The effects of urban density on the provision of multiple health related ecosystem services

Emily Kroft

A thesis in the Department of Biology

Presented in partial fulfilment of the requirements for the
Degree of Master of Science (Biology) at
Concordia University
Montreal, Quebec, Canada

July, 2021

CONCORDIA UNIVERSITY
School of Graduate Studies

This is to certify that the thesis prepared

By: Emily Kroft

Entitled: The effects of urban density on the provision of multiple health related ecosystem services

and submitted in partial fulfillment of the requirements for the degree of

Master of Science (Biology)

complies with the regulations of the University and meets the accepted standards with respect to originality and quality.

Signed by the final examining committee:

________James Grant______________ Chair

___________Jochen Jaeger_____________ Examiner

______Emma Despland___________ Examiner

___________Carly Ziter___________ Thesis Supervisor

Approved by __________Selvadurai Dayanandan_____________
Chair of Department or Graduate Program Director

____________________Pascale Sicotte_____________________________

Dean

Abstract

The effects of urban density on the provision of multiple health related ecosystem services

Emily Kroft

North American cities are currently expanding at an unprecedented rate. Rapid growth in urban development has sparked debate about how to grow cities in a way that minimizes environmental impact and provides ecological benefits to people. Part of this conversation has involved the idea that urban areas should be densely built, so as to minimize their environmental impact, and promote sustainable development goals. However, there is minimal research available on whether there may be a point at which urban areas become too dense for ecosystem service provision (the provision of benefits to humans by nature). Our research explores the relationship between urban density and ecosystem service provision by measuring indicators of health related ecosystem services (temperature regulation, air pollution regulation, and green space accessibility) at 250 study sites in Montreal across a range of building densities (ranging from 0-100%) and population densities (number of households). Using data derived from Landsat-8 and SENTINEL 5P images as well as GIS based analyses, this study addresses the questions: 1) How does building density and its associated landscape features affect multiple health-based ecosystem service indicators? And 2) Is population density related to the provision of ecosystem services at the scale of investigation once building density is accounted for? Results indicate that higher building densities lead to decreases in temperature regulation, but do not impact air quality regulation. High population density sites tend to be more exposed to high temperatures, but are not more exposed to high levels of air pollution. We did not find a statistically significant relationship between household density and distance to public green space, and although marginally significant, the relationship between building density and distance to public green

space was weak. Area of private green space per household is negatively correlated with both building density and population density. These results indicate that while some ecosystem services are unaffected by densification, maintenance of multiple services may require creative solutions at the interface of ecology, planning and design.

Acknowledgements

I would like to thank my committee members, Dr. Jochen Jaeger and Dr. James Grant for guiding me through the process of completing a Master's thesis and answering any questions I had along the way. I would also like to thank Dr. Eric Pedersen for assisting me with my statistical analyses and Dr. Emma Despland for providing me with helpful feedback. I would also like to thank NSERC-CGS and Concordia Faculty of Arts and Sciences for funding my research and the entire Ziter lab for being a support system during the last two years, but especially Kieu-Nhi Vu and Daniel Cretu who assisted me in data collection for an entire summer. My biggest thank you however goes to my incredible supervisor, Dr. Carly Ziter who was there to answer questions, provide support, and help guide me through problem solving the entire time I have been working on this project.

Table of Contents

List of Tables

List of Figures:

Perspective and Positionality

This research pertains to the Island of Montreal, which is located on the unceded Indigenous lands of the Mohawk and Haudenosaunee People. For the purposes of this project, I have conducted my research within the framework of western science and ecosystem services, which focuses on the benefits provided to humans by nature. I acknowledge that this is an anthropocentric framework and that in reality, many non-human species benefit from nature, and that humans can even be considered an integral part of nature. There are many different perspectives one could take when researching urban ecosystems, and the ecosystem services framework is just one of them.

Introduction

By 2050, 68% of the global population is expected to live in cities (United Nations, 2019). In many parts of the world, urban land area is expanding at a faster rate than that at which the urban population is increasing, a phenomenon that is often termed 'urban sprawl' (Nazarnia et al., 2016; Schwick et al., 2012). This type of rapid urban growth can have serious environmental implications, including habitat loss and reduced environmental quality due to land use conversion (Mcdonald et al., 2008; van Vliet, 2019). Therefore, the concept of building compact cities has gained popularity. However, compactness of cities can potentially lead to decreases in the provision of ecosystem services (the benefits provided to humans by nature) due to loss of green space and increased pollution (MEA, 2005). This potential trade-off between compact and low density cities has frequently been examined under the framework of 'land-sharing' vs. 'land-sparing' (Collas et al., 2017). While this framework provides a starting point to understanding potential benefits and drawbacks of densification, it ignores the fact that urban developments can exist along a full range of densities and cannot simply be categorized as 'dense' or 'sprawling'. This gap in the existing scientific literature is the focus of our investigation.

