIC and PCC was present in two of the five years, meaning that the PCC and PIC were modestly warming the air temperature (Figure 3-5). All multivariate regression coefficient values varied slightly, but a negative cooling trend is associated with the numbers of trees planted, over time. The data from 19 July 2020 represent the maximum and minimum coefficient values and this may be due to a low sample size (n = 6). The overall model metrics improved over time, with assumed tree canopy growth, with 2019 and 2021 both having adjusted R^2 values greater than 0.5, and significant p-values that subsequently decreased (< 0.01)—becoming increasingly significant over time. We used linear regression for each individual modeled variable, and found that none of the variables were significant or had low p-values (Figure 3-5). However, the number of trees planted variable was consistently negative over the study period, and it had the lowest p-values of all the modeled variables (0.09-.0.23).

A t-test was used to identify if there was a significant difference in the maximum daily temperatures on the hottest day of each study year (2017-2021) between the PZ (n = 14) and the CZ (n = 18). The t-test found that none of the maximum daily temperature values of the hottest days were significantly different between the PZ and CZ. The p-values from each year ranged from 0.39-0.99, well above a 0.1 significance level. The p-values were also variable from each study date and did not show a trend of decreasing over time. Next, we investigated individual 6 HOBO sensor sites to understand their diurnal temperature profiles on the hottest days of each year of the study period (Figure 3-8). We compared sites that have similar landcover types. For example, the top row of diurnal temperature profiles are sites where PCC is low, approximately 10%, and PIC is in a similar range (58%-70%). Each site has the number of trees that were planted within 100 m—one with a high number and one with a low number—and the last site in each row which is a sensor from the CZ. The top row PZ sites show greater variation in peak temperature than the CZ site. The five-year average maximum temperature for the PZ sites are also lower than CZ. The bottom row of diurnal temperature profiles are from sites with high PCC (31%-40%) and moderate PIC (32%-44%). The bottom row PZ sites also show greater variation in their peak temperature than the CZ site, but their five-year average maximum temperature is higher than the CZ site.

3.2 Comparison of LST and Air Temperature; LST and Tree Cooling Effects

We analyzed the spatiotemporal thermal patterns of all the HOBO sensor sites through the comparison of the air temperature and LST (point-based and buffer-based) measurements. Figure 3-7 provides an illustrative example of the offsets between air temperature and LST over the study period (2017-2021) of four different sensor sites. Some air temperature measurements were missing due to data collection issues, and these dates were removed from further exploratory analysis. Temperature anomalies, or times where the LST was lower than the air temperature at the time of measurement, can be seen occurring on 23 July 2017 and 5 July 2019. These anomalies were investigated for their cause as they can happen due to the land surface being wet from rain. Precipitation data was examined from the National Weather Service weather station at Chicopee Falls/Westover Air Force Base (approximately 3 miles from the PZ) but there was no evidence of rainfall occurring within 48 hours of the two dates in question. The temperature anomalies were also removed from further exploratory analysis.

Over the study period, the PZ sensor sites MAE and RMSE between the Landsat LST and the air temperature averaged 9.85°C and 10.29°C, respectively, while the CZ sensor sites averaged 7.65℃ and 8.07℃, respectively (Table 3-1). The lower MAE and RMSE values typically came from sensor sites that have high PCC (>30%), while the higher MAE and RMSE values came from sensor that have low PCC (<15%). This also occurs at the zone level with the CZ having lower offsets while having a higher average PCC, and lower average PIC than the PZ.

The MAE and RMSE average values for both point-based and buffer-based LST approaches are similar (Table 3-1), with higher average differences occurring in the planting zone.

A significant relationship was not found between the LST and the number of trees planted across the study period for either the point-based (Table 3-3) or buffer-based (Table 3-4) approaches. The point-based approach (Table 3-3) did not have any significant coefficients, but the model did seem to perform relatively well with adjusted R^2 values greater than 0.53, except in 2021 which had an adjusted R^2 value of 0.36. The model p-values were significant at the 0.05 level from 2017-2020 with a higher p-value in 2021, and trend upward over time, indicating that the model as a whole explains some of the LST variability, but the model coefficients do not. The buffer-based approach (Table 3-4) had significant coefficients for PIC in 2017 and 2018, but there were no other significant coefficients. The buffer-based model performed better across the entire study period compared to the point-based model. This may be because the buffer-based model incorporates the average LST of the 100 m buffer, which better represents the local surface temperatures. The adjusted R^2 value was higher (>0.70) across all years, except for 2020 (adjusted R^2 = 0.52). The p-values were all significant and consistently low throughout the study period.

4 Discussion

We found that trees planted as part of the GGC TPI have a measurable and significant effect on local maximum air temperatures, and on the hottest days during the summer months 4-7 years after planting, as trees are newly established and growing (Figure 3-4; Figure 3-5). There also seems to be some collinearity in the predictors as PCC displays a slight warming effect, but when the predictors are modeled separately, these effects are lower and typically not significant.

When looking at how the predictors influence average temperature and maximum temperature, it seems that average temp is better explained by years since planting whereas maximum temp is better explained by number of trees planted (Figure 3-4). The data show that as time progresses the average temperature is increasing, and at a larger rate than the maximum temperature. More research is needed to uncover this trend as it deals with topics that are not covered by this research. Overall, a cooling effect of the individual GGC trees can be seen on the maximum temperature and on the hottest days, albeit a modest one, and it is possibly slightly negated by the positive PCC temperature association. While on its own, this cooling effect is small (approximately -0.031°C, Figure 3-4), it has the possibility of being amplified through dense planting. For example, sensor site 19226 had 35 GGC trees planted within a 100 m radius (Figure 3-8), and the diurnal chart is showing greater variation of peak temperature on the hottest days that could signifying cooling, especially when compared to CZ sites. This mirrors findings in other studies that have found cooling related to tree canopy cover at similar buffer sizes (Ziter et al., 2019; Alonzo et al., 2021), but that research only considered canopy cover as a whole, and not the effects of juvenile trees, or tree planting density. Moody et al. (2021) calculated ecosystem services of juvenile trees and found that a tree planting density of 2-3 trees per acre achieved significant energy savings. Our findings support this 2-3 trees per acre tree planting density (3 trees per acre matches the average number of trees planted within a 100 m of a HOBO sensor, see Table 3-1) as it can begin to mitigate local air temperature on hot days only 4-7 years after planting. However, this outcome is highly dependent on tree survivorship. Since 2017 there has not been a tree heath survey to verify the survivorship of the GGC trees in Holyoke (Breger, 2019), which can greatly impact the future ecosystem services of the trees (Ko et al., 2015;

Moody et al., 2021). More research is needed to understand how juvenile tree mortality impacts expected cooling effects of TPIs.

We did not find significant air temperature differences between the PZ and CZ using the maximum air temperature on the hottest days of each year of the study period. This suggests that both zones are more similar in air temperature than previously thought. We believed that the higher average PCC and lower average PIC of the CZ would lead it to have significantly cooler temperatures than the PZ. Not finding significantly different air temperatures may be due to the saturation of heated air at the sensor sites on hot days and their physical location (sensors in both zones placed in full sun may negate some of the neighborhood cooling effects), proximity of the zones to one another, and small amounts of change taking place in these neighborhoods and other external factors that are not covered in this research (e.g., municipal tree maintenance, resident yard modifications). We furthered investigated temperature differences between the zones by comparing individual sensor sites which can better represent the neighborhood cooling effects. We found that PZ sites with high density of trees planted had more variable peak temperatures on hot days over the study period than sites in the CZ (Figure 3-8). The peak temperatures in the PZ also look to be decreasing over the study period, while the CZ sites are more stable. These findings are promising as the CZ has more residential land use with more existing tree canopy cover than the PZ, but more research and data are needed to confirm these changes are taking hold permanently.

By analyzing all the temperature sensor sites, we observed that PCC reduced the offset between LST and air temperature measurements by approximately 2°C (Table 3-1). Similarly, Elmes et al. (2020) reported that sites of concentrated, high PCC ($\geq$ 47%) showed significantly lower LST and air temperature variability in Worcester, Massachusetts. The MAE and RMSE values in this study were larger than Elmes et al. (2020) because the PCC of the highest canopy sites was ≤ 37%, and there were very few instances of concentrated canopy cover over the study area. Overall, the temperature sensor sites with higher PCC (e.g., sensor sites 11865 (PZ) and 10574538 (CZ), see Table 3-1) had less temperature variability, by 2°C or more, when compared to the temperature sensor sites with lower PCC (e.g., sensor sites 19226 (PZ) and 10574553 (CZ), see Table 3-1). This suggests that the density of tree canopy in a small area can have a significant temperature mitigation effect.

Using either a point-based or buffer-based approach to analyze LST resulted in negligible differences between the LST values themselves, their MAE and RMSE values, and linear fit. The biggest difference was observed when either the point-based or buffer-based LST approach was used in the multiple regression model to analyze the influence of the number of trees planted by the GGC TPI, the PCC, and the PIC on LST. The buffer-based model, as a whole, had higher adjusted R^2 values and lower p-values than the point-based model, even though none of the coefficients were significant. This may be the result of conducting this analysis utilizing a buffer distance that better matched the resolution of the satellite sensor, but if this was the case, we would also expect the buffer-based MAE and RMSE values to be lower than their point-based values. More research is needed to understand the differences of point-based or buffer-based LST use in modeling.

Other studies have used LST for investigating the cooling effects of trees and greenspaces typically through the lens of land-cover change and the density of urban tree canopy (Rogan et al., 2013; Elmes et al., 2017; Jung et al., 2021) but not exploring the effect of individual trees. We show the cooling effects of juvenile trees using air temperature data, but we were not able to report cooling effects using satellite LST. While it is broadly known that urban forests with mature trees and larger crowns produce a strong cooling signal (Speak et al., 2020), there is an unknown tree age or size threshold at which cooling effects can be detected using LST approaches. Because of the extreme heterogeneity of urban areas and fragmented distribution of trees and greenspaces within them, LST is explained more by the fractional coverage of different land-cover types (Zhang et al., 2009; Zhou et al., 2014). Therefore, more consideration of LST variability is needed, especially in urban areas, when utilizing LST products that have an overpass time that is not ideally suited for LST imagery. Other factors that my influence satellite-based LST values include aspect, azimuth, surface emissivity, humidity, and aerosols (dust, haze). More research is needed to understand these factors in urban settings and how they influence different LST products.

