The University of Chicago

Renters, Buildings and Scale:

A Spatial Analysis of Urban Tree Cover in Chicago

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By

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Introduction

Trees play an important role in the function of urban ecosystems. Beyond their role as a habitat for birds and other animals, trees provide an array of essential ecosystem services: stormwater management (Berland et al. 2017), temperature control (Coseo and Larsen 2014), air pollution reduction (Nowak, Crane and Stevens 2006), and carbon sequestration (Kendall and McPherson 2012), among other ecosystem services.

Recent research has focused on the relationship between rentership rates and the distribution of this environmental amenity in residential areas. The landmark research on this subject is from Perkins, Heynen and Wilson (2004), who examined a tree-planting program in Milwaukee and found a statistically significant negative correlation between rentership rates and residential canopy cover at the census tract level. Based on their review of the literature on differences between renter and owner-occupied housing, the authors suggest two factors that may produce this relationship: residential mobility, as renters are relatively transient and are unlikely to ever benefit from the trees they plant, and housing maintenance, as it is disadvantageous for renters to invest in improvements that enhance property values and may result in rising rents. Several other studies have investigated this relationship and have similarly found an inverse correlation between rentership and lower tree cover, in various cities at various spatial scales (Heynen, Perkins and Roy 2006; Landry and Chakraborty 2009; Koo et al. 2019). Although tree cover is not necessarily a perfect proxy for the ecosystem services provided by the urban forest, as Riley and Gardiner (2020) found in their comparison of tree canopy cover with a spatial measure of ecosystem service dollars, it remains a readily-available and widely-used figure to assess the distribution of these benefits.

One potential explanation for this relationship that appears under-studied, however, is the role of the built environment in shaping disparities in tree cover. Renters tend to live in neighborhoods with a greater proportion of impervious surfaces, as well as other land-use types that leave less space for planting trees; it seems reasonable to question the extent to which the observed relationship between renters and tree cover is merely the product of renters disproportionately living in neighborhoods where the built environment allows less space for the growth of urban trees. This paper addresses that gap in the literature by investigating the relationship between tree cover, rentership, and a variety of built environment variables in Chicago.

In this study, I found that considering aspects of the built environment, including single-family housing, housing age, and transit use, erases the apparent relationship between rentership and tree cover. While a "traditional" model using the socioeconomic indicators commonly used in the literature shows a negative relationship between rentership and tree cover in Chicago, I found that this relationship between rentership and tree cover may, in fact, be the product of other factors in the built environment.

This finding indicates that previously-accepted explanations for the relationship between tree cover and rentership — residential mobility, housing maintenance, and the increased political influence of homeowners discussed by Landry & Chakraborty (2009) — have to be re-evaluated in the light of this new evidence. While additional research is necessary to firmly establish that the observed relationship between rentership and tree cover is the product of urban form, these results provide a preliminary indicator that previous explanations for the spatial distribution of tree cover may not fully reflect all drivers of that distribution, requiring a re-evaluation of the broader literature around the distribution of environmental amenities.

Literature Review

Importance of the Urban Forest

Trees provide essential habitat to a variety of urban animals. While birds are perhaps the best-known example (Parsons, Major, and French 2006), trees host a variety of species: tree cavities provide shelter and food storage for several species (LaMontagne et al. 2015), animals ranging from cottontails (Abu Baker, Emerson, and Brown 2015) to arthropods (Yasuda and Koike 2009) and bats (Rhodes et al. 2006) to squirrels (van der Merwe, Burke, and Brown 2007) rely on urban trees for habitat. Many of the animals that characterize urban environments rely on urban trees, and even the spontaneous flora that tends to generate around street trees has been the subject of scientific study (Wittig and Becker 2010).

Beyond their benefits to wildlife, however, urban trees also play an important role by providing a wide array of ecosystem services. One is their crucial role in stormwater management: as Berland et al. (2017) explain, urban trees capture rain in their canopy that would otherwise fall to the ground, decreasing the impact of rainfall events on urban sewage networks. They go on to explain that trees also promote soil infiltration by loosening the soil through their root networks, as well as the absorption and evapotranspiration performed by the tree itself. This also has the beneficial side effect of nitrogen filtration, reducing nitrogen runoff that promotes algal blooms in lakes, rivers, and other bodies of water (Denman, May, and Breen 2006). While trees are often a feature of "green infrastructure" systems like bioswales, Berland et al. specify that these benefits apply to all urban trees, which can play a role in controlling stormwater independent of an engineered application.

Urban trees can also play a role in mitigating the "urban heat island" effect, the increased temperatures in urban areas compared to surrounding rural areas as the result of land use change. In Chicago, Coseo and Larsen (2014) found that tree canopy is the second most powerful explanatory variable for daily nighttime air temperatures, after only the proportion of impervious surface. Relative to other efforts to reduce urban temperatures, such as painting surfaces light colors that reflect heat, Rosenzweig et al. (2006) found that street trees provide the greatest cooling potential per unit area. This finding, however, should come with a note of caution, as some authors have found that efforts to increase light-colored surfaces can be deployed more widely than planting trees, suggesting this approach may result in a greater reduction in urban temperatures than tree-planting programs (Mackey, Lee, and Smith 2012). Cities, therefore, may see greater impacts by focusing on strategies other than planting trees if their priority is reducing urban temperatures. Nevertheless, a large-scale tree planting program in Chicago could reduce city-wide temperatures by up to 1.4ºC (Akbari, Pomerantz and Taha 2001). This has been a specific goal in Chicago's Climate Action Plan, which emphasized tree plantings by the Park District and Bureau of Forestry as a strategy for reducing urban heat (Coffee et al. 2010). Several local non-profit conservation organizations, such as the Chicago Region Trees Initiative and Openlands, have also embraced this goal, and academics including Drescher (2019) have urged us to understand disparities in tree cover, and the resulting impact on heat, as a matter of environmental justice.

Another important benefit of urban trees is their role in controlling urban pollution — both directly, through absorbing pollutants like ozone and sulfur dioxide, and indirectly, by reducing energy usage. Akbari, Pomerantz and Taha (2001) analyzed both the energy and smog reduction benefits of shade trees in urban environments and found seasonal cooling-energy savings of up to 30% and heating-energy savings of around 10-15%. By reducing urban temperatures, these trees could also reduce smog concentrations. Nowak, Crane and Stevens (2006) built on this work, estimating that the combined total effects of trees on air pollutants are significant, particularly for ozone and sulfur dioxide, and that tree planting should be a major part of efforts to address air pollution. Solecki et al. (2005) also found fairly substantial pollution reduction from urban vegetation, which McPherson et al. (1997) suggested could amount to $9.2 million in pollution reduction for Chicago. Vos et al. (2013) qualified that finding with an urban design note: they found that trees planted near roadways had the unintended effect of reducing ventilation, increasing the concentrations of pollutants along roadways.

That shielding effect along roadways, however, also provides another service to adjacent properties: reducing traffic noise. The canopy of street trees can reflect and contain high-frequency noise from automobile traffic, reducing the volume of traffic noise in adjacent structures (Jang et al. 2015). Even fairly narrow tree belts along highways have been successful in delivering significant reductions in road noise (Ow and Ghosh 2017).

Urban trees can also play a role in reducing carbon emissions, though their specific effect is disputed. Research by Jo and McPherson (2001) in Chicago found that trees adjacent to buildings could reduce energy demand by shading the building and deflecting wind, resulting in a reduction of carbon emissions between 3.2% and 3.9% for buildings with 33% tree cover, and -0.2% to 3.8% when tree cover was just 11%. Trees also absorb and fix carbon through the process of photosynthesis, though McGraw et al. (2010) found that this effect is fairly minimal; in Chicago, the carbon stored in urban trees amounts to just 0.3% of citywide emissions. The authors do argue that tree-planting programs, despite this minor effect, could still be worthwhile because they are relatively easily deployed and have significant additional benefits. The trees in

Los Angeles studied by Akbari, Pomerantz and Taha (2001) also achieved most of their impact on carbon emissions through energy savings, with the authors calculating that urban trees are the carbon equivalent of 3-5 trees in a rural forest because of the added energy savings from heating and cooling.