Research on the effects of urbanization on biodiversity demonstrates that many species become threatened when urbanization is intense because the process of urban land expansion degrades their habitat (Mcdonald et al., 2008; Soga et al., 2014). It has been estimated that 16% of all natural habitat loss between 1992-2000 was due to urbanization (McDonald et al., 2019). Species that live in urbanized areas are more likely to become endangered than species living in non-urbanized areas due to a combination of direct and indirect effects (Mcdonald et al., 2008). There is a negative relationship between the number of buildings in an area and the population density of certain insect species (Soga et al., 2014) and popular urban landscaping practises have been found to threaten biodiversity (Aronson et al., 2017). These findings from previous research are related to the close relationship between increasing building densities and decreasing green space in urban areas (Pham et al., 2013). This growing body of work suggests that expanding cities into surrounding wild areas is often detrimental to biodiversity, and should be avoided in favour of compact development (although see Spotswood et al., 2021). The idea of the compact city has also gained popularity for its potential benefits to humans. These benefits could include higher rates of active transportation and easier access to community resources (Adams et al., 2014).

However, before accepting the compact city as a superior design, potential drawbacks of densification should also be examined.

An important factor to consider regarding urban landscape structure is that increased urban density often leads to reduced urban green space, which can have implications for ecosystem service provision (Artmann et al., 2019; Koprowska et al., 2020). Within an urban ecosystem, the provision of ecosystem services is dependent on the presence of green space (Tzoulas et al., 2007). For example, in very dense urban areas, people are more exposed to air pollutants in part due to lack of uptake by vegetation (Nowak., 2014). Even short term exposure to air pollutants has been found to increase daily hospital admissions and mortality rates in the United States (Dominici et al., 2006; Rustgi et al., 2018). Urban areas are also vulnerable to hotter temperatures than rural areas, which has been linked to heat related illness (Jenerette et al., 2016). When densification is combined with reduced urban tree cover this can lead to further detrimental health impacts including greater prevalence of asthma in children, increased heat stress, and increased psychological stress (Wolf et al., 2020). Reduced access and exposure to green space can also lead to negative impacts on mental health; for example, a recent study found that people who were less exposed to green space during childhood are more likely to develop psychiatric diseases (Engemann et al., 2019). More recently this trend has become more widely acknowledged by the public as well; since the beginning of the COVID-19 pandemic, over 80% of Canadians have reported that public green spaces have become more important to their mental health (Park People, 2021).

The more people who are present in an area with clean air and comfortable temperature, the more efficient the provision of ecosystem services is considered to be. Because of this, it is critical to not only measure the landscape's capacity to provide ecosystem services but also the flow of ecosystem services to people (Villamagna et al., 2013). The ultimate goal of quantifying ecosystem services is to consider how biophysical effects of nature, like temperature regulation, benefit humans specifically. If temperature regulation is happening, but there are no humans around to benefit, this cannot be considered ecosystem service provision. From a purely biophysical standpoint, if a tree falls in a forest and nobody is around to hear it, it still makes a sound because it still creates sound waves. From an ecosystem services standpoint, the vibrations created by the falling of the tree are irrelevant unless there is a human nearby to experience them.

To account for this, our research will also account for population density as an important factor in quantifying ecosystem services.

Population density is an important factor in determining ecosystem service provision because it takes a simple biophysical measurement, like amount of green space (which can be considered as supply or capacity) and provides the context for whether this measurement is adequately meeting demand. The ecosystem services framework frequently uses the general idea of supply and demand to conceptualize whether nature's benefits are reaching the people who need them (Opdam & Steingröver, 2018). While these terms often have an economic connotation, it should be noted that in the context of ecosystem services this is not always the case. In this investigation, the idea of ecosystem services being supplied to people will be used only in terms of biophysical measurements, and without any economic assumptions, to address the needs of people in regards to urban development.

The potentially conflicting interests between humans and wildlife when it comes to urban expansion poses a challenge for urban development. Further research is needed at the intersection of urban ecology, geography, and planning to determine whether there are ways in which cities can grow that would promote both biodiversity conservation and human wellbeing. To begin answering this question, some scientists have adapted the land-sharing – land-sparing concept from agriculture (Collas et al., 2017; Phalan, et al., 2011; Phalan, 2018) (Fig. 1).