Our study investigated the cooling effects from juvenile trees planted by a TPI in Massachusetts, and like all TPIs, this initiative requires a steady source of funding to be able to accomplish its goals (Eisenman et al., 2021; Young, 2011) which brings into question their long-term role in tree planting and increasing canopy cover. Trees may also die or get removed after planting, and if these lost trees are not replaced, cooling effects will not materialize (Ko et al. 2015). We found that dense tree plantings (at least 2-3 trees per acre) can exhibit cooling effects on the hottest day temperatures when a sufficient number of trees are planted, leading to benefits in the short-term (Figure 3-8). However, repeated cycles of planting and replacement are needed in dynamic urban landscapes (Nowak and Greenfield, 2018; Roman et al, 2018; Roman et al., 2022), especially in areas with unjust environmental legacies (Healy et al., 2022b: Drescher, 2019), to create long-term, mature tree canopy which requires long-term commitments from TPIs and other stakeholders. Increasing tree canopy through sustained tree planting is an effective tool to mitigate the UHI and can be combined with other mitigation strategies (e.g., white and green roofs, solar canopies, high albedo materials, increasing greenspace, see O'Malley et al., 2015; Wang et al., 2016) to amplify cooling effects. Overall, juvenile trees begin to impact the UHI in their local areas 4-7 after planting, especially when planted in dense configurations. The benefits of juvenile trees should be more readily considered by TPIs and urban forest managers when proposing tree planting projects and better shared with residents who can receive and benefit from the trees earlier than previously expected. Cities and TPIs may find a quicker return on investment than previously thought, which could help to justify expenditures for urban greening programs.

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6 Tables

Table 3-1. HOBO temperature sensor sites used for land surface temperature analysis (see Section 2.3.2), their land-cover attributes, and number of trees planted by the Greening the Gateway Cities tree planting initiative within a 100 m buffer. Summaries of MAE and RMSE of the four sensor sites, for both point-based and buffer-based approaches. All temperature anomalies were removed before analysis.

	Sensor ID	**Number of trees planted**	**Canopy cover (%)**	**Impervious cover (%)**	**MAE (Point-based)**	**MAE (Buffer-based)**	**RMSE (Point-based)**	**RMSE Buffer-based)**
Planting Zone	2212	36	11.5	69.2	11.50	12.05	11.97	12.42
	2484	11	9.4	70.0	13.14	13.14	13.61	13.58
	4588	23	1.8	83.8	11.92	11.91	12.80	12.81
	5110	31	14.8	61.6	6.56	7.20	7.16	7.70
	6568	37	16.0	67.5	10.47	10.83	10.91	11.17
	7576	25	17.5	46.8	11.81	11.01	12.12	11.39
	11865	13	31.0	36.6	7.16	7.16	7.36	7.37
	16525	14	19.6	50.6	8.39	8.99	8.63	9.20
	19226	35	8.8	70.1	9.88	9.99	10.09	10.16
	19429	47	6.6	73.6	13.49	13.90	14.08	14.46
	23734	25	7.3	75.0	9.66	9.45	9.98	9.77
	25383	4	39.0	32.3	5.49	4.92	6.04	5.28
	26919	7	16.3	49.0	9.74	9.20	9.89	9.54
	27802	14	9.6	75.4	8.96	8.82	9.45	9.29
	Average	**23**	**14.9**	**61.5**	**9.87**	**9.90**	**10.29**	**10.29**
Control Zone	10574538	NA	37.0	50.9	6.47	5.82	6.94	6.41
	10574539	NA	42.1	31.5	7.46	8.75	7.84	9.09
	10574553	NA	11.8	58.5	9.17	9.10	9.33	9.27
	10574582	NA	22.9	50.7	5.61	5.17	6.09	5.69
	10574583	NA	18.8	47.7	9.95	9.44	10.14	9.65
	10574584	NA	24.0	48.5	8.43	9.26	8.81	9.57
	10574587	NA	6.7	50.5	7.64	7.48	8.02	7.90
	10574588	NA	22.6	35.3	4.73	4.16	4.81	4.31
	10574607	NA	16.1	54.3	10.26	10.70	10.56	10.97
	10574618	NA	39.7	44.2	8.73	5.18	8.93	5.52
	10574624	NA	19.0	46.6	4.94	5.04	5.79	5.79
	10574625	NA	14.7	55.6	5.98	6.21	6.68	6.75
	10574638	NA	19.6	48.5	9.17	7.84	9.55	8.25
	10574639	NA	48.6	32.7	5.97	5.79	6.37	6.14
	10574643	NA	24.3	51.3	6.18	6.57	6.76	7.18
	10574645	NA	16.9	59.3	12.31	11.94	12.67	12.32
	10574647	NA	20.4	67.3	7.59	8.29	8.38	8.91
	10574677	NA	10.7	73.6	9.64	8.75	9.90	9.01
	Average	**NA**	**23.1**	**50.4**	**7.79**	**7.52**	**8.20**	**7.93**

Table 3-2. Landsat Analysis Ready Data Surface Temperature images from 13 June 2017–27 August 2021 covering Holyoke, MA.

Date	**Satellite/Sensor**
13 June 2017	Landsat 7 ETM+
23 July 2017	Landsat 8 OLI/TIRS
24 August 2017	Landsat 8 OLI/TIRS
16 June 2018	Landsat 7 ETM+
02 July 2018	Landsat 7 ETM+
10 July 2018	Landsat 8 OLI/TIRS
18 July 2018	Landsat 7 ETM+
27 June 2019	Landsat 8 OLI/TIRS
05 July 2019	Landsat 7 ETM+
21 July 2019	Landsat 7 ETM+
30 August 2019	Landsat 8 OLI/TIRS
21 June 2020	Landsat 7 ETM+
31 July 2020	Landsat 8 OLI/TIRS
08 August 2020	Landsat 7 ETM+
16 June 2021	Landsat 8 OLI/TIRS
24 June 2021	Landsat 7 ETM+
26 July 2021	Landsat 7 ETM+
27 August 2021	Landsat 7 ETM+

Table 3-3. Model metrics and coefficient values for variables in a linear regression with point-based LST (°C) as the independent variable. This was conducted across all 14 PZ temperature sensor sites for the hottest days of each year between 2017-2021.

	13 June 2017	**2 July 2018**	**21 July 2019**	**21 June 2020**	**27 August 2021**
Coefficient Values					
Number of trees planted	-0.042	-0.040	0.011	-0.031	-0.007
Percent Canopy Cover	-0.053	-0.015	-0.068	0.036	0.070
Percent Impervious Cover	0.064	0.102	0.026	0.137	0.029
Model Metrics					
Adjusted R^2	0.53	0.75	0.68	0.56	0.36
p-value	0.015	0.001	0.002	0.032	0.061

Table 3-4. Model metrics and coefficient values for variables in a linear regression with buffer-based LST (°C) as the independent variable. This was conducted across all 14 PZ temperature sensor sites for the hottest days of each year between 2017-2021.

	13 June 2017	**2 July 2018**	**21 July 2019**	**21 June 2020**	**27 August 2021**
Coefficient Values					
Number of trees planted	-0.010	-0.023	0.022	-0.011	0.007
Percent Canopy Cover	0.044	0.026	-0.106	0.010	-0.011
Percent Impervious Cover	0.145**	0.146*	0.009	0.120	0.074
Model Metrics					
Adjusted R^2	0.88	0.80	0.70	0.52	0.71
p-value	0.000	0.000	0.002	0.011	0.001

7 Figures

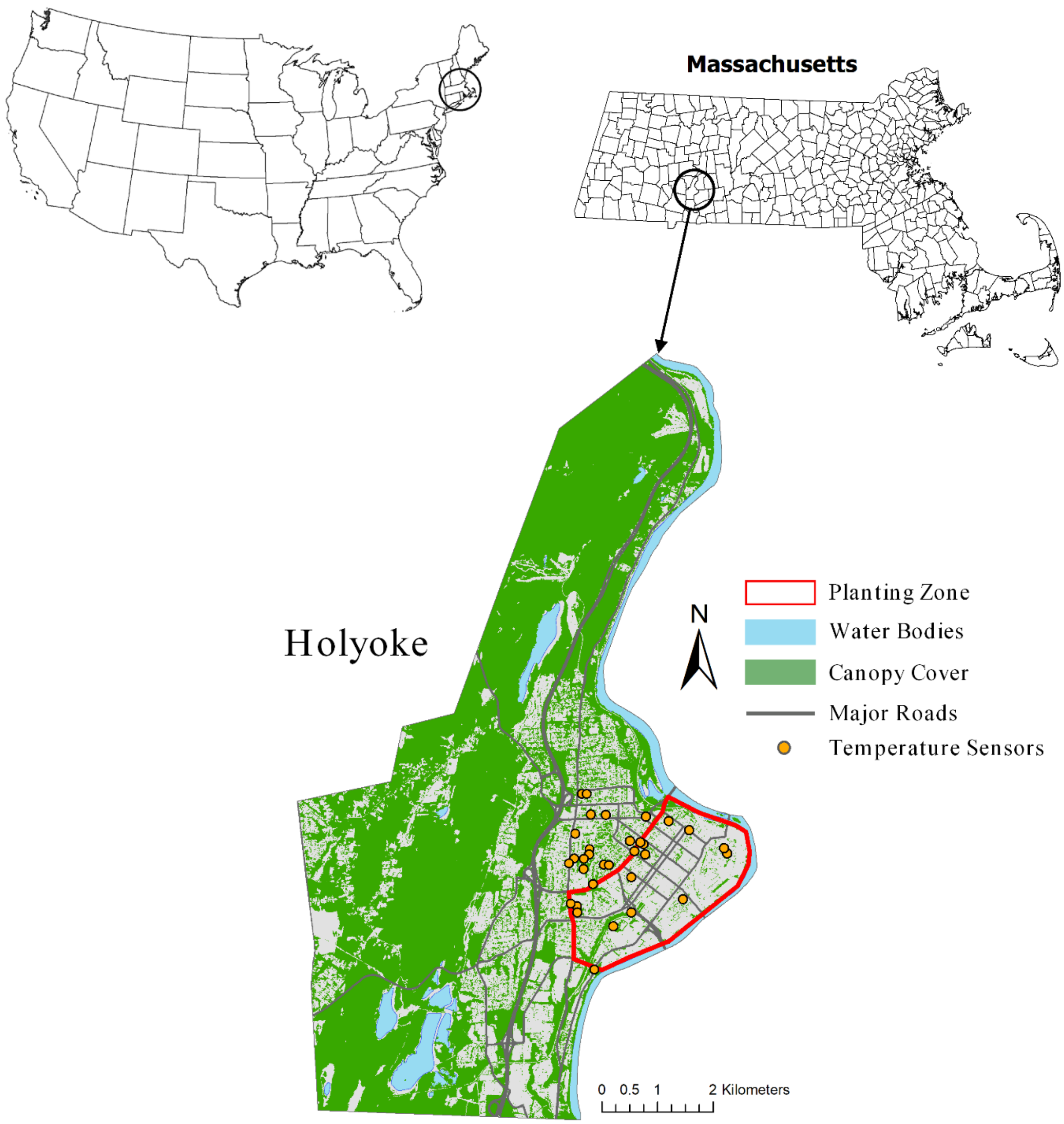

Figure 3-1. Holyoke, Massachusetts city boundaries displaying the planting zone, water bodies, 2016 urban tree canopy cover, major roads and DCR temperature sensor network.