Beyond energy savings, whether urban trees actually sequester a net positive amount of carbon over the course of a lifetime remains an open question. Kendall and McPherson (2012) conducted a comprehensive life-cycle analysis of urban trees, considering inputs like transportation, fertilization, and the plastic containers used to store trees before planting. They found that these inputs add up to 20-50% of the carbon stored by a tree over its lifetime, contradicting findings by Nowak et al. (2002) that the regular maintenance required by urban trees makes them a net emitter of carbon. Both authors highlight the importance of end-of-life carbon control, such as using wood in products like lumber or furniture rather than quickly-decomposing mulch, to ensure that trees do not constitute a net carbon source rather than a sink.

Cumulatively, the value of these benefits is reflected, to some extent, in residential and commercial property values. Conway et al. (2010) used a novel model that controlled for spatial autocorrelation to evaluate the effect of "green cover" (determined by remote sensing) in Los Angeles and found that it substantially increased housing prices. Escobedo, Adams and Timilsina (2015) relied on site-specific field measurement rather than remote sensing, and still found that assessed property values increased, on average, by $1,586 per tree on the property. Donovan and Butry (2010) used a hedonic model and found that the number of street trees fronting a property also publicly influenced home values. While these new models offer different ways to evaluate the precise impact of trees and tree cover on property values, studies since Anderson and Cordell (1988) have consistently found that urban trees increase nearby property values.

Interestingly, however, public opinion on urban trees does not seem to be driven by the value of any of the services trees provide, or even by their effect on property values. Peckham, Duinker and Ordóñez (2013) conducted extensive survey research and found that urban residents value trees because of a perceived "naturalness," as well as other aesthetic and psychological benefits. While many residents also mentioned their role as wildlife habitat, benefits like property values and carbon storage were only mentioned rarely. Survey research by Zhang et al. (2007) confirmed that these perceived aesthetic and psychological benefits play a strong role in shaping where people live, with 75% of Alabama residents saying trees on their property are important in selecting a home, while 77% said trees in a community were important in selecting a community to live in.

Spatial Inequities in the Urban Forest: The Case of Renters

Several studies have examined the relationship between vegetation and concentrations of marginalized communities, such as low-income communities or communities with a large proportion of minority residents, using spatial patterns to identify distributional inequalities. Among the oldest is the work in New Orleans performed by Talarchek (1990), which found a strong negative relationship between tree cover and both percentage of non-white population and poverty rate, while other work has found that high canopy cover also correlates with higher levels of education and older housing stock (Heynen and Lindsey 2003). A relationship between income, education, and dense tree cover has been established in a variety of contexts, including in Brazil (Pedlowski 2002) and Canada (Greene et al. 2018). While these studies did not specifically examine the question of renters, they collectively provide early evidence that neighborhood tree cover can be influenced by the social and economic composition of that neighborhood.

The negative relationship between renters and the urban forest has since been well-established, in part because rentership often correlates closely with the socioeconomic metrics used in other studies on this subject (Vlist et al. 2002). While the Perkins, Heynen and Wilson (2004) study established the pattern of high rentership correlating with low tree density in their study of Milwaukee's tree-planting program, later work in Milwaukee found that this relationship applies more broadly to residential canopy cover throughout Milwaukee, beyond the context of the planting program (Heynen, Perkins and Roy 2006). In some cities, residential programs may simply exclude renters as a matter of course, or through requirements for participation that are onerous for renters, such as proof of homeowners insurance (Ragsdale 2012).

This pattern reoccurred in research by Landry and Chakraborty (2009) in Tampa, Fla., who found that tree cover in residential rights of way displayed a negative correlation with neighborhoods with higher proportions of renters. They suggested that this resulted, in part, from the mechanism Perkins identified: renters may be less willing to plant trees on rights of way adjacent to their residence because of residential mobility and housing maintenance. They also suggested that the effect of trees on property value may lead homeowners to use their political influence to demand public tree planting in neighborhood rights of way. The "opportunity cost" of trees on private land, occupying space that could otherwise be used for a swimming pool or outdoor patio, means that homeowners may see a higher net benefit from public trees than private ones (Pandit et al. 2013).

Beyond these studies, several others have comprehensively established a relationship between several metrics of urban vegetation and renter-occupied housing. Raciti, Hutyra and

Newell (2014), for instance, used remote sensing data and field observations to evaluate both canopy cover and carbon storage potential, which they found was negatively correlated with the percentage of renters and no other neighborhood demographic indicators. Li et al. (2016) used an innovative methodology, mapping street greenery through Google Street View, that found a significant and positive association between owner-occupied units and both private yard vegetation and tree coverage. A rare longitudinal study from Koo et al. (2019) found that Atlanta's urban canopy consistently displayed a negative relationship with the proportion of renters in a neighborhood in both 2000 and 2013, even as the city's demographics changed and the relationship between African-American and Hispanic population and tree cover shifted from a negative to a positive correlation.

Complicating what seems to be a well-established relationship is research that suggests historic demographic patterns influence tree cover. Troy et al. (2007) found that rates of owner-occupied housing in inner-city Baltimore correlated positively with yard stewardship and expenditures, but not tree stewardship. The authors suggested a "legacy effect," where trees planted before white flight altered the demographic composition of the neighborhood contribute to present-day tree canopy. Later work by Boone et al. (2010) established this relationship, again in Baltimore, finding that historic demographic patterns are more predictive of the current urban canopy than present demographics.

There are a handful of examples of cities where there is not a clear relationship between housing type and tree cover, but, in each case, the relationship can be explained by unique characteristics of that city that are unlikely to apply to Chicago, the focus of this study. Krafft and Fryd (2016) found that past rates of homeownership showed a positive relationship with future tree cover, but also found that homeownership displayed a negative relationship with tree cover over the decade they studied in five local government areas in Melbourne, Australia. This finding contradicts past research, but the researchers explained that the largest of the observed changes may be the product of industrial areas in a specific LGA being redeveloped into new residential developments, increasing the area suitable for planting trees. Chicago, by contrast, is nearly five times larger in area than the total area studied in Melbourne and has not experienced a spate of large-scale industrial redevelopment, so those factors are unlikely to apply, though studying the relationship in Chicago after several currently-planned redevelopment projects (Lincoln Yards, the 78) are complete would provide an interesting case study to investigate that finding. In their study of six eastern Australian cities, Kirkpatrick, Daniels and Davison (2011) explain their finding that trees are predicted by a high proportion of renters in similar terms, suggesting that renters are concentrated in recently-gentrified areas. In Chicago, by contrast, more than half of all residents are renters, with large populations of renters in all income and age categories (2019 State of Rental Housing), a population that includes more than the "university students, young managers and professionals" referenced by Kirkpatrick, Daniels and Davison. Both Australian studies do confirm the importance of past land use remarked on in other studies (Boone et al. 2010; Troy et al. 2007), as both found that the seeming discrepancies between their findings and the literature could be explained by recent shifts in the distribution of certain demographics.

Demographics, however, may not be the only city-specific explanation for similar puzzling results; Pham et al. (2013) speculated that the positive correlation between renters and backyard vegetation observed in Montréal may be the product of the city's history as a "city of tenants," where home ownership has historically been rarer than in other North American major cities, as well as a unique mix of housing types where high-rise buildings are often bordered not by other high-rises, but by owner-occupied detached houses with lots of area for vegetation.

Another possible explanation for variation between cities is different legal frameworks that may disincentivize homeowners from planting trees. In the city of Portland, Ore., Ramsey (2019) found no significant relationship between owner-occupied housing and tree canopy, which he suggests is because Portland's municipal code establishes that the city owns all trees in rights-of-way, but homeowners are responsible for all maintenance and upkeep. Fischer and Steed (2008) mention that ownership and management duties for trees in the public right of way may vary between municipalities, streets, and even road segments, potentially explaining some variation between cities, though no study has directly examined this effect across multiple cities.

Spatial inequalities: the product of preferences?