The 'land-sharing' model describes what we would recognize as a suburban neighbourhood. It involves dispersed housing, interspersed with green spaces across a wide area. The land-sharing model is frequently criticized as promoting urban sprawl, which eliminates natural habitats and puts many species at risk (Tzoulas et al., 2007). For example, the population size of both butterflies and ground beetles has been observed to decrease in land-sharing environments when the level of urbanization was high (Soga et al., 2014). However, a number of recent scientific studies have found that urban areas can support medium to high biodiversity when planned correctly (Geschke et al., 2018; Knapp, 2020; Spotswood et al., 2021).

In contrast, the 'land-sparing' model describes a dense urban area that takes up a small geographic footprint. Under the land-sparing model, biodiversity may benefit due to the presence of intact habitat outside the developed area, but within the densely built city, there is potential for depleted environmental quality due to compact development and pollution (Geschke et al., 2018; Nelson et al., 2009). This suggests a potential trade-off between the best-case scenario for biodiversity conservation and the best-case scenario for ecosystem service provision. To address this, more research is needed on how various patterns of urban development impact nature and people (Knapp, 2020).

An area of research that warrants particular attention includes cases in which past a certain threshold of density or greenness, ecosystem services are not delivered. For example, Ziter et al. (2019) found that when canopy cover is below ~40%, the ecosystem service of temperature regulation was not adequately provided in Madison, WI. Similarly, Cheng et al. (2015) found that the cooling efficiency of parks in China is non-linear, with increases in park size being more important in smaller parks than larger ones. The nonlinearity of these relationships can make it difficult to quantify how much green space is necessary to provide ecosystem services. This highlights a limitation of the land-sharing – land-sparing model, in that it represents a false dichotomy in which a neighbourhood is either dense or sprawling. In real cities, building densities exist along a gradient, which has not been sufficiently considered in urban landscape ecology research, and may interact with thresholds of ecosystem service provision. This study will thus build on the land-sharing – land-sparing framework to consider the full range of building densities.

The goal of this research is to use existing data and remote sensing techniques to determine how multiple ecosystem services vary along a gradient of urban density. We compared measurements of temperature, air pollution and green space accessibility (private and public) across the entire island of Montreal. These three ecosystem service indicators were selected due to their influence on human health, and their expected sensitivity to changes in green space as a city densifies.

High temperature is a known issue in urban areas due to the urban heat island (UHI) effect (Arnfield, 2003). The UHI effect refers to how cities tend to experience hotter temperatures than their rural counterparts (Arnfield, 2003). This effect is mainly caused by the difference in reflectivity between the surfaces that dominate the urban landscape (cement, asphalt etc.) and the surfaces that dominate rural landscapes (grass, soil, etc.) (Crum & Jenerette, 2017). The urban surfaces tend to absorb more solar radiation, re-emitting it later in the form of heat, which causes high surface and air temperatures in the city (Oke, 1982). Natural surfaces tend to reflect more solar radiation allowing for the surrounding areas to be cooler (Bowler et al., 2010), while evapotranspiration by vegetation further cools the surrounding air (Alexander, 2021); thus, vegetation provides important temperature regulation services. While this phenomenon is often studied for the difference in temperature between urban areas and rural areas, it is also an important factor within urban areas (the intra-urban heat island effect). The increase in temperature with increasing pavement causes problems for people living in poorly vegetated parts of cities because it makes them more vulnerable to extreme heat events and related detrimental health effects (Patz et al., 2005).

Poor air quality is a problem that affects urban areas all over the world. Concentrations of air pollutants tend to be higher in urban areas because of the high density of pollutant sources, such as road vehicles or production facilities (Mayer, 1999) combined with a lack of vegetation to filter pollutants from the air (Escobedo et al., 2011). This is a concern because levels of air pollution are positively associated with increased prevalence of cardiovascular disease and death from all causes (Dockery et al., 1993; Dominici et al., 2006; Samet et al., 2000). In contrast, areas that are less dense with higher vegetation cover tend to have much better air quality regulation services (Escobedo & Nowak, 2009). Seeing as it is unlikely that urbanization will stop any time in the near-future, intra-urban patterns of air quality were selected as an important variable in this investigation.