Figure 3-2. Examples of tree planting in residential areas by the Greening the Gateway Cities tree planting initiative.

Figure 3-3. Map of the four sensor points and associated 100 m buffer used in the surface temperature analysis. Sensors 10574553 and 10574538 were in the control zone while Sensors 19226 and 11865 were in the planting zone. The map displays the land-cover classes and number of trees planted by the Greening the Gateway Cities tree planting initiative.

	Average Temperature	Maximum Temperature
Coefficient Values		
Number of trees planted	-0.012*	-0.043***
Years since planting	0.133	0.046
Percent Canopy Cover	0.072***	0.193***
Percent Impervious Cover	0.052***	0.140***
Model Metrics		
Adjusted R^2	0.003	0.01
p-value	1.22e-08	< 2.2e-16

	Average Temperature	Maximum Temperature
Coefficient Values		
Number of trees planted	-0.006	-0.031***
Model Metrics		
Adjusted R^2	6.98e-05	0.003
p-value	0.18	6.53e-09
Coefficient Values		
Years since planting	0.129***	0.026
Model Metrics		
Adjusted R^2	0.001	-5.01e-05
p-value	8.35e-05	0.53
Coefficient Values		
Percent Canopy Cover	0.002	0.015*
Model Metrics		
Adjusted R^2	-6.75e-05	0.0003
p-value	0.66	0.03
Coefficient Values		
Percent Impervious Cover	0.004	0.004
Model Metrics		
Adjusted R^2	1.91e-05	-6.67e-07
p-value	0.27	0.32

Figure 3-4. Pooled multivariate regression on the left and pooled linear regression on the right for each variable displaying model metrics and coefficient values. One asterisk represents significance at 0.1, two asterisks represent significance at 0.05, and three asterisks represent significance at 0.01.

	12 June 2017	1 July 2018	20 July 2019	19 July 2020	29 June 2021
Coefficient Values					
Number of trees planted	-0.045	-0.084*	-0.072**	-0.139*	-0.087**
Percent Canopy Cover	0.199	0.143	0.296**	0.453	0.369**
Percent Impervious Cover	0.125	0.117	0.232**	0.336	0.274**
Model Metrics					
Adjusted R^2	0.15	0.17	0.59	0.68	0.57
p-value	0.22	0.21	0.01	0.19	0.009
n	14	13	13	6	14

	12 June 2017	1 July 2018	20 July 2019	19 July 2020	29 June 2021
Coefficient Values					
Number of trees planted	-0.042	-0.069	-0.047	-0.104	-0.053
Model Metrics					
Adjusted R^2	0.06	0.17	0.04	0.33	0.04
p-value	0.21	0.09	0.24	0.14	0.23
Coefficient Values					
Percent Canopy Cover	0.044	0.028	0.0007	0.019	0.021
Model Metrics					
Adjusted R^2	0.01	-0.06	-0.09	-0.24	-0.07
p-value	0.32	0.60	0.99	0.84	0.73
Coefficient Values					
Percent Impervious Cover	-0.01	-0.005	0.023	-0.0008	0.018
Model Metrics					
Adjusted R^2	-0.07	-0.09	-0.04	-0.25	-0.06
p-value	0.66	0.88	0.46	0.99	0.63

Figure 3-5. Multivariate regression on the left and linear regression on the right for each variable displaying model metrics and coefficient values with air temperature (°C) as the independent variable. The n varies due to data availability by date. One asterisk represents significance at 0.1 and two asterisks represent significance at 0.05.

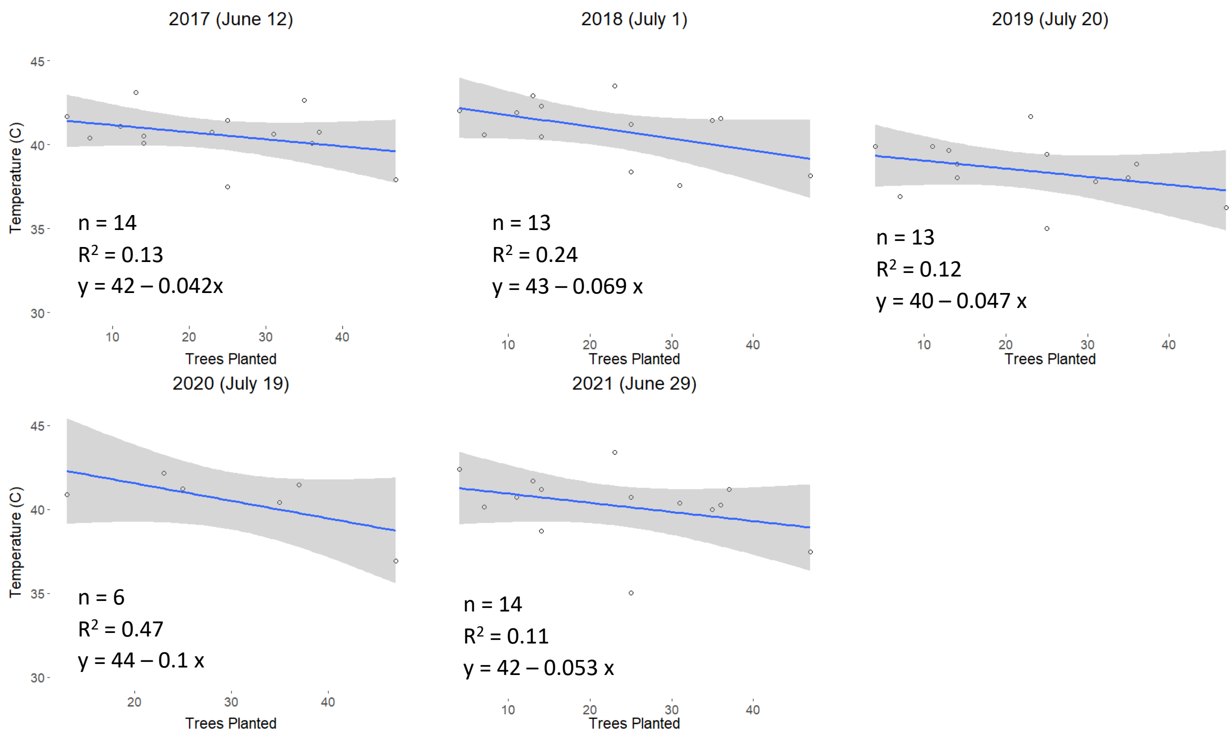

Figure 3-6. Comparison of planting zone maximum daily air temperature and the number of GGC trees within 100 m of the HOBO sensors on the hottest day of 2017-2021. The number of sensors used for each year's comparison is given by the n value. 2020 had a low sample size (n = 6) due to sensor malfunction.

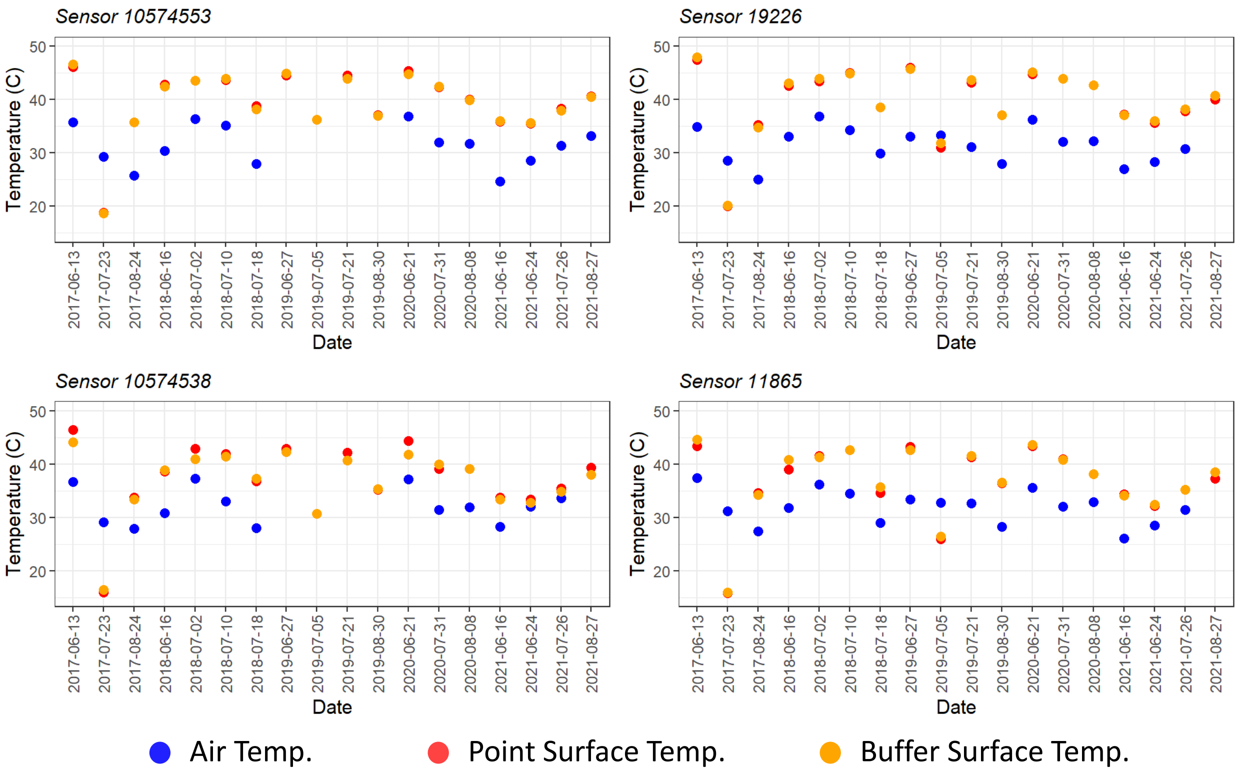

Figure 3-7. Air temperature, point-based surface temperature, and buffer-based surface temperature across four temperature sensor sites. Some air temperature measurements are missing due to data collection issues. Temperature anomalies occur on 23 July 2017 and 5 July 2019.