Grove et al. (2006) introduced the concept of "neighborhood lifestyle characteristics," finding that lifestyle behavior — not merely demographic variables — is the best predictor of tree cover on both private lands and public rights-of-way in Baltimore. These characteristics, developed for marketing, classify households into sixty-two consumer categories in an attempt to capture the complexity of American social class. In explaining their finding, the authors said that household land management decisions may be driven by a desire to "uphold the prestige of the household's neighborhood," suggesting that neighborhood inequalities may be the product of different values of urban trees assigned by lifestyle groups. These social class distinctions may be able to explain seemingly counterintuitive results, such as the patterns identified by Locke et al. (2016) in Philadelphia, where neighborhoods with more renters tended to have more tree canopy, except in areas of higher parcel market value, where the relationship was reversed.

The idea that renters may be less invested in upholding their neighborhoods, however, seems to contradict research directly investigating the preferences of renters. Conway, Shakeel and Atallah (2011) found that, in Toronto, residents' associations with higher percentages of owner-occupied dwellings were more likely to engage in forestry activity, but they specify that there was no relationship between race, language, or immigration and forestry activity, suggesting that the disparity does not reflect cultural preferences. In New Haven, Conn., while existing tree canopy displayed a moderately negative association with the percentage of renters, requests for new trees came equally as often from neighborhoods where renting is commonplace, suggesting renters are at least as interested in homeowners in developing their neighborhood's canopy. Additionally, survey research by Winter (2017) found that both homeowners and renters felt overwhelmingly positive about having trees on their property, with no statistically significant difference between the two groups on that question, while Donovan and Butry (2011) found no difference in price elasticity with regard to street trees among both renters and homeowners, suggesting both groups value street trees similarly. Further survey research by Conway and Bang (2014) found no relationship between rentership and support for a variety of municipal tree-planting policies, and intriguing research by Donovan and Mills (2014) found that long-tenured homeowners are actually *less* likely to invest in their property by planting street and yard trees than newer homeowners. While some of the disparity between renters and homeowners may be the product of the social class distinctions identified by Grove et al. (2006), the lack of any differences between renters and homeowners in multiple studies using diverse methodologies to evaluate their preferences for urban vegetation suggests that some mechanism related to renting, beyond lifestyle preferences, must be responsible for the observed differences in canopy cover.

There is a possibility that opposition to urban forestry programs among renters may reflect concern about "green gentrification," where the development of environmental amenities threatens to raise property values and produce displacement (Dooling 2009; Checker 2011; Wolch, Byrne and Newell 2014). This pattern, where environmental amenities burden established low-income residents, has been described as an "environmental rent gap" (Anguelovski et al. 2019). In this theory, renters may have similar preferences as homeowners for urban vegetation, but may be suspicious of organized tree-planting efforts that may serve as a harbinger of higher rents and displacement.

The initial problem with the green gentrification argument is that trees do not appear to have a relationship with property values that is significant enough to produce large-scale displacement. Trees certainly do have an impact on property values, but an assessment of a tree-planting program in Los Angeles by McPherson et al. (2008) found that the economic impact of an individual tree on nearby property values was negligible: $1,100 to $1,600 over the course of a 35-year period, or less than $50 in added property values per year. Research by Donovan and Butry (2011) in Portland, Ore., found that having trees on a lot tended to increase monthly rents by an average of $5.62, while adjacent public trees increased rents by around $21. While that small increase in rent may make a difference to a low-income family on the margin, it seems unlikely that tree plantings can be responsible for large-scale "green gentrification." One resident of a lower-income neighborhood in Baltimore interviewed by Battaglia et al. (2014) mentioned his concerns about tree planting presaging gentrification, but comprehensive research on attitudes surrounding this phenomenon are scarce. Residential opposition certainly shapes the distribution of some tree-planting programs (Carmichael and McDonough 2018), but teasing out the relationship between past negative experiences with city tree maintenance, concerns about gentrification, and the renter/homeowner dynamic requires additional research.

Overall, the literature does not support the theory that the preferences of owners and renters is sufficient to explain the observed relationship between areas of high rentership and low canopy cover. While the "green gentrification" literature provides a theoretical framework by which renters may oppose tree-planting programs that may result in higher rents, economic research shows that the impact of trees on rents is negligible, while survey research suggests that renters do not differ from homeowners in their preferences for urban vegetation.

Forest inequalities and the built environment

One interesting note from Pham et al. (2013) is their finding that the most important factors determining urban vegetation in Montréal are characteristics of the built environment, such as urban form and land-use types, that determine the space available for planting. Shakeel and Conway (2014) actually found that available planting space, along with resident attitudes, shows a strong correlation with both canopy cover and tree density in their study of four neighborhoods in suburban Toronto, while the traditional suite of socioeconomic variables showed no significant relationship. Jesdale, Morello-Frosch and Cushing (2013) and Solecki et al. (2005) similarly found that renters are both more likely to live in areas of both no tree cover and areas of high impervious surfaces (where trees cannot grow), while Ossola et al. (2019) recently found a relationship between vegetation structure and architectural style, as architecture determines the physical availability of planting space. The importance of space is underscored by Heckert and Rosan (2016), who found that efforts to develop "green infrastructure" in Philadelphia were more difficult in neighborhoods where residents were unlikely to own their own property, both because of the program's structure and because many of those residents lived on properties that simply did not have room for green infrastructure, a category that includes street trees.

In summary, the existing literature on the topic shows a clear relationship between rentership and tree cover in a variety of urban areas: higher levels of tree cover, an important environmental amenity, are disproportionately present in areas with fewer renters. The mechanism behind this relationship, however, is uncertain. Some authors have suggested that this inequity results from characteristics unique to rentership, such as the higher mobility of renters or the impact of trees on property value. The role of built form, though only a handful of studies have identified it as an influential factor, presents an alternative explanation for this relationship: renters may live in areas where the built environment leaves little room for trees.

The current study aims to further investigate the relationship between rentership and the built environment. Through the use of neighborhood-level data on a number of aspects of the built environment in Chicago, including impervious acreage, auto dependence, walkability, housing size and housing values, I will be able to analyze the role of the built environment in shaping the relationship between renters and tree cover. This aspect of that relationship is not frequently discussed in the literature, allowing this study to make a unique contribution to the current conversation.

Methodology

Study Area

The study area is the City of Chicago. While most other studies that assess the distribution of urban forests do so at the Census tract or block group level, I use the community area, a neighborhood-equivalent unit unique to Chicago. Chicago is divided into seventy-seven community areas, ranging in size from 1.61km^2 to 27.71km^2, with an average is 7.6km^2 (depicted in Figure 1a). These geographic units have been used as statistical units by a wide variety of researchers and governments since they were delineated by the Social Science Research Committee at the University of Chicago in the 1920s. The Chicago Department of Public Health, for example, presents their Chicago Health Atlas by community area, while the Department of Planning and Development's Green Healthy Neighborhoods initiative focuses on specific community areas on the South Side. The definition of these areas has remained constant, with only two changes since their original publication. The barriers between community areas often continue to reflect the socioeconomic lines of segregation that divide Chicago; in 2010, only around a third of community areas qualified as "integrated," and the boundaries between community areas are often recognized by city residents (Emmanuel, Cruz and Unrau 2017). By conducting my analysis at a scale that approximates local neighborhood areas, I am able to examine the urban forest at the scale most often used in urban planning (Talen 2019). By using a locally meaningful definition of community, furthermore, I am able to present my findings in a way that will resonate with local policymakers and residents.

In order to ensure that the relatively large unit of the community area does not cause this study to miss important distinctions that occur at a finer scale, as noted by Locke et al. (2017), I replicated my procedure at the Census block group level for all block groups (2,335) that overlap the City of Chicago. Block groups range in area from 0.004 km^2 to 17.74 km^2, with an average of 0.30 km^2. The block groups used in this study are depicted in Figure 1b.