Disparities in access to green space, often measured both as distance to green space and amount of nearby green space, in urban areas is a well-documented trend (Heynaen et al., 2006; Koprowska et al., 2020; Lin et al., 2015). However, the drivers of these disparities can differ depending on the local context and could include wealth, racism, and differences in urban form as well as many others (Schell et al., 2020). Inequities in green space access are concerning because access to urban green space has been demonstrated as an important factor in human physical and mental health (Demoury et al., 2017; Engemann et al., 2019; Frumkin et al., 2017). If certain areas do not contain enough green space to provide health benefits this would be a relevant issue to address, especially during the COVID-19 pandemic when public green spaces are one of the only safe places that people can gather. Differences in urban density may play a role in green space disparities (Dennis et al., 2019), which is what our research hopes to address.

Our research aims to answer the following questions:

1) How do building density and its associated landscape features affect multiple health-based ecosystem service indicators (temperature, air pollution and access to green space)?
2) Is population density related to the provision of ecosystem services at the scale of investigation once building density is accounted for?

We hypothesized that there would be a negative relationship between local building density and total provision of ecosystem services, as a result of a lower amount of green space and vegetation in dense neighbourhoods. We predicted that this relationship would be linear for the provision of green space accessibility, but non-linear for the provision of temperature regulation and air quality maintenance. Our second hypothesis was that population density is not related to air quality or temperature once building density is accounted for but will be negatively related to access to green space because as more people must share the same green space, this translates into less green space per capita.

Methods

Study Area and Site Selection

Montreal is a large Canadian city of approximately two million inhabitants (Statistics Canada, 2016). It is an island in the St. Lawrence River in the province of Quebec (Fig. 2). The climate of Montreal is characterized by humid continental conditions (Government of Canada, 2021).

Two hundred fifty study sites were selected for analysis (Fig. 2). Each study site consists of a 120 m x 120 m cell, located on the Island of Montreal. This size was selected because it captures variability at the local scale (i.e., a few city blocks), which is a scale at which we can reasonably expect to see differences in the provision of, and access to, the selected ecosystem services. It is also a scale at which we see the full range of building densities and may see densification via infill development. All spatial analyses including site selection were done in QGIS 3.8, using the Quebec Albers projection (QGIS Development Team, 2020). We created a 120 m x 120 m grid over the study area, and calculated percent building cover for each grid cell, using building density data from the Microsoft open layer of building footprints (Microsoft, 2019). Study site selection was done so that the final set of sites was evenly distributed across the island of Montreal, and also evenly distributed across the gradient of building densities from 0-100%. To achieve this, we stratified sites by building density in 10% increments (e.g., 0-10% building cover, 10-20% building cover, etc.), and geographically (by borough across the 34 boroughs on the island, also ensuring that sites were not clustered in areas subject to particular policies or governance). The result yielded 25 sites within each building density class, distributed geographically throughout the city. No two sites were adjacent to each other or adjacent to water and no borough contained two of the same density strata. The purpose of ensuring no site was adjacent to water is that water-adjacent areas tend to have cooler temperatures due to the high heat capacity of water and could therefore lead to trends in the data that are not related to the research question at hand.

To determine how many people are benefitting from the ecosystem services being provided at each site, we calculated the number of households present within each study site. We considered number of households as a proxy for population density, calculated using household data from the Montreal open data portal (Ville de Montreal, 2020). Exact numbers of how many individuals live within each site was not available due to privacy concerns.

Ecosystem Service Indicators

Land surface temperature (LST), an indicator of temperature regulation services, was obtained from a Landsat 8 image of the Montreal area with a resolution of 30 m x 30 m (Fig. 3). The image used was taken on July 27, 2019 in the afternoon (Appendix I), and was selected based on time of year and minimal cloud cover. Mid-summer was chosen because this is the time of year when heat stress becomes a health concern, and when ecosystem services provided by vegetation are most pronounced. The image was pre-adjusted for brightness temperature and atmospheric correction. We calculated LST using the thermal band according to the method described by Parastidis et al. (2017). We measured LST as the mean LST value for pixels within each study site.

Concentrations of NO_2 were used as a representative measurement of air quality. We chose this pollutant for a few reasons. First, it is a pollutant that we would expect to be correlated with building density since vegetation can reduce NO_2 through stomatal uptake (and there is likely to be more vegetation in less building dense areas). Secondly, it is frequently used in similar studies, making our results comparable to related studies in this field (Apparicio et al., 2016; Lee, 2019; Restivo et al., 2019). We obtained NO_2 concentrations for the Montreal area from a SENTINEL 5P image of the Montreal area. The image has a resolution of 500 m x 1000 m. We selected an image taken on the same July afternoon as the LST image. This product is pre-adjusted for clouds and provides values of mol/m^2 of NO_2 for each pixel. We measured NO_2 as the mean NO_2 concentration for each study site.