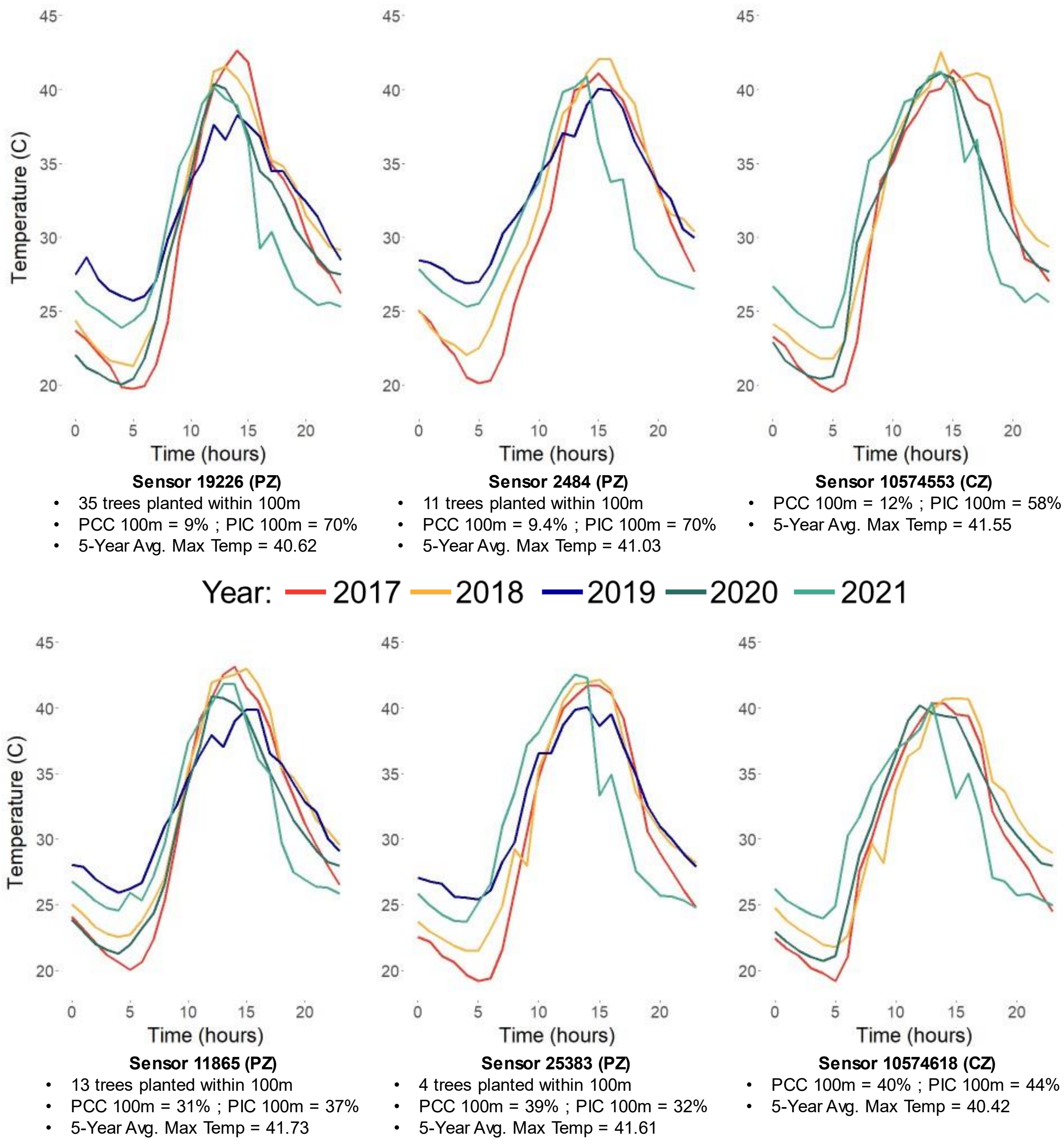

Figure 3-8. Diurnal charts of HOBO sensors on the hottest days of each year (2017-2021). The top row of charts has similarly low levels of tree canopy while the bottom row has high levels of tree canopy. Each sensor has various quantities of tree planting, except for the CZ sensors which have none.

Chapter 4: Urban forest management motivations and practices in relation to a large-scale tree planting initiative[3]

Abstract

For urban tree planting initiatives (TPIs) to deliver promised benefits, trees must be able to successfully establish and grow to maturity amid a complex web of local governance and stewardship actors: residents, staff at municipal departments and non-profit organizations, and other stakeholders. For trees in the public municipal realm (i.e., street trees, other right-of-way trees, and park trees), TPI success can depend on how well the new trees fit into existing municipal structures and capacities, including department structure, tree maintenance practices, staffing and budget. We sought to understand municipal management of trees through a case study in Massachusetts (US) involving a state-funded and state-managed TPI. Data was collected through structured interviews with tree wardens (municipal urban forest managers) to understand the various impacts that maintenance practices, municipal support and funding, and department structure may have on the recently planted trees. We concluded that the municipal departmental structure influences the number of proactive management practices as well as the size of the tree activity budget. Consideration of municipal roles and structure by TPIs may allow for more effective implementation of these initiatives.

[3] Healy, M., Geron, N., Harper R.W., Rogan, J., Martin, D.G., Roman, L.A. (In Review). Urban forest management motivations and practices in relation to a large-scale tree planting initiative. Submitted to *Society and Natural Resources*.

1 Introduction

Tree planting initiatives (TPIs) have become an important means for local urban forestry organizations to pursue goals of creating more sustainable and equitable urban environments (Eisenman et al., 2021). Some initiatives led by municipalities and non-profits have focused on tree giveaways for residential yards, with residents obtaining free trees and assuming responsibility for their planting and maintenance (Nguyen et al., 2017). Other initiatives employ professional arborists to oversee and train volunteer-led street and park tree planting (Roman et al., 2015; Fisher et al., 2015; Hauer et al., 2018). TPIs are often touted as a sound public investment that will generate a return-on-investment from the bevy of increasing environmental benefits that trees are expected to generate over time (Pincetl et al., 2013; McDonald et al., 2016; Breger et al., 2019). However, TPIs do not detail costs such as labor and maintenance (Koeser et al., 2016), nor the disservices that trees may produce (Conway and Yip, 2016; Roman et al., 2021). TPIs could leave municipalities or residents on the hook for these costs, and it is unknown if residents and/or municipalities have the individual or institutional capacity to manage future costs and disservices (Breger et al., 2019).

In order for TPIs to deliver on promised urban tree benefits, trees must be able to reach maturity amid a complex web of governance and stewardship that involves numerous actors, including residents; personnel working in municipal parks, forestry, or public works departments; staff at non-profit organizations; and other stakeholders. When ownership, stewardship and maintenance responsibilities are clearly defined, well-funded, and supported by arboricultural expertise, juvenile trees may be expected to have high rates of survival (Roman et al, 2015; Geron et al., in review). In cities in the United States (US), municipal agencies and staff are critical for providing stewardship and maintenance for newly planted trees within their boundaries – including street trees and other trees in the public right-of-way as well as trees in municipal parks – where these agencies have formal jurisdiction over tree management (Braverman, 2008; Eisenman, 2016; Roman et al., 2018). As trees age they require different kinds of maintenance, that if not routinely completed, may lead to nuisance complaints (e.g., messy fruit droppings), public safety risk (e.g., falling limbs or trees), and fostering negative opinions towards trees (Shatz et al, 2013; Conway and Yip, 2016; Carmichael and McDonough, 2018). Thus, the long-term success of TPIs remains challenged by existing frameworks of municipal government, tree management, and perhaps most importantly, fiscal budgets (Pincetl et al., 2013; Hauer and Peterson, 2016; Geron et al., 2022).

To aid in coordination of governance and maintenance of urban trees, Urban Forest Management Plans (UFMPs) are a policy tool for assessing the current state of an urban forest system, and making plans for planting, maintenance, and monitoring. UFMPs also enable municipalities to justify expenditures related to urban tree programs (Ordóñez and Duinker, 2013). Researchers assessing Scottish urban forest management found that policy frameworks, such as UFMPs, were important in promoting preventative tree care, creating consistency in communication, and aiding in the allocation of resources (van der Jagt and Lawrence, 2019). In the US, a study of Massachusetts tree wardens (equivalent to municipal arborists or foresters with unique legal responsibilities; see Harper et al., 2017) found ‘lackluster support’ (32%) for management plans as an important performance parameter (Rines et al., 2010, p. 299). This lack of support could be based on tree wardens’ past experiences with UFMPs being created but failing to be implemented, possibly due to budget constraints (Rines et al., 2010).

Municipal tree ordinances and by-laws are another widely recognized policy tool that outline standards for the protection and regulation of both municipally managed trees and trees on private property (Hill et al., 2010; Conway and Lue, 2018). In Florida, municipalities with heritage tree ordinances (i.e., protections for trees above a certain size or in certain taxonomic groups) have significantly higher canopy cover (Hilbert et al., 2019), and municipal officials across Pennsylvania recognized the importance of ordinances as sound urban tree management (Stevenson et al., 2008). In Atlanta, Georgia, urban tree land-use ordinances were associated with an increase in urban tree canopy over ten years (Hill et al., 2010). A municipality without an UFMP and/or tree ordinance may not have the ability to plan for tree care, and this lack of formal policy and planning may be compounded by lack of financial resources to maintain trees, which may unduly impact the survivorship of recently planted young trees (Roman et al., 2015; Widney et al., 2016; Breger et al., 2019).

Urban forest management may vary by municipality based on factors including community size (Grado et al., 2013; Treiman and Gartner, 2004), resource availability (Stobbart and Johnston, 2012; Miller and Bates, 1978) and residents' priorities (Treiman and Gartner, 2005). These factors individually or collectively impact municipal tree care practices and residential stewardship. Furthermore, urban forest managers are employed across many different municipal departments such as parks and recreation, public works, planning, transportation, and forestry (Hauer and Peterson, 2016). Larger municipalities (>50,000) have been found to have more forestry-focused departments (e.g., parks and recreation, forestry) than smaller communities (Hauer and Peterson, 2016). This mismatch in departmental focus may lead to different priorities across departments and may impact tree care practices and management (Breger et al. 2019). A systematic review of municipal urban forest management has highlighted a lack of planning and proactive management as one of the most common challenges to managers, second to a lack of budget (Ordóñez et al., 2019). It is unknown whether proactive management can be enabled by TPIs in the context of differing municipal departmental structures and capacities.