Figure 1a. Chicago community areas

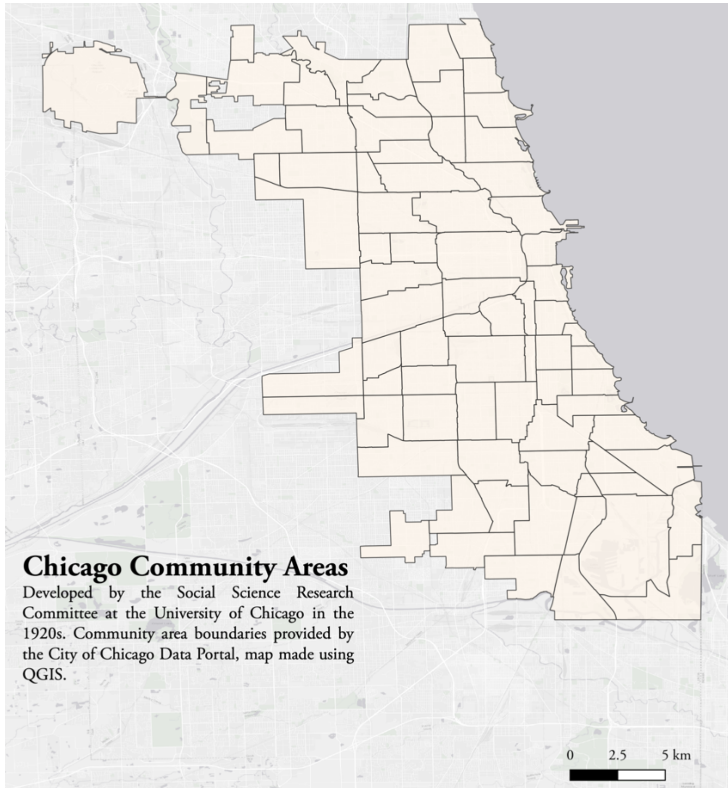
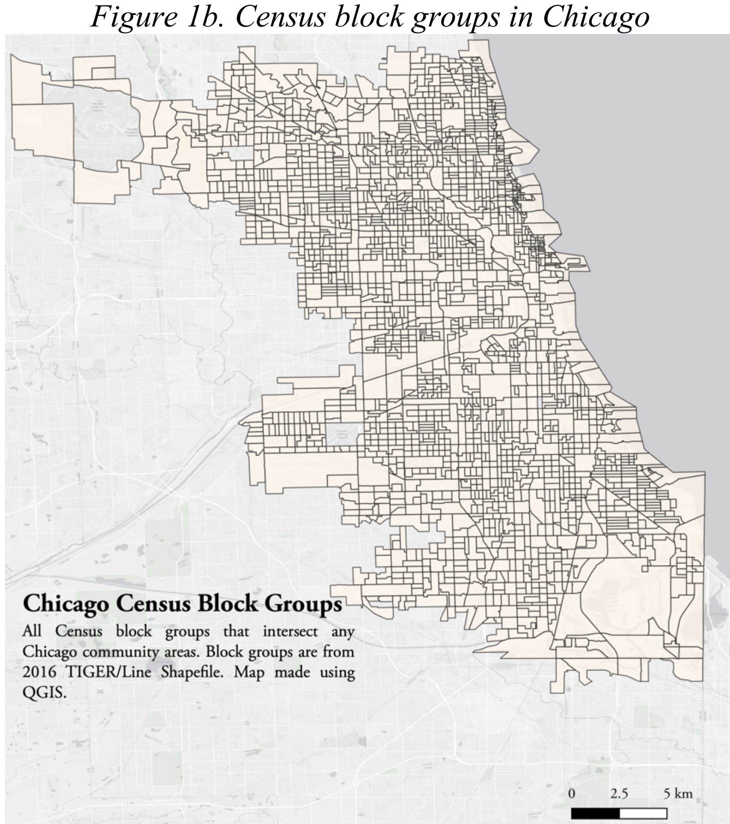


Figure 1b. Census block groups in Chicago



Data

This paper relies on a seven-class Chicago regional land cover dataset produced by the University of Vermont's Spatial Analysis Laboratory (SAL) and published in 2016. This dataset uses LiDAR and high-resolution imagery ($1m^2$) from a range of years to classify the entire study area into seven land cover categories: tree canopy, vegetation (foliage under 10 feet), bare soil, water, buildings, roads/railroads, and other paved surfaces. Tree canopy overhanging other classes was assigned to the tree canopy category. This data was also used to calculate impervious acres per household at the block group level. This data is the most detailed and accurate land cover dataset for Cook County. For every community area and block group, the percentage of land area covered by tree canopy was calculated using the SAL's Tree Canopy Assessment Tool in ArcGIS.

At the community area scale, almost all demographic, housing, land use, and other variables are drawn from the Community Data Snapshots prepared by the Chicago Metropolitan Agency for Planning (CMAP). I used the November 2018 release, as it includes data through 2016, the year the land cover dataset was published. The underlying data is primarily drawn from the 2012-2016 American Community Survey (ACS) 5-Year Estimates, which CMAP prepared by aggregating ACS estimates from the census tract and block-group level to the community area level. When possible, I attempted to use data collected in the year 2016 in order to avoid the uncertain geographic context problem (Kwan 2012a, 2012b). Two additional variables were prepared by CMAP: annual vehicle miles traveled per household, a metric of automobile dependency that serves as a proxy for automobile-oriented land use patterns, was calculated by CMAP from data from the ACS, the Illinois Environmental Protection Agency, and the Illinois

Secretary of State, while open space per 1000 residents was calculated by CMAP from ACS data and their own land use inventory. This data allows me to account for variation in community area demographics and investigate which of these demographic variables are correlated with tree cover and health.

Three additional variables, individual poverty rate, the percentage of residents living in crowded housing and the percentage of residents paying more than 35% of their income on housing costs, were provided by the City of Chicago's Health Atlas. The city calculated these variables at the community area level from the ACS 5-Year Estimates for 2012-2016.

I joined the data discussed in the above two paragraphs to a shapefile of Chicago's community areas downloaded from the City of Chicago's Data Portal in 2019. I then added the land use percentages, which I calculated from the Chicago regional land cover dataset for each community area using the University of Vermont's Tree Canopy Assessment Tool in ArcGIS, to produce a single file containing all metrics of tree distribution and demographic variables by community area.

I replicated this procedure at the block group level, using 2012-2016 ACS 5-Year Estimates for all variables included at the community area level, with the exception of average vehicle miles traveled and open space per capita, which were not available through the ACS. I joined this data to a TIGER/Line shapefile of all 2,325 populated block groups partially or entirely within the City of Chicago, then added the land use percentages.

Variable selection

The group of explanatory variables were chosen based on their use in previous work on the topic and evidence of some association with urban tree cover. Specific variables, along with the previous work that used those variables, are detailed in Tables 1a and 1b (separated into socioeconomic and built environment variables for ease of reading). All variables in Table 1a are drawn from prior studies that used the same or similar variables. Table 1b includes several novel variables reflecting aspects of housing and transportation not present in prior studies on the topic.