We obtained a map of public green spaces from the Montreal land use map in the CMM data portal (Communauté métropolitaine de Montréal, 2016). According to the World Health Organization (WHO) guidelines, individuals should live within a five-minute walk of the nearest public green space that is larger than 0.5 ha (WHO Regional Office for Europe, 2016). Based on this guideline, we selected all public green spaces that were larger than 0.5 ha to use for this analysis. We verified the accuracy of each green space using Google Street View, and hand corrected for any inaccuracies in green space presence, size, or shape. We also digitized any barriers around the parks (ie: fences) to ensure accuracy in how and where each park could be accessed (Appendix II). Using QGIS, we calculated the walking distance along the road network from each study site centroid to the entrance of the nearest qualifying public green space (Appendix II). While many studies use Euclidean distances, this leads to potential inaccuracies in describing real world conditions, therefore, only road network distances were used in analysis.

In addition to travel distance, we also measured public green space access in terms of area of green space surrounding each study site. We measured this at multiple spatial scales (300, 500, 800, and 1000 m) to correspond to different recommendations regarding green space access or other aspects of urban mobility. Three hundred meters corresponds to the value provided by WHO as the maximum distance one should live from a green space (WHO Regional Office for Europe, 2016). Five hundred meters is the Canadian government recommended maximum distance one should live from the nearest public transit stop (Statistics Canada, 2020). Similarly, 800 m was selected based on other recommendations from the transportation literature (Castel & Farber, 2017). Public transit stops were used as a reference point because if people can reasonably walk a certain distance to a public transit stop, it would be reasonable to expect them to walk a similar distance to a public green space. A final distance of 1000 m was selected based on the green-space walkability literature (Hogendorf et al., 2020). Based on additional WHO recommendations, analyses regarding green-space accessibility were repeated using a size threshold of 2 ha, but results did not significantly change, so we opted for the more inclusive 0.5 ha threshold for all analyses.

To measure access to private residential green space, we clipped the Montreal Open Data Portal's map of land parcels to only include parcels of land that included at least one household. We then subtracted any impervious cover from the remaining layer by using a custom generated impervious features layer (Appendix III). This left us with only green space that was located on privately owned, residential land parcels. The metric of private green space access used in analyses was the surface area of private green space per household for the study site.

In addition to our independent variables (building and population density), we considered a range of covariates related to landscape structure and socio-economic factors anticipated to influence ecosystem service provision. Euclidean distances to the river and to the nearest major road were calculated using the NNjoin plugin in QGIS. A 'major road' is defined as all roads class six or higher. A class six road has two lanes of traffic going in each direction. To measure the percent canopy cover of each study site, we used the map of canopy cover from the Montreal Open Data Portal. We converted this map to a raster image, and then used the ‘zonal statistics’ tool to calculate the percentage of each study site occupied by canopy cover. We created a map of ground-level impervious surfaces by compiling multiple existing datasets on the locations of roads, sidewalks, buildings, and industrial sites, and then adding in missing features through hand digitization based on satellite imagery (Appendix III). This process was important for achieving more accurate results because it ensured that types of impervious cover that are often excluded, such as impervious surfaces under canopy, were included for analyses. For analyses, we included all non-building impervious surfaces in the category of ‘impervious cover’. We calculated median household income based on data from the most recent Canadian census (Statistics Canada, 2016).

Statistical Analysis

To determine the effects of building density and associated landscape structure on LST and access to public green space, we used linear mixed models (using the lme function in nlme in R (Pinheiro et al., 2020)). To model LST, we included canopy cover, log(income), log(distance to river), and impervious cover as co-variates in the model in addition to building density. This analysis was repeated for multiple times of year to ensure the observed relationships were robust to changes in seasonal conditions (Appendix I). To model access to public green space, log(income) was included as a covariate in addition to building density. This was done in all models of public green space (one using distance as a metric and four to represent various buffer sizes using surface area as a metric). For all models, we included borough (the official term for the municipal subdivisions of Montreal) as a random effect to account for potential spatial structuring, or influence of local policies on observed relationships. Residuals were visually inspected for normality and homoscedasticity using diagnostic plots and covariates were checked for collinearity using the VIF function in R. All variables were checked for inclusion by removing variables one at a time and comparing all possible models using likelihood ratio tests and Akaike weight ratios. For both LST and income, all original co-variates (above) were retained in the final models.

To test for the effects of building density and associated landscape structure on NO_2 concentrations and access to private green space, we used general additive models (using the gam function in MGCV in R (Wood, 2011)) to account for non-linearities. When modelling NO_2, we used the Spatial+ approach described by Dupont et al. (2020) to account for spatial patterns beyond those accounted for by model covariates. The