The success of TPIs, in terms of the establishment and long-term growth of the new trees, relies on municipal management of urban trees, but it is not well understood how TPIs intersect with existing municipal organizational structures and capacities. In this paper, we examine urban forest management motivations and practices of municipalities taking part in a state-funded TPI in Massachusetts. We characterize the variation of urban forestry practices (proactive and reactive) and resources used for tree maintenance to demonstrate how a given municipality's existing capacity may impact outcomes for public trees. We also analyze municipal department structure and support to understand how urban forestry operations are organized within each municipality and to ascertain if there are ramifications relative to municipal funding and management strategies. Our overarching goal is to explain how municipal objectives, motivations, and resources are used to care for and manage urban trees installed as part of a TPI. We then close with insights about municipal tree management in relation to TPIs.

2 Methods

2.1 Study area

This study focuses on a state-funded and state-managed TPI in Massachusetts titled Greening the Gateway Cities (GGC). A "Gateway City" is defined as a mid-sized, regional economic center that historically provided manufacturing jobs; today, these urban centers face a range of socio-economic challenges in the post-industrial economy (MassINC, 2015), including chronically low urban tree canopy cover. The Commonwealth of Massachusetts (2010) defines a Gateway City using three metrics: 1) population between 35,000 and 250,000; 2) median household income below the state average [US$70,954]; and 3) average educational attainment level (Bachelor’s degree or above) below the state average [41.3%] (US Census Bureau, 2016).

The GGC TPI is administered by the Massachusetts Department of Conservation and Recreation (DCR) and is similar to other TPIs across the US as it involves tree planting on a variety of urban landscape types (e.g., residential yards, sidewalks, and institutional grounds spanning both public and private lands). This differs from other TPIs in the US that focus on tree giveaways for residential yards (i.e., tree planting is the responsibility of residents, Nguyen et al., 2017) or that have arborists from non-profits or municipal government train volunteers to carry out tree planting (Roman et al., 2015; Fisher et al., 2015; Hauer et al., 2018). The Massachusetts DCR plants all GGC trees using professional arborists and seasonal staff with an expressed focus on planting trees for their affiliated ecosystem services in zones that have been designated by the state in need of environmental justice (Breger et al., 2019). The GGC uses an 80:20 planting ratio of private residential versus public sites, and works with landowners, municipal agencies and local non-profits for watering and maintenance (Breger et al., 2019). The GGC TPI aims to increase neighborhood urban tree canopy by 5-10 percentage points in order to reduce residential winter-heating and summer-cooling costs (Commonwealth of Massachusetts, 2015). The GGC initiative is also expected to provide co-benefits such as reduced stormwater runoff, improved air quality, increased property values and tax receipts, and a safer, healthier environment for residents (Commonwealth of Massachusetts, 2015). Between 2014 and 2021, this TPI has planted over 20,000 trees across an initial 14 cities and towns in Massachusetts, and since 2020 has expanded to include eight more cities. This study focuses on the original 14 cities and towns in the GGC TPI (Figure 4-1).

2.2 Analytical Framework and Interview Data Collection

We conducted qualitative research interviews with Massachusetts tree wardens and/or other related tree management personnel following similar methodological guidelines from Harper et al. (2020). Massachusetts tree wardens are a legally mandated position for every municipality in the Commonwealth but are typically not a full-time paid position (Rines et al., 2010). Tree wardens are commissioned to preserve and steward a municipality's public trees and have legal authority over their management (Steiner, 2016). As part of the interview process, we collected information from tree wardens and other related tree management personnel regarding their municipality's urban forestry organizational structure (referred to as departmental structure hereafter), tree management practices and goals, and municipal budget in relation to the municipality and the GGC initiative. Qualitative analytical methods were used to explore themes related to urban tree management and departmental structure across the GGC participating municipalities. Specifically, we aimed to draw conclusions about municipal capacity (including budget and personnel) for tree care and management amid the GGC TPI. The insights were then linked to data concerning budgets, management practice, tree planting and tree removal to illustrate the impacts of TPIs on municipalities.

In describing the departmental structure, we categorized municipalities as either having multi-hat (MH) or tree-focused (TF) urban forestry staff. MH refers to a situation in which a tree warden has different roles and responsibilities besides urban forestry [e.g., a Department of Public Works (DPW) tree warden might also be responsible for utility services], while a tree-focused (TF) municipality has a tree warden whose roles and responsibilities are solely concerned with urban forestry. Urban forestry practices and goals for each municipality were also tied to departmental structure as a way to understand current goals and abilities, but to also contrast them across both organizational structures (MH or TF). The conceptual framework (Figure 4-2) illustrates how departmental structure, tree management, and municipal capacity interact, and ultimately impact the public (municipal residents) which lead to feedback from the public to the municipality.

Tree management practices identified through interview responses were each labeled as either proactive (planned, preventative tree care practices that anticipate and preempt disservices from trees) or reactive (reactionary tree care practices that respond to existing threats or hazards, including responses of absent proactive practices) and encompass approaches for increasing tree canopy as well as tree care and maintenance. Absent proactive practices were included with reactive practices because other researchers have qualified the lack of action as reactive for tree-related emergencies (e.g., limb or tree failure due to neglect of regular inspections, pruning, or removal) (Hauer and Peterson, 2016; Ryder and Moore, 2013). While the term 'reactive practice' implies action, what is actually observed is typically negligible and insubstantial practices. The number of each type of practice was quantified to determine: 1) how reactive/proactive each municipality is; 2) the most commonly implemented reactive/proactive practices; and 3) if the number of reactive/proactive practices are influenced by their departmental structure.

To identify municipal capacity, specifically resource allocations, that coincided with the GGC, we also quantified the pace of tree planting and tree removal, as well as fiscal budgets. Budgetary information for each municipality was collected using the Tree Activity Budget format from Hauer and Peterson (2016) which includes all tree activity expenses such as personnel (salary), overhead, equipment, supplies, tree care and contract payments. Personnel (salary) costs were removed from the initial budget data to denote monetary variation for tree care activities only.

Interview questions (Appendix, 4-A) were structured using both close-ended and open-ended questions to find specific tree-related municipal characteristics and to allow tree wardens to share the depth and meaning of their management experience and interactions with the GGC. The interview questions were first trialed with tree wardens in cities that did not participate in the GGC and were refined based on feedback from the trial (Dampier et al, 2015; Harper et al., 2017). Tree wardens from the 14 original communities participating in the GGC initiative were contacted by phone and email, with the latter containing a cover letter explaining the research project and a consent form. Interviews typically took 30-60 minutes to complete. All interviews took place between April and June 2021. Interviews with the first author were conducted in an online, video format due to COVID-19 restrictions, and were recorded.

Interviews were transcribed and coded using NVivo (2020) qualitative analysis software by the first author. Interviews were first coded based on the section themes of the interview questions. Sub-themes were either predetermined by the interview questions or were identified through the coding process in discussion with the second author. Once the initial coding was complete, transcripts were reviewed again to confirm or edit the existing themes and sub-themes.

3 Results

Tree wardens and other related tree management personnel from seven of the 14 original GGC municipalities (50% response rate) agreed to be interviewed. Five interviews were with tree wardens and two interviews were based on referrals, with: 1) a forester working at a non-profit organization that contractually maintains city parks and greenspaces; 2) the Director of Planning and Development who spoke on behalf of the tree warden.

Of the seven municipalities in this study, only three had TF tree wardens whose full-time job was focused on tree management while the other four were MH and had to divide their time as a tree warden with other responsibilities (Table 4-1). The three TF tree wardens led their department or division and each reported that they were 100% focused on tree management in their municipality. The MH tree wardens typically had a higher administrative role (i.e., Director of Public Works) and managed multiple divisions, each with different municipal responsibilities (i.e., Parks, Sewers, Roads, Cemeteries), but with the added responsibilities of being tree warden. These MH interviewees reported that 10-40% of their time was spent on tree management.

Tree wardens from both MH and TF municipalities indicated that their urban forest management goals included wanting to create cyclical tree pruning programs, updating tree inventories, and planting more trees. MH tree wardens also prioritized hiring a city forester, creating a local tree ordinance, and stump removal while TF tree wardens prioritized creating urban forest management plans and increasing community education.

The number of reactive and proactive tree management practices was recorded and listed for each municipality in Table 4-1. Tables 4-2 and 4-3 include the list of practices in their entirety. The average number of proactive and reactive practices for MH municipalities was 5.5 and 6.0, respectively. For TF municipalities those average values were 10.3 and 3.7, indicating that TF municipalities had more proactive and fewer reactive tree management practices than MH municipalities (Table 4-5). Four of the municipalities featured a majority of proactive practices, while two had a majority of reactive practices, with one having an equal number of reactive and proactive practices.

The list of proactive and reactive practices (Tables 4-2 and 4-3) provides a snapshot of current urban tree management and the practices that municipalities are able to accomplish. The most common practices enacted across nearly all municipalities include local tree ordinances, community engagement (i.e., tree related activities for residents such as tree planting or pruning), and stump removal tied to planting or other process. The most common practices shared solely by municipalities with a majority of proactive practices were community education events (i.e., Earth/Arbor Day at public schools, tree care classes, tree hearings, community forums), watering for newly planted trees, planting more trees than removing per year, current or updated tree inventory, and proactively managing hazardous tree removal. Tree wardens from majority proactive practice municipalities expressed how important a tree inventory and management plan were relative to forming proactive practices, with one individual stating:

> *Before we did the management plan, the Forestry Department was staffed with four people. It's now staffed with eight…We also have a dedicated tree warden now, which our tree warden before 2014 was the Superintendent of the DPW who was wearing about four or five different hats. So, being the tree warden wasn't the top priority… The city has invested substantially in [our] infrastructure and equipment so that [we] can do more work in-house, which I think has worked in our favor.*

The most common practice shared solely by municipalities with a majority of reactive practices is reactive hazardous tree removal. One tree warden explained,

> *For the prior 20 years, the city just wasn't spending money on trees unless it was an emergency. That neglect led to a lot of stuff that had to be cleaned up…*

While most municipalities interviewed used the Right Tree Right Place philosophy (i.e., appropriate tree species are matched to suitable planting sites, see Ko et al., 2015 and Vogt et al., 2015) for tree planting, almost all of them mentioned having little to no funding dedicated primarily to tree planting. Outside of the GGC TPI, tree planting typically occurred as part of road and sidewalk infrastructure projects, federal, state, or local grants, or through volunteer initiatives. The yearly average number of tree plantings for each municipality for three separate time periods: before the GGC, during the GGC, and after the GGC, is shown in Table 4-4. Municipality 1 was the only municipality to see a decline in tree planting while the GGC TPI was active, and it dropped to zero. The tree warden mentioned the reason for the drop was that since the GGC was already engaged in tree planting, the municipal staff would be focusing their efforts elsewhere. This respondent also indicated, however that after the GGC TPI concluded, the municipality would likely resume tree planting efforts, depending on funding:

> *It depends on funding right now and the funding's not much… I'd like to do more, but we don't have the money.*

Municipality 2 noted an increase in tree planting after the GGC because of city government restructuring which led to the hiring of a tree warden and increased funds for tree planting, removal, and maintenance. While the municipality increased the number of tree plantings, it also increased the number of tree removals as it endeavored to clear a backlog of hazardous trees. Tree planting in Municipality 3 was managed by a non-profit, who also happens to run the GGC TPI in their municipality. Before the GGC TPI, the non-profit planted about 125 trees per year, while the municipality planted zero. However, due to their participation in the GGC initiative, the non-profit planted 700 trees a year. The non-profit plans to continue a substantially higher rate of tree planting after the GGC has wrapped up, as long as they can obtain the necessary funding (e.g., from local, state and federal grants). Municipalities 4 and 5 had a steady rate of tree planting across all three periods. Comments from the tree warden at Municipality 5 suggested there was higher variability from year to year, depending on capital projects, grant funding—and more recently, restrictions due to COVID-19—but that tree planting probably averaged around 200 installations per year. At the time of the interview, the tree warden at Municipality 6 did not have tree planting information on hand, but commented that they heavily rely upon a local non-profit for tree planting in the city. Municipality 7 saw a large increase in average tree planting for the periods during and after the GGC. This increase was due to a completed management plan which coincidentally started at the same time as the GGC TPI and led to a steady stream of tree planting, with plans continuing into the future.