Table 1a: Socioeconomic variables

Variable name	Previous studies using this variable	Data source
Median age	Landry & Pu 2010, Landry & Chakraborty 2009, Shakeel & Conway 2014	2012-2016 American Community Survey (ACS) 5-Year Estimates (prepared by CMAP at the community area level)
White percentage	Landry & Pu 2010, Locke & Baine 2015, Ramsey 2019, Li et al. 2016	2012-16 ACS (CMAP)
African-American percentage	Perkins, Heynen & Wilson 2004, Duncan et al. 2014, Landry & Chakraborty 2009, Ramsey 2019, Koo et al. 2019, Li et al. 2016	2012-16 ACS (CMAP)
Hispanic percentage	Ramsey 2019, Duncan et al. 2014, Landry & Pu 2010, Landry & Chakraborty 2009, Koo et al. 2019, Li et al. 2016	2012-16 ACS (CMAP)
Asian percentage	Shakeel & Conway 2014, Ramsey 2019, Koo et al. 2019	2012-16 ACS (CMAP)
High school diploma or higher	Ramsey 2019, Locke & Baine 2015	2012-16 ACS (CMAP)
Bachelor's degree or higher	Pham et al. 2013, Shakeel & Conway 2014, Ramsey 2019, Li et al. 2016, Riley & Gardiner 2020	2012-16 ACS (CMAP)
Median household income	Perkins, Heynen & Wilson 2004, Pham et al. 2013, Locke & Baine 2015, Greene, Robinson & Millward 2018, Landry & Chakraborty 2009, Ramsey 2019, Riley & Gardiner 2020	2012-16 ACS (CMAP)
Unemployment rate	Ossola et al. 2019	2012-16 ACS (CMAP)
Poverty rate	Duncan et al. 2014, Koo et al. 2019, Riley & Gardiner 2020	2012-16 ACS (Chicago Health Atlas)
Population density	Pham et al. 2013, Duncan et al. 2014, Locke & Baine 2015,	2012-16 ACS (CMAP)

	Ramsey 2019, Riley & Gardiner 2020	
Rentership percentage	Perkins, Heynen & Wilson 2004, Ramsey 2019, Pham et al. 2013, Landry & Pu 2010, Locke & Baine 2015, Landry & Chakraborty 2009, Shakeel & Conway 2014, Koo et al. 2019, Li et al. 2016, Riley & Gardiner 2020	2012-16 ACS (CMAP)
Percentage foreign-born	Pham et al. 2013	2012-16 ACS (CMAP)
Linguistic isolation	Pham et al. 2013 (as "recent immigrants")	2012-16 ACS (CMAP)

Table 1b: Built environment variables

Variable description	Papers using variable	Data source
Percentage of homes built before 1940	Pham et al. 2013, Landry & Pu 2010, Landry & Chakraborty 2009, Shakeel & Conway 2014, Ramsey 2019, Koo et al. 2019	2012-16 ACS (CMAP)
Detached single-family homes (percentage)	Pham et al. 2013, Landry & Pu 2010, Shakeel & Conway 2014	2012-16 ACS (CMAP)
Median number of rooms		2012-16 ACS (CMAP)
Median house value	Landry & Pu 2010	2012-16 ACS (CMAP)
Severe housing cost burden (35%+)		2012-16 ACS (CHA)
Housing units with more than one person per room (percentage)		2012-16 ACS (CHA)
Vacancy rate	Landry & Pu 2010, Heynen, Perkins & Roy 2006	2012-16 ACS (CMAP)
Impervious area per household (m^2)	Heckert and Rosan 2016	UVM SAL
Trips to work via non-SOV modes		2012-16 ACS (CMAP)
Open space per 1000 residents (acres)	Pham et al. 2013, Duncan et al. 2014, Shakeel & Conway 2014, Heckert and Rosan 2016	CMAP
Average vehicle miles traveled (VMT)		CMAP

Descriptive statistics

Table 2 includes descriptive statistics for all variables used at both the community area (CA) and block group (BG) level. Average vehicle miles traveled and open space per 1000 residents are not available at the block group level because they were calculated by CMAP at the community area level.

Table 2: Descriptive statistics of all variables

	Mean		Standard Dev.		Min		Max	
Variable name	CA	BG	CA	BG	CA	BG	CA	BG
Tree canopy cover	19.19%	20.01%	7.20%	8.53%	7.27%	0.59%	48.78%	76.20%
Median age	35.41	36.24	4.80	8.40	21.30	13.9	47.12	85.4
White percentage	27.94%	32.71%	27.24%	31.64%	0.38%	0.00%	88.66%	100%
African-American percentage	38.25%	34.73%	39.59%	40.59%	0.47%	0.00%	99.06%	100%
Hispanic percentage	26.07%	25.99%	28.14%	29.79%	0.00%	0.00%	92.62%	100%
Asian percentage	5.99%	5.12%	10.71%	9.77%	0.00%	0.00%	75.18%	97.11%
High school diploma or higher	82.05%	82.87%	10.71%	13.74%	50.36%	30.81%	98.39%	100%
Bachelor's degree or higher	30.27%	32.92%	21.25%	25.59%	5.00%	0.00%	82.77%	99.33%
Median household income ($)	48931	54795	22166	31781	14287	0	108146	207969
Unemployment rate	13.65%	12.67%	8.08%	11.07%	3.22%	0.0%	36.93%	91.58%
Poverty rate	22.98%	21.44%	12.21%	16.15%	1.6%	0.0%	65.8%	92.79%
Population density (/km^2)	5018	7905	2676	9126	380	58	12330	253318
Rentership percentage	52.83%	52.26%	19.05%	24.48%	9.82%	0.00%	89.58%	100%
Percentage foreign-born	20.38%	18.93%	15.94%	16.68%	0.88%	0.00%	62.33%	96.15%
Linguistic isolation	13.95%	8.48%	13.03%	10.56%	0.36%	0.00%	53.11%	66.18%
Percentage of homes built before 1940	41.25%	44.70%	31.76%	25.72%	1.40%	0.00%	83.38%	99.14%
Detached single-family homes (percentage)	33.67%	33.75%	25.59%	30.48%	1.87%	0.00%	88.17%	100%
Median number of rooms	5.44	4.99	0.74	0.96	3.72	0.0	7.20	9.0
Median house value ($)	213351	242221	94392	151417	56875	0	488678	1104200
Vacancy rate	13.20%	12.67%	6.88%	10.40%	4.75%	0.00%	35.44%	70.00%
Severe housing cost burden (35%+)	37.09%	44.08%	9.14%	23.28%	14.6%	0.00%	56.0%	100%
Impervious area per capita (m^2)	153.50	142.28	138.30	335.18	48.41	3.69	1051.55	11949.05
Housing units with more than one person per room (percentage)	4.37%	4.31%	2.99%	5.53%	0.5%	0.00%	14.3%	41.14%
Trips to work via non-SOV modes	44.16%	44.25%	13.24%	18.47%	14.16%	0.00%	75.34%	100%
Open space per 1000 residents (acres)	2.91	n/a	2.83	n/a	0.02	n/a	15.59	n/a

Average vehicle miles traveled (VMT)	12639	n/a	3974	n/a	6581	n/a	31817	n/a

Regression diagnostics and analysis

In order to quantify the relationship between variables, I used the software package GeoDa (version 1.14.0.10) to perform a three-step process. First, I conducted two ordinary least squares (OLS) regressions, one using the covariates in Table 1a and the other using the covariates in Tables 1a and 1b. Percent canopy cover was the dependent variable in both cases.

The second step was to test for spatial autocorrelation, which occurs when values at certain locations are more similar to (or different from) nearby values than a random distribution would produce, violating the assumption of independent observations used in standard models. Failure to identify and account for spatial dependency can produce inaccurate regression estimates and higher standard errors (Schwarz et al. 2015). These errors can impact the conclusions of studies: Duncan et al. (2014) found that an OLS regression in Boston indicated a significant inverse relationship between black neighborhoods and tree density, but found no significant relationship once they accounted for spatial autocorrelation.

A Moran's I test was used to test for spatial autocorrelation. If the Moran value is near zero, there is little or no spatial autocorrelation, while a value close to -1 suggests that areas with large and small values of canopy cover are likely to be neighbors, while a value close to 1 suggest that adjacent neighborhoods are likely to have similar tree cover.

The third step, if spatial autocorrelation is present, is to use the variables in the original OLS regression in a new spatial autoregression model, chosen using the process described in Anselin (2005). (For more detail, see Appendix 1.) By controlling spatial dependence, I can improve the model fit and generate a model that does not violate the assumption that observations are independent. In both cases, I used a queen's contiguity spatial weights matrix with one order of contiguity, which treats community areas as neighbors if they share a boundary or a corner.

Results

Regression output

An ordinary least squares (OLS) regression, considering all covariates in Table 1a with canopy cover as the dependent variable, displayed substantial spatial dependence, with a remarkably high Moran's I value of 5.45 ($p<0.001$). The Lagrange multiplier tests for lag ($p=0.00005$) and error ($p=0.00018$) are both significant, which provides further confirmation that spatial dependence is present in the data. The robust Lagrange multiplier test for lag is not as significant ($p=0.12$), but substantially more significant than the robust Lagrange multiplier test for error ($p=0.70$), both of which suggest that adding a spatially lagged dependent variable will do more to correct for spatial dependence than adding a spatially lagged error term. In particular, the results of the robust Lagrange multiplier test for error suggests that most of the error dependence detected in the simple LM test would be addressed through a spatial lag model. The Lagrange multiplier test for a spatial autoregressive moving average (SARMA) is also significant ($p=0.00027$), but less so than either the standard LM-lag or LM-error tests. It is likely that the LM-SARMA statistic is simply detecting the need for a spatial lag or error model, rather than suggesting the need for a higher-order model such as those discussed by Elhorst (2010). Using the decision rules from Anselin (2005), these results suggest that adding a lag dependent variable would address the error dependence. As a result, this paper relies on a spatial lag model (SAR_{lag}) in order to control for spatial dependence.

Table 3 shows the regression result of two models: the SAR_{lag} model with canopy cover as the dependent variable, considering only the demographic variables used in prior literature (the "socioeconomic model"), as well as a SAR_{lag} model that incorporates additional variables that reflect characteristics of the built environment (the "combined model").

Table 3: SAR_{lag} model results for canopy cover at the community area scale

	Socioeconomic model			Combined model		
Variable	*Coefficient*	*Standard error*	*z value*	*Coefficient*	*Standard error*	*z value*
Rho	0.483***	0.111	4.35	0.0952	0.0951	1.00
Constant	122	99.8	1.22	-196*	80.6	-2.43
Rentership percentage	-28.3**	9.03	-3.14	62.9***	13.7	4.58
Median age	0.135	0.260	0.520	0.529*	0.21	2.56
White percentage	-99.0	97.1	-1.02	78.6	71.5	1.10
Black percentage	-104	97.3	-1.07	68.4	71.5	0.957
Hispanic percentage	-110	94.3	-1.17	70.9	71.1	1.00
Asian percentage	-124	93.6	-1.33	67.1	71.6	0.937
High school diploma percentage	-13.5	21.6	-0.625	4.68	15.8	0.295
Bachelor's degree percentage	28.8*	11.6	2.49	58.4***	9.65	6.05
Median income	-0.000104	0.000115	-0.902	6.05e-05	0.000101	0.600
Unemployment rate	2.96	18.2	0.163	19.6	13.2	1.49
Poverty rate	0.378	0.209	1.81	0.299*	0.146	2.04
Population density	-0.000135	0.000319	-0.424	-0.000174	0.000348	-0.500
Foreign born	-21.2	19.5	-1.09	45.9**	16.3	2.82
Linguistic isolation	35.7	33.4	1.07	-35.5	24.2	-1.47
Percentage of housing units built before 1940				6.48	4.41	1.47
Single-family detached housing (% of all units)				38.5***	9.95	3.87
Median number of rooms				8.48***	1.90	4.46
Median house value				-2.22e-05	1.81e-05	-1.23
Housing cost burden				0.272**	0.129	2.11
Crowded housing				0.649**	0.331	1.96
Vacancy rate				-8.84	13.3	-0.664
Impervious surface per capita				-0.00679	0.00662	-1.03
Open space per 1000				0.293	0.209	1.40
Average vehicle miles traveled				-0.000181	0.000284	-0.638
Commute via non-SOV modes				-30.7**	10.3	-2.97
Number of community areas	77			77		
R-squared	0.535			0.835		
Log likelihood	-234			-192		
Akaike information criterion	500			438		
Schwarz criterion	538			501		
Breusch-Pagan test	85.4***			37.0		
Likelihood ratio test	14.9***			0.949		

*p<0.05, **p<0.01 ***p<0.001

The results of the demographic model indicate that, of the variables tested, only rentership and 4-year college education display any significant association with urban tree cover. Based on the literature, these results make sense: education tends to positively correlate with tree cover, while rentership tends to negatively correlate, both confirmed in these results. Once the built environment variables are added, however, foreign-born population, poverty rate and median age also display a significant association with tree cover, as do trips to work via modes other than single occupancy vehicle (carpooling, transit, bicycling or walking), crowded housing rate, housing cost burden, median number of rooms and the percentage of single-family homes. The increase in R-squared and log likelihood values and decrease in the AIC and Schwarz criterion also demonstrate that the combined model is a better fit. The lower, less significant value of the Breusch-Pagan test also indicates that heteroskedasticy is less of a problem in the combined model. The relatively large coefficient for *rho* in the demographic model indicates that the spatial lag term may be standing in for other important variables, while the much smaller coefficient in the combined model suggests at least some of those variables have been addressed in the new model.