Removal rates for the municipalities varied, but this was dependent on several factors like hazardous tree backlogs, personnel, and funding. Tree removals in relation to Municipality 4 seemed abnormally high but there was nothing emergent in the interview data that explained this occurrence.

Some tree wardens mentioned how they are more attentive to the newly planted trees in their municipality: they see the trees frequently and take note of their condition, and residents ask them questions about the trees. As for long term maintenance, nearly all tree wardens responded that they anticipate being able to maintain the GGC trees that were planted on public property after the program ends. Two tree wardens mentioned that they may request future funding or engage a local volunteer group for maintenance. The tree warden from Municipality 3 (where the non-profit works to maintain city parks and trees) initially expressed some reservations, but ultimately concluded that maintenance was possible:

> *In my organization I think we do have some capacity because we do park cleanups and park improvement projects… The city does not have the capacity or the wherewithal. They know it's important and they agree, but they just don't have the resources. They have to choose: can I water a tree or can I fill a pothole?*

The tree warden from Municipality 5 was the only one who expressed serious doubts about being able to maintain public GGC trees. As with Municipality 3, the tree warden indicated that difficult decisions would need to be made by the municipality regarding funding priorities, making the externally-funded GGC TPI critical:

> *If the Greening the Gateway Communities program leaves, we're going to be in dire straits. It's going to be hard to sustain it… I feel like we've gained a lot of momentum and if they leave, that momentum will come to a grinding halt.*

Tree activity budgets (excluding salary for personnel) for each municipality are shown in Table 4-1. According to the interview, the budget for Municipality 3 was unique because this level of funding was approved only for three years (2019-2021) and was paid to a contractor for the pruning and removal of hazardous trees. Previously, the tree activity budget was $0, and will revert back to $0 for fiscal year 2022—the end of the temporary funding period. Average tree activity budgets were compared across MH and TF municipalities (Table 4-5) with the budget of

Municipality 3 being removed from the average calculation due to its temporary status. There is a drastic difference between tree activity budgets of MH and TF municipalities with TF municipalities having a tree activity budget 3.4 times larger than MH municipalities.

When tree wardens were asked about what they would do with a budget increase, the most common response was to plant more trees (5 out of 7), regardless of being a MH or TF municipality. The next most common response (4 out of 7) was to increase the capacity of their in-house crews by hiring more arborists and tree care professionals—with all 3 TF municipalities responding and 1 MH municipality. One tree warden explained why hiring in-house tree crews was important:

> *It's too expensive to have outside vendors trimming, so that's why we've moved to more in-house crews and over the years we've sacrificed contractual money from our contractual line to be able to hire more in-house personnel and that has proved hugely beneficial to us.*

Only MH municipalities indicated that they wanted funds used for hazardous tree removal, while both MH and TF municipalities wanted funds to be used for pruning.

Finally, all tree wardens responded positively when asked if city officials view urban forestry as a priority and if they support the role of tree warden. Some expressed this sentiment by explaining that the city has increased their budget or staffing levels, while others expressed that since their budget had not been cut, this was a sign of support. One tree warden talked about the need for “proving” the departments “worth” because of recent restructuring that made the respondents’ position possible:

I feel that because I'm here, because this position is here, that's step one for the amount of support we're getting. Now what we need to do is show him [mayor] and the City Council what we can bring.

One tree warden commended local government officials for their awareness of the importance of trees and maintenance, but indicated concern that it may get lost among other priorities:

We love trees, we're very supportive of trees. But again, if you got the pie and you keep carving out of the pie for trees... And it's not just trees. There's a slew of important things that we gotta prioritize.

When asked about community support for urban forestry from residents, all tree wardens expressed that there is support but that it varied by local context. One tree warden shared how it took several years for residents to overcome a negative perception of trees after a severe ice storm damaged many trees and buildings. Another tree warden expressed how local volunteering for tree stewardship and advocacy has become a higher priority for residents in the municipality, but that conflict ensued when the volunteers tried to formalize their group as a non-profit. Other municipalities with established tree-focused advocacy groups or non-profits expressed how their support is critical for helping to maintain and plant new trees. One tree warden said that issues around environmental justice, the urban heat island, and air quality are what connect residents to trees—that they act as an intersection to discussing these issues. Tree wardens in this study did express concerns about climate change specifically around the urban heat island, effects on certain species and street trees, increased pests, and changing weather patterns.

4 Discussion

We found variation in how municipal personnel interact with and perceive a statewide TPI. Specifically, we observed large differences in urban forest management between MH and TF municipalities. Furthermore, the TPI has inspired forms of proactive management among all the municipalities and tree wardens who participated in our study. Our analysis also illustrates how the burden of public trees planted by the TPI is perceived to be minimal by a majority of the municipalities we interviewed. This may be due to inherent factors associated with the GGC TPI: it is externally funded and managed by the Massachusetts DCR; it plants trees with professional foresters and seasonal staff and maintains them for up to three years, with help from community partners. While some municipalities choose to assist and work in cooperation with the GGC, this arrangement is not mandated, and so the municipalities do not bear the initial financial and staffing burdens associated with the new installations. The maintenance burden of the trees is perceived to be low because, in the view of the tree wardens, most of the required maintenance will take place in the future, therefore relinquishing responsibility in the present. This perspective is concerning because of the possibility that GGC trees will be insufficiently cared for once the state foresters' labor and maintenance commitment comes to an end, which can result in higher costs in relation to future maintenance (Vogt et al., 2015), and may contribute to residents' negative experiences and perceptions of trees (Carmichael and McDonough, 2018, Roman et al. 2021). It is especially concerning for MH tree wardens to have a perceived low-maintenance burden of the GGC because they typically have smaller urban tree-related budgets and manage forests on more of a reactive basis than their TF counterparts.

Concerns surrounding municipal funding for tree planting and maintenance in the US has been a consistent theme in urban forestry literature (Pincetl, 2010; Danford et al., 2014; Hauer and Peterson, 2016; Breger et al., 2019), and were also evident in this research as budget issues impacted tree wardens' desires for more tree planting. Five of the seven tree wardens in this study would plant more trees if they had the funding, a sentiment shared by other staff from municipalities and TPIs (Young, 2011; Eisenman et al., 2021). Taking into consideration the financial context for each municipality is extremely important, especially for those with limited tree activity budgets. In a few municipalities, tree planting and maintenance is competing with built infrastructure needs, especially if the tree warden is in a MH role. The lack of municipal funding for tree planting and maintenance may foster a value and appreciation for the state-funded GGC TPI in each municipality, but many other TPIs in the US have sourced their funds from a diverse array of sources (e.g., citizens, corporation, non-profits, municipal, state and federal agencies) which may affect their long-term feasibility (Eisenman et al., 2021; Young, 2011). The size of the municipality has also been found to effect the funds for urban tree management due to the size of the tax base (Miller & Bates, 1978), resident knowledge of urban tree benefits (Grado et al., 2013), and the demand for tree-related services (Ries et al., 2007) which may explain some of the context surrounding the municipalities in this study as they are all small- to mid-sized municipalities (<160,000 residents). Tree wardens also raised concerns about costs and quality of tree maintenance when comparing in-house staff to contracted labor. They believed that in-house staff was cheaper and better overall for the quality of tree care in their municipality. Our research shows that the motivations of a municipality to hire more in-house tree maintenance staff is related to factors like departmental structure, municipal capacity, and level of proactive practices; more research is needed, however, to understand this relationship and its impacts on a more detailed level.

Tree wardens felt supported by departmental or tree activity-related budget increases, but they also felt supported by not having their budget cut when other municipal departments faced cuts. The tree wardens we interviewed portrayed their budgets as being constantly stretched, but on average, TF tree wardens have much larger tree activity budgets than their MH counterparts. This may have governance implications for MH municipalities as there is potential for them to increase tree activity budgets if they are able to restructure their departments to be TF instead of MH.

Management practices can be greatly influenced by the structure of the municipal department, as shown with TF municipalities having double the proactive practices as MH municipalities. Rines et al. (2010) surveyed 143 Massachusetts' tree wardens in 2006 about urban tree management priorities and found nearly unanimous support for hazard tree removal as a priority for urban tree management policy, while just below half of tree wardens supported preventative tree maintenance and tree planting as priorities for management policy. This difference was due to part-time and volunteer tree wardens not prioritizing preventative tree maintenance and tree planting relative to full-time tree wardens. Therefore, our findings corroborate Rines et al. (2010): TF tree wardens employ more proactive practices than MH counterparts. Urban forest managers in Scotland were found to mostly employ reactive practices and this was influenced by limited funding, resources, knowledge, poor management structures, and their perception of trees as a liability (van der Jagt and Lawrence, 2019). There is a research gap concerning urban forest management regimes and their impacts on costs as well as tree-related outcomes and benefits (Vogt et al. 2015). Our study begins to address this gap by identifying that TF municipalities also had much higher budgets than MH counterparts. However, the top practices that TF tree wardens engaged with, such as creating/enforcing local tree ordinances, community engagement activities, and maintaining a tree inventory, do not require expensive equipment or the purchasing of new trees. Rather, it would appear that a TF tree warden has the time to prepare and execute a plan that engages the municipality, residents, and the existing trees. Consequently, proactive tree management benefits from both having a TF tree warden and giving that tree warden sufficient program budget.