Figure 3. Rentership and trips by non-single-occupancy vehicle by community area

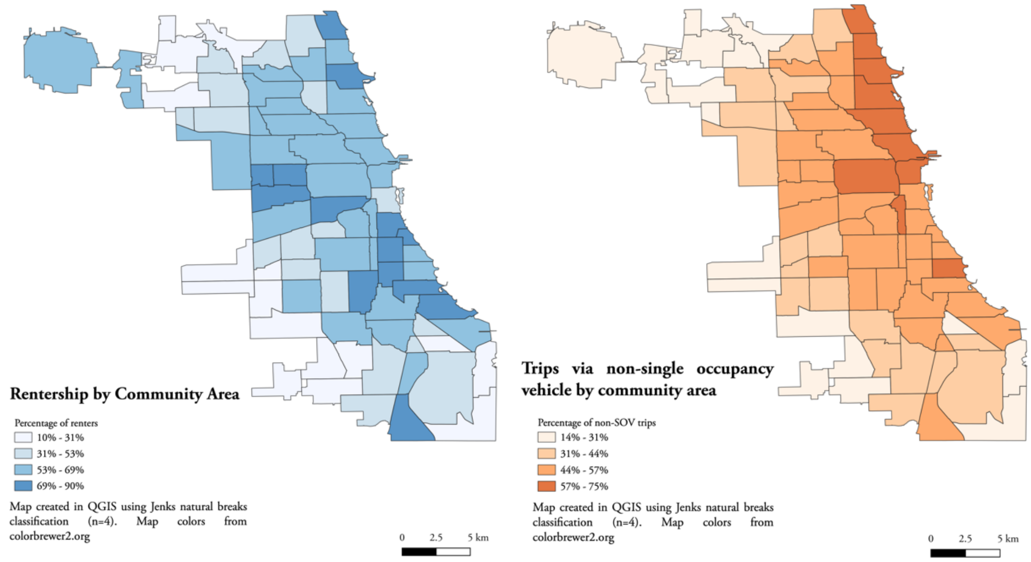


Figure 4. Canopy cover by community area

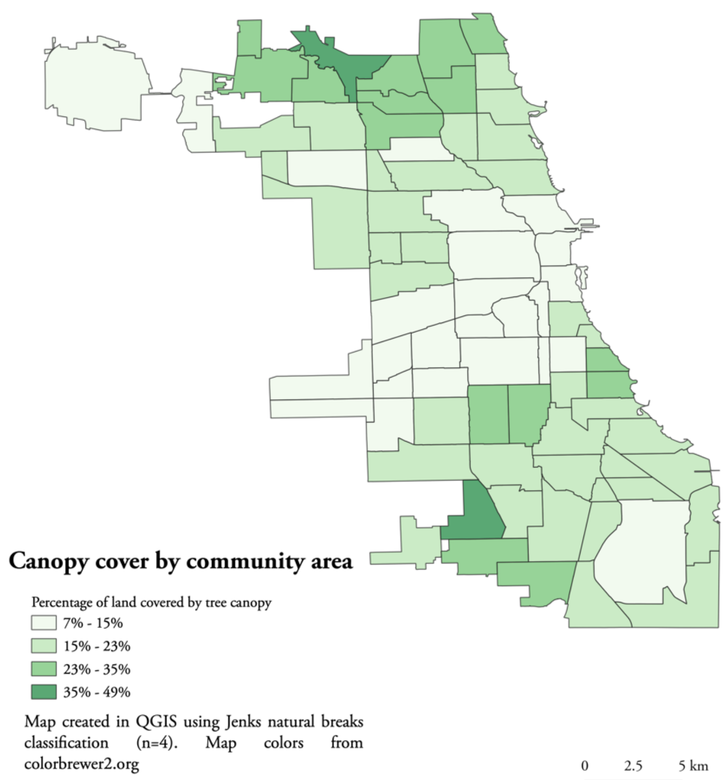


Figure 3 displays how this relationship functions spatially: the map of rentership, on the left, looks fairly similar to the map of non-single-occupancy vehicle commutes. Areas where few people rent are also areas where the largest percentage of people commute by single-occupancy vehicle — and, as Figure 4 displays, these are also the areas with the most tree cover. This relationship, however, only goes so far: while the significance of the built environment variables confirms the hypothesis that these variables could explain a significant amount of the variation in tree cover, it is difficult to speculate why age and foreign-born percentage is also significant in the combined model. Notably, several demographic variables often used in past research, including race, income and population density, displayed little relationship to canopy cover in either model, though this may simply be the result of the small sample size of 77 community areas. Similar past studies have relied on larger samples: Koo et al. (2019) included 288 block groups in their study of Atlanta, Duncan et al. (2014) used 167 Census tracts in their study of Boston, and Ramsey (2019) used 442 block groups in his study of Portland, Ore.

Scale sensitivity

The aggregation to the community area level (as well as the small sample size) may have covered up important variation that explains the relatively low number of significant variables and the lack of significance for race, income and population density in the community area model. The boundaries of community areas, while based on community boundaries determined by sociologists, are ultimately arbitrary units, which raises the possibility of ecological fallacy problems (Openshaw 1984). In order to test this, the procedure was replicated at the block group level, the smallest geographic unit for which most of the data used was available, using the 2,325 populated block groups that overlap with the boundaries of the City of Chicago.[1] The only modification to the procedure described in the methodology section was the use of a spatial weights matrix with two orders of contiguity rather than one to account for the smaller scale of block groups. An OLS regression, considering all covariates in the combined model with canopy cover as the dependent variable, displayed substantial spatial dependence (Moran's I=28.6, $p<0.001$). The Lagrange multiplier tests for lag ($p<0.00001$) and error ($p<0.00001$) are both significant, indicating that spatial dependence is present. The robust LM test for lag ($p<0.00001$) and error ($P=0.00002$) are both highly significant, as is the Lagrange SARMA ($p<0.00000$); though the difference between the two is extremely slight, the results of the robust tests suggest using the SAR_{lag} model.

[1] I intended to also replicate this procedure using Census tracts, but I lost access to the library computers with ArcGIS that I would have used to do so as a result of the 2020 coronavirus pandemic.

Table 4: SAR_{lag} model results for canopy cover at the block group scale

	Socioeconomic model			Combined model		
Variable	*Coefficient*	*Standard error*	*z value*	*Coefficient*	*Standard error*	*z value*
Rho	0.701***	0.0304	23.0	0.339***	0.0223	15.2
Constant	1619.02***	5.09	3.73	35.9***	3.70	9.70
Rentership percentage	-3.51 ***	0.835	-4.20	1.73	0.890	1.94
Median age	0.0392	0.0208	1.89	0.0376*	0.0148	2.54
White percentage	-10.6*	4.61	-2.29	-7.50*	3.12	-2.41
Black percentage	-11.0*	4.59	-2.39	-7.48*	3.10	-2.41
Hispanic percentage	-11.9**	4.56	-2.61	-7.91*	3.09	-2.56
Asian percentage	-14.4**	4.92	-2.93	-9.18**	3.33	-2.76
High school diploma percentage	-3.58	1.91	-1.87	-2.61*	1.33	-1.97
Bachelor's degree percentage	4.82***	1.17	4.13	6.12***	0.900	6.81
Median income	5.48e-06	7.71e-06	0.711	-1.74e-06	5.58e-06	-0.312
Unemployment rate	4.91**	1.48	1.30	2.99*	1.27	2.35
Poverty rate	1.32	1.33	0.99	-0.330	1.05	-0.315
Population density	-9.70e-05***	1.69e-05	-5.75	8.86e-05***	1.21e-05	7.30
Foreign born	-5.47**	1.90	-2.88	-1.83	1.32	-1.39
Linguistic isolation	5.73*	2.36	2.43	4.31**	2.07	2.74
Percentage of housing units built before 1940				3.4.37***	0.488	8.95
Single-family detached housing (% of all units)				3.60 ***	0.710	5.07
Median number of rooms				0.478*	0.195	2.45
Median house value				4.55e-06***	9.70e-07	4.70
Housing cost burden				0.0374	0.484	0.0772
Crowded housing				-0.286	2.17	-0.132
Vacancy rate				1.02	1.12	0.913
Impervious surface per capita				-0.450***	0.00914	-49.3
Commute via non-SOV modes				2.36***	0.715	3.30
Number of observations	2,325			2,325		
R-squared	0.383			0.719		
Log likelihood	-7766			-6814		
Akaike information criterion	15564			13679		
Schwarz criterion	15656			13822		
Breusch-Pagan test	255***			1326***		
Likelihood ratio test	648***			251***		

*p<0.05, **p<0.01 ***p<0.001

The results of the demographic model at the block group level confirm the significance of rentership and bachelor's degree attainment in determining local tree cover. Additionally, several new variables, including population density, foreign-born percentage, linguistic isolation, median age, unemployment rate and several racial variables show a significant relationship to tree cover. Also notable are the high results for the Breusch-Pagan test, suggesting heteroskedasticity in the model, and the likelihood ratio test, suggesting that the introduction of the spatial lag term has not fully controlled spatial effects.

As in the community area model, adding the transportation and housing variables dramatically changes the model. Several of the housing variables — housing age, single-family housing, number of rooms, median house value and impervious surfaces per capita — are highly significant, the R-squared is significantly better, and the improvements to the log likelihood, AIC and Schwarz criterion are all relatively apparently. The Breusch-Pagan test and likelihood ratio test, however, remain highly significant, suggesting that the additional variables have not addressed all of the underlying sources of misspecification. Additionally, the fairly large value of the coefficient of rho in both models suggests that unmeasured important variables may continue to exist that are not captured in the model.

Discussion

Neighborhoods with a higher proportion of renters have been previously observed to correlate with lower tree canopy cover (Heynen, Perkins and Roy 2006; Koo et al. 2015). Landry and Chakraborty (2009) theorized that this relationship could be explained by residential mobility: as renters are more mobile than homeowners, they are less likely to reap the benefits of a tree that may take twenty years to grow and therefore have less motivation to plant and steward trees. They also introduce an additional mechanism, suggesting that homeowners may exert their political influence to demand public tree planting because of the positive impact of trees on housing value. Renters, by contrast, would oppose higher property values that are passed on to renters in the form of higher rents and eventual displacement (Wolch, Byrne and Newell 2014).

This explanation initially appears to be supported by this study of Chicago. In the model of tree cover containing only demographic variables, rentership stands out: along with education, it is the only variable with a significant relationship to tree cover ($p<.05$). Moreover, this relationship appears consistent with past literature: rentership has a strong negative correlation with tree canopy. Once variables reflecting the built environment, like auto dependency, age and composition of housing stock and neighborhood-level open space are added to the model, however, the relationship flips: rentership demonstrated a significant and positive correlation with canopy cover, and the overall explanatory power of the model increases. The higher R-squared, higher log likelihood, lower AIC value and lower Schwartz criterion all indicate a much better model fit for the combined model relative to the demographic model.

Several of the variables associated with higher tree canopy cover, including percentage of single-family homes, size of dwelling units and vehicle miles traveled, are typical features of more suburban-style residential areas with fewer multi-unit buildings and less mass transit. This relationship between canopy and these variables associated with low density supports the hypothesis that rentership itself is not the variable that determines areas of low tree cover, but rather the product of renters disproportionately living in dense areas with lots of multi-unit buildings, where land use allows less space for vegetation.

This finding is in line with findings by Pham et al. (2013) that urban form and land-use type is the most important factor in determining urban vegetation, as well as a study performed by Shakeel and Conway (2014) that found property characteristics and resident attitudes were more significant in determining canopy cover than a traditional suite of socioeconomic variables.

However, these results do not fully support the hypothesis advanced by Jesdale, Morello-Frosch and Cushing in their 2013 study of urban New Jersey and Solecki et al. in their 2005 national study, which both identified that renters tend to live in areas of high impervious surfaces where trees cannot grow. Impervious surface area per capita did not appear significant in the community area model, but it was highly significant in the block group model. In the community area model, it was variables that relate specifically to housing, such as the percent of homes built before 1940 and the percentage of single-family homes, that have a significant positive relationship with higher tree cover. These results suggest that impervious surfaces do not provide a full explanation for areas of low tree cover, at least not at the large spatial scale of community areas. Instead, considering the impacts of residential built form and transportation networks is essential to understanding patterns of tree canopy cover in urban environments.

The results at the block group level demonstrate a similar pattern, with some additional caveats. While rentership displays the same flip from a negative to a positive coefficient, it's not significant at all in the combined model; instead, a variety of additional demographic variables are significant in both the demographics-alone and the combined model. Additionally, while the R-squared demonstrates a similar improvement, the other statistical tests indicate that the additional variables do not fully address the heteroskedasticity and spatial effects that may be affecting the model. While the initial results at the community area level present a nice and clear-cut verdict on the importance of built environment variables in the relationship between renters and tree canopy cover, the block group results suggest that there is still more work to do to investigate all the aspects that contribute to this relationship. Some of the difference between the community area and block group results is likely explained by the modifiable areal unit problem (MAUP); correlations that appear pronounced when analyzing using geographically larger units can often vary substantially at smaller scales (Fotheringham & Wong, 1991).

Limitations

One limitation is a lack of historical data, preventing a comprehensive test of the “legacy effect” identified by Troy et al. (2007) and Boone et al. (2010). While some simple historical statistics, such as race, are available from the decennial Census at the community area level, more complex modeling such as the "lifestyle clusters" used by Boone et al., which included historic data on home values, incomes, occupations and education levels, was beyond the scope of this paper.

Additional testing of historical demographic variables would be necessary to fully rule out any "legacy effects" in the results.

Another limitation is the lack of any policy data. Past research by Landry and Pu (2009) demonstrated that municipal ordinances and other legal measures to encourage the growth of tree cover can have a substantial impact; it's possible that programs at the neighborhood or ward level in Chicago could account for some of the apparent differences across the city, but it was not possible to model these programs and their effects in this paper.

The results of the block group analysis show that neither the OLS regression or the SAR_{lag} model adequately capture all the variables influencing the distribution of tree canopy. One possible explanation is that there are simply other influential variables not considered, while another is that a more sophisticated regression could better account for spatial effects. It is possible that both may be necessary to produce a regression that closely matches the actual distribution of tree canopy at the block group level, opening an extensive avenue for further research.

Conclusion

Urban trees are an environmental amenity that deliver several important benefits, including pollution reduction, energy savings, and stormwater and noise control, to nearby residents. Ensuring that this environmental amenity is equitably distributed is an important consideration, particularly given the history of environmental inequities in urban areas like Chicago. The literature on inequities in the current distribution of urban trees is substantial (Talarchek 1990; Pedlowski 2002; Heynen, Perkins and Roy 2006; Landry and Chakraborty 2009; Koo et al. 2019), suggesting that particular socioeconomic groups receive an inequitable share of their distribution of tree cover. Many of these studies identified renters as a group that would naturally be associated with areas of less tree cover; after all, renters tend to move around frequently and do not benefit from rising property values, suggesting they would not benefit from newly planted trees.

None of those studies, however, considered the array of built environment variables included in this study. When those variables are included at both the community area and block group level, the relationship disappears or reverses; the relationship identified in prior studies appears to be the result of renters favoring dense urban areas dominated by multi-unit buildings, rather than something inherent to rentership. These results suggest that future research into environmental inequities ought to consider the role of the built environment in shaping the distribution of environmental amenities, while also highlighting that the distribution can appear different at different scales. Municipalities and other parties interested in redressing environmental inequities should also take care to account for both the built environment and scale in their efforts; a plan that targets specific areas based on a community area-level analysis, for example, might fail to detect important relationships that influence the inequality the plan seeks to address, such as the handful of variables that only appeared to be significant at the block group level. It may also be worth considering the implications of scale in future research; most studies have relied on either Census tracts or block groups, which may miss inequities that would be detected by using spatial units, like community areas in Chicago, that mirror how local residents define their own neighborhoods. Ultimately, these results highlights the need for additional research into the relative influence of the built environment in determining the spatial distribution of environmental amenities, as well as the implications of that distribution for strategies to address distributional inequities.

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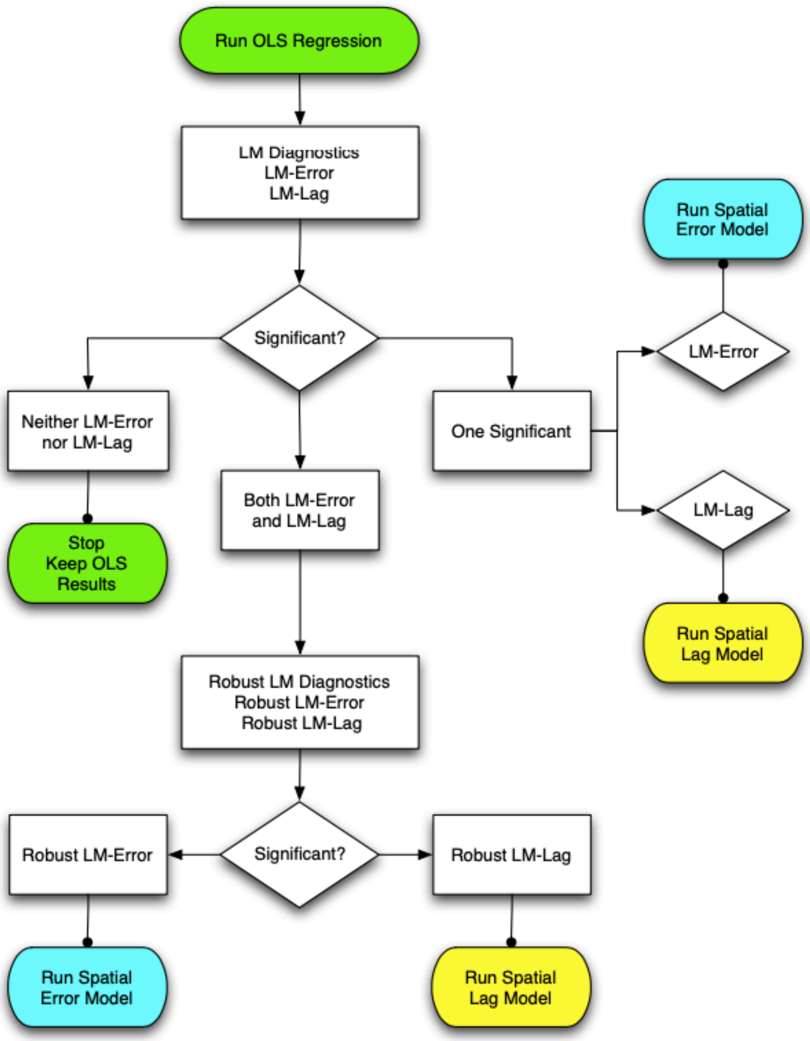
Appendix 1

GeoDa provides five variations of Lagrange multiplier tests to identify whether spatial autocorrelation is present in an OLS regression and, if the answer is yes, whether the problem can be best addressed by adding a spatially lagged dependent variable or a spatial autoregressive error term, following the decision process depicted in Figure 2 (Anselin 2005).

Lagrange multiplier decision-making framework (Anselin 2005)



The simple Lagrange multiplier tests for lag and error test for a spatially lagged dependent variable and a missing error term, respectively, while the robust forms of each test for a missing lagged dependent variable in the possible presence of error dependence, and vice versa, respectively.