Urban forestry governance literature demonstrates that proactive management can lead to increased tree canopy cover but through the typical channels of a large budget and proactive tree planting practices (Roman et al., 2017; Roman et al., 2022); what often goes undiscussed or disclosed, however, are the detailed management processes needed for success (Ordóñez et al., 2019). Detailed management processes not only occur at the municipal level, but that they also include additional actors who liaise with both the urban forest and the municipality (i.e., residents, NGOs, TPIs, etc.), which produces a co-governance model (Ordóñez et al., 2019; Geron et al., in review). TPIs can facilitate, or be part of an existing co-governance model, but the role of the municipality may change depending on its structure (MH vs TF) and the involvement of other stakeholders and groups that participate in urban tree stewardship and planting. For instance, the tree wardens in our study acknowledged how important outside groups and resident tree stewardship were in accomplishing their work. Further research is needed to better understand the complexities of TPIs in small to medium cities by examining a co-governance model that includes staff at municipal and state levels, as well as nonprofit tree groups and consulting arboriculture professionals. Another detailed management process that is needed for success are local tree ordinances (Hill et al., 2010; Conway and Lue, 2018; Hilbert et al., 2019). Our research labels local tree ordinances as a proactive practice, but it does not assess the local ordinances themselves. The efficacy of local tree ordinances is highly dependent on how well they describe their requirements or protections, as well as the enforcement mechanisms. More research is needed to understand the varying levels of implementation and enforcement of local tree ordinances, as well as their sociopolitical origins (e.g., whether proactive practices codified through ordinances arose out of challenges from reactive management).

The lack of proactive urban tree management, particularly in conducting routine maintenance, can also impact municipalities. Deferred tree maintenance can result in increased costs for future urban forest managers and municipal residents (Vogt et al., 2015). In our study, deferred maintenance typically revolved around tree pruning and hazardous tree removal. Without regular maintenance, urban trees may pose an excessive risk to both people and property, leading to high removal costs and ramifications for the municipal budget (Ryan 1985). This has played out in two municipalities in our study that have endeavored to clear large backlogs of hazardous trees after years of deferred maintenance. The short-term budget for Municipality 3, above the average tree activity budget, was used entirely for contractual hazardous tree removal, and was renewed for two additional years to complete the work. After years of neglect, Municipality 2 found that an increased tree activity budget led to both increases in tree removals and tree planting. These examples demonstrate that inconsistent maintenance regimes can affect the municipal budget, but that they are also subject to municipal administration priorities, which may change due to new administrations or disturbance events (Konijnendijk et al., 2021; Harper et al., 2018). Climate change may further exacerbate poor maintenance practices and increase municipal costs as urban trees may have increased stressors and become more vulnerable to pests and disease (Tubby & Webber, 2010; Khan & Conway, 2020). More research is needed to fully understand municipal urban forestry viewpoints and policies related to climate change.

Our study focused on small to mid-size municipalities in one US state which may represent different dynamics than may be uncovered in large municipalities, in other regions of the US, or in other countries, and therefore more research is needed. However, many TPI studies focus on larger cities (e.g., Young, 2011; Pincetl et al., 2013, Campbell, 2014; Locke and Grove, 2014), as they account for most tree planting, but in the context of the northeast US, smaller municipalities collectively comprise the larger population (Doroski et al., 2020). Thus, the need for our study, and others like it, to begin filling this research gap regarding small to mid-size municipalities and their role in tree planting and maintenance. More research is needed, however, to understand the deficit of tree planting between larger and smaller municipalities and its application to other regions nationwide, and internationally.

Our research suggests that the management of urban forests is influenced by departmental structure and municipal capacity, particularly through the role of the urban forest manager (such as tree wardens in Massachusetts). This has implications for the success of TPIs as departmental structure and municipal capacity may affect the long-term survival and health of publicly planted trees. A deeper understanding of urban forest management at the municipal level is needed so that more proactive practices may be employed in the service of municipal trees.

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6 Tables

Table 4-1. Municipal positions for tree wardens in this study. Multi-hat (MH) and tree-focused (TF) signify if urban forestry is the focus of the tree warden. The number of proactive and reactive practices were derived from the interviews about the practices (see Tables 4-2 and 4-3). Tree activity budgets, excluding salaries, are shown for each municipality.

Municipality	Position Title	Department	MH or TF	Number of Proactive Practices	Number of Reactive Practices	Tree Activity Budget
1	Assistant Director of Public Works	Department of Public Works	MH	2	8	$50,000
2	Tree Warden	Department of Natural Resources	TF	11	4	$360,000
3[1]	Parks Department Supervisor	Department of Public Works	MH	10	5	$500,000[2]
4	City Forester	Parks, Buildings and Recreation Management Department	TF	8	5	$90,000
5	Commissioner of Public Works	Department of Public Works	MH	4	4	$50,000
6	Cemetery and Tree Division Manager	Department of Community Maintenance	MH	6	7	$65,000
7[1]	Tree Warden	Department of Public Works	TF	12	2	$120,000

1. Interviews conducted with associated representatives (i.e., not the tree warden).
2. Temporary funding increase that was paid to a contractor to prune and remove hazardous trees.

Table 4-2. Proactive urban forestry practices identified from the interviews and the number of municipalities that mentioned a given practice.

Proactive Practices	Number of municipalities
Local tree ordinances	6
Community engagement	6
Community education	5
Information shared with residents	5
Right Tree Right Place philosophy	5
Watering for newly planted trees	4
Resident/volunteer stewardship	4
Proactive hazardous tree removal	4
Current management plan (< 10 years old)	4
Current or updated tree inventory	4
Planting more trees than removing per year	3
Building resident trust	2
Stumps removal tied to cutting process	1
Pruning cycles - building up or currently have	1

Table 4-3. Reactive urban forestry practices identified from the interviews, including responses of absent proactive practices, and the number of municipalities that mentioned a given practice.

Reactive Practices	Number of municipalities
Stump removal tied to planting or other process	6
Little or no dedicated tree planting funding	6
Reactive pruning	5
Reliance on outside contractors	4
Old or no tree management plan (> 10 years old)	3
Old or no tree inventory	3
Reactive hazardous tree removal	2
No community education	1
No information shared with residents	1
No community engagement	1
No local tree ordinances	1
Removing more trees than planting per year	1

Table 4-4. Average annual number of trees planted (TP) based on three phases: 1) before GGC; 2) during GGC, and 3) after GGC; and yearly average of tree removal.

Municipality	Yearly Avg. TP before GGC	Yearly Avg. TP during GGC	Yearly Avg. TP after GGC	Yearly Avg. Tree Removal
1	60	0	60	70
2	150	150	350	250
3	125	700	450	200
4	160	160	160	600
5	200	200	200	NA
6	NA	NA	NA	60
7	15	125	125	75

Table 4-5. Average number of proactive and reactive practices as well as average tree activity budget compared between multi-hat (MH) and tree-focused (TF) municipalities.

	Average Proactive Practices	Average Reactive Practices	Average Tree Activity Budget
MH Municipality	5.5	6.0	$55,000[1]
TF Municipality	10.3	3.7	$190,000

1. The tree activity budget for Municipality 3 was removed from this calculation because the funding increase was temporary.

7 Figures

Figure 4-1. Fourteen original Greening the Gateway Cities in Massachusetts, US.

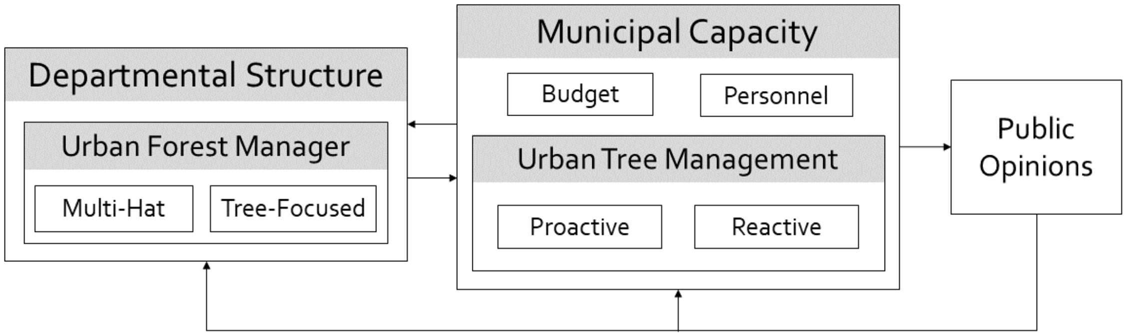

Figure 4-2. Conceptual framework demonstrating how a municipality's departmental structure and role (multi-hat or tree-focused) of the urban forest manager may influence urban tree management. Both the urban forest manager and municipal capacity impact and constrain the approach of tree management. Municipal capacity may also influence the departmental structure; hence there is a two-way relationship. Public opinions (including trust in municipal tree management) are shaped by perceptions of how successfully a municipality may (proactively or reactively) manage trees. This process can recursively feedback to influence municipal capacity and possibly the departmental structure. This simplified figure does not include factors that, while important, are not covered by this research, such as local governance history, city size, and resident attitudes, values, beliefs, or preferences.

Appendix

4-A. Tree warden interview questions.

I would like to start by clarifying that when I am saying "you" or "yours" I am asking for official municipal viewpoints/policies.

Position & Structure

1. Tell me about your position, and what department is it located in?
2. What does a typical day doing your job look like?
3. What percentage of your job is tree management? What other responsibilities do you have?

Management

4. Does your community have an urban forest inventory?
5. Does your community have an urban forest management plan?
6. Has the Greening the Gateway Cities program impacted these or their development?
7. What are your current tree management practices? Follow ups to help:
    a. Care Cycles – Watering, Pruning (Mature vs Juvenile), Mulching
    b. Canopy Cover Goals
    c. Species Diversity
    d. Pest and Disease
8. How has the Greening the Gateway Cities program changed your management practices, if at all?
9. Do you have any future goals for tree management in your municipality?
10. Tell me about educating the public in your community about trees.
11. Do you share information with residents (e.g. pamphlet, brochure, webpage)?
12. Are there any activities the public can engage with (e.g. community meetings, ceremonial tree planting, arbor day activities)?

Tree Planting & Removal

13. How many trees did the city plant yearly, on average, prior the GGCP?
14. How many did the city plant yearly, on average, during the program?
15. How many does the city plant on average now, or after the program?
16. How many trees are removed yearly, on average?
17. Are the stumps removed and does it impact planting efforts?

Support & Funding

18. How supportive is the community of urban forestry?
19. Do you partner with any outside groups? Follow ups to help:
    a. Tree Council/Commission
    b. Utility
    c. Community Organizations
    d. State Foresters
    e. Residents

Is urban forestry a priority for: (If so, how?)

f. the Mayor
g. the City Council
h. Your Department
i. Residents

20. In your role as [Tree Warden] do you feel supported by:
    a. the Mayor
    b. the City Council
    c. Your Department
21. What is the annual budget for tree care activities and management?
22. Do you have the resources to maintain the GGCP trees after the program wraps-up?

Improvements & Future

23. What are your municipality's biggest accomplishments for tree management?
24. What are your municipality's biggest challenges?
25. If you had more funding what would you do with it?
26. How did you envision the future state of urban canopy cover within your city before the GGC program? How about now?

Benefit & Costs

27. What are some examples of tree benefits in your municipality?
28. What are some examples of tree disservices in your municipality?

Chapter 5: Dissertation Conclusion

The research in this dissertation situates itself in the broader context of urban forestry and analyzes different parts of the framework proposed by Roman et al. (2018), which explains how the drivers of human and biophysical legacy effects co-produce the urban forests we observe in cities today (Figure 5-1). Together, the three articles of this dissertation demonstrate how the production of urban forests have come about in Massachusetts Greening the Gateway Cities (GGCs) program through human and biophysical legacies and the interaction of tree professionals. For example, Holyoke's post-industrial history—along with its history with pests and disease outbreaks—have produced decades of low canopy cover in its urban-industrialized core. This history also involves the role of the tree warden, whose job is to oversee the public urban forest, but they may have a split position that divides their responsibilities and, in turn, how the city maintains and manages urban trees. The human and biophysical legacies combine with the influence of tree care professionals to reveal the spatiotemporal dynamics of Holyoke's current urban forest, as well as demonstrate why Holyoke was selected to be part of the GGC tree planting initiative (TPI) and urban heat island research. Participating in the GGC represents a feedback loop that effects the human legacies and further feeds into the urban forest spatiotemporal dynamics, hopefully expanding the benefits for all in Holyoke who live in and among the shade of urban forests.

As the role of TPIs continue to grow as part of broader urban greening programs, especially in the face of climate change and the effects of the urban heat island, they can help provide greater environmental and human benefits in cities. The benefits TPIs hope to provide largely depends on the trees being able to establish and reach maturity amidst a complex web of co-governance that can be made up of many stakeholders, and in harsh growing environments—

as illustrated by the municipalities analyzed as part of this dissertation. Massachusetts Greening the Gateway Cities (GGC) face a range of social and environmental challenges due to the effects of their post-industrial legacies. These legacies have influenced the spatiotemporal dynamics of urban tree canopy (UTC) cover across decades, determining who has access to this resource and who does not, which was highlighted in Chapter 2. This chapter also demonstrates the use of a new UTC metric, maximum saturation, which displays the total extent of UTC and represents an approximate UTC maximum. Maximum saturation can be used to provide TPIs and cities with better information for overcoming legacy impacts, while helping to create more realistic, equitable, and targeted tree planting goals. Recently the city of Holyoke incorporated my findings from Chapter 2 into their Urban Forest Equity Plan (City of Holyoke, 2021), which aims to ensure that all of Holyoke's neighborhoods and residents have access to a healthy urban forest and the many benefits trees provide. The research work from Chapter 2 added historical depth to Holyoke's report by showing how underserved, poor neighborhoods have had consistently low canopy cover, underscoring the need for the equity plan. The Holyoke Urban Forest Equity Plan (City of Holyoke, 2021) is one example of how understanding historical trends of tree canopy in a city can influence future tree planting.

The post-industrial legacies of GGCs also influence municipal tree management regimes as they directly relate to fiscal policies and priorities. GGCs have lower property values and household incomes than the state average, which impact tax revenues for municipal programs and services. From the interviews conducted in Chapter 4, some tree wardens mentioned the difficult choices their municipalities have to make—the decision, for instance, to provide essential services or plant a tree. Yet, amidst these challenges, this dissertation has been able to demonstrate measurable cooling effects of the GGCs juvenile trees in residential neighborhoods (up to -0.087°C per tree). It is still in the early stages of the GGC trees' growth and development (i.e., 4-7 years since planting) which is promising for the future environmental and human benefits they will be able to provide, but more is still required for the GGCs to reach an equitable parity of tree canopy. A sustained effort to increase urban tree canopy is needed to offset ongoing losses and overcome historical legacies that have suppressed urban tree canopy across decades. Many of the GGCs are not currently able to provide a sustained effort for tree planting themselves and rely on the Commonwealth of Massachusetts for the uniquely funded GGC program, but this dissertation—particularly its findings surrounding the early-stage cooling capacities of juvenile trees—may influence how urban forest managers or TPIs advocate for trees. Cities and TPIs may find a quicker return on investment than previously thought, which could help to justify expenditures for urban greening programs.

Tree wardens play a unique role in Massachusetts as a legally mandated position, and this dissertation found that municipalities with a dedicated tree warden position employed more proactive tree care practices and had higher tree activity budgets than their non-dedicated position counterparts. This finding from Chapter 4 can help put into focus the need for dedicated tree warden positions in Massachusetts, especially for GGC participating municipalities, and for dedicated urban forest managers in cities globally. With the future of the GGC possibly being a one-off program, the role of municipal tree wardens become very important as they have legal authority over public trees and can shape the maintenance and management of trees in their municipality. This dissertation also suggests that dedicated positions for urban forest managers could greatly enhance urban tree canopy maintenance and management and across the country, and internationally.

This dissertation focuses on the cycle of human and biophysical legacy effects and how they co-produce the urban forests we observe in Massachusetts post-industrial cities. The overall findings may be indicative of broader changes that have taken place in other post-industrial cities located within forested biomes, but the methods used can be applied in cities with different socioeconomic histories and environmental conditions. More research is needed to understand human and biophysical legacy effects across cities with differing socioeconomic histories, environmental contexts. This dissertation also begins to fill a research gap in urban forestry literature regarding small to mid-size cities, which currently have a dearth of research concerning them, even though these areas can collectively comprise the larger population. While the methods used in this dissertation can be used by cities of any size, more research is needed to understand the different dynamics and applications of urban forestry between small/mid-size cities and large cities in different regions and countries.

In conclusion, this dissertation offers a more complete examination of themes that have had little prior investigation: the human and biophysical legacy effects on urban forests through the lens of a tree planting initiative, Greening the Gateway Cities. The most important contributions of this work are these: a more holistic understanding of urban tree canopy cover dynamics in Massachusetts' post-industrial cities using novel canopy metrics—metrics which can be used across cities regardless of their size, socioeconomic history or environmental condition; the quantification of local-scale cooling effects of recently planted trees, which had not been previously well understood or measured; and the exposition of dedicated urban forest manager positions and their influence on municipal tree management and budgets.

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Figures

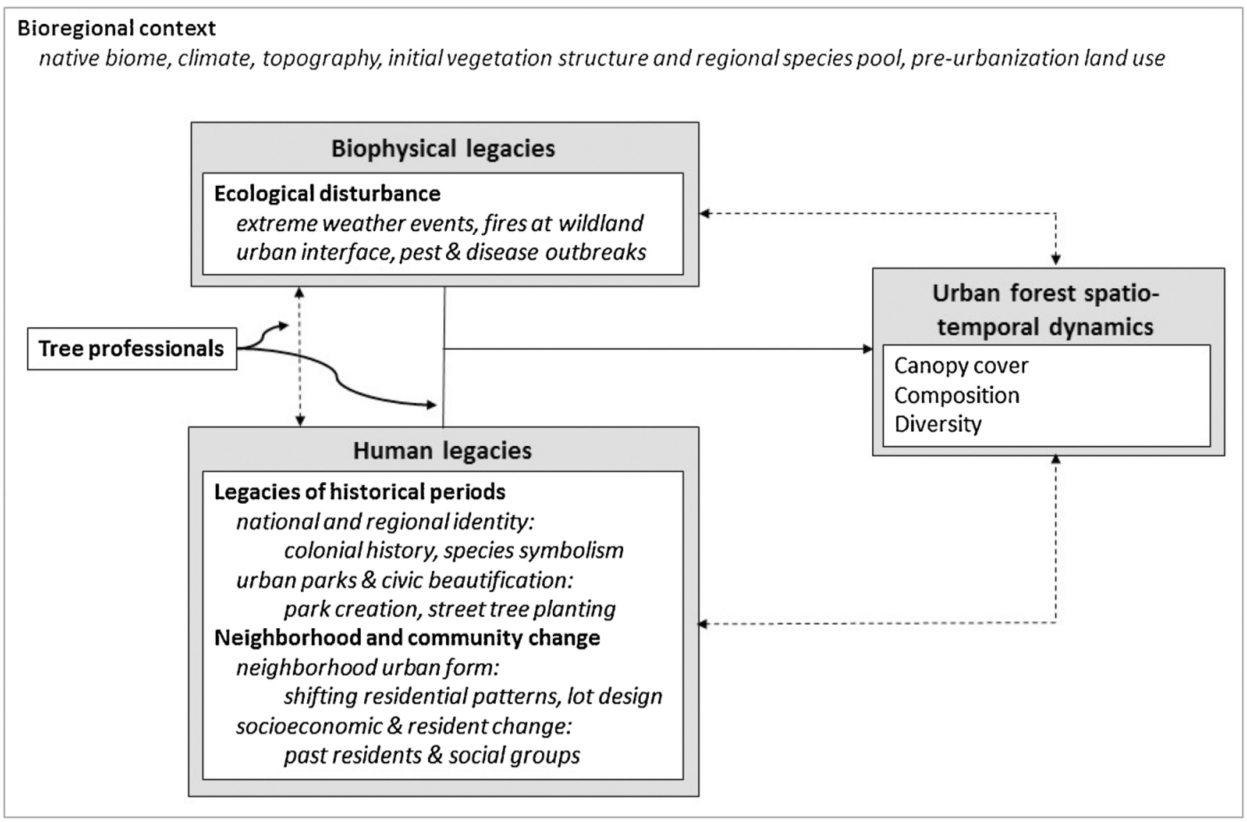

Figure 5-1. Framework of the drivers of human and biophysical legacy effects in urban forests from Roman et al. (2018). The bioregional context provides the background conditions from which a city's urban forest grows. The drivers (human and biophysical legacies), with interactions from tree professionals, co-produce the spatiotemporal dynamics of the urban forest, which has feedback loops as the legacies and dynamics interact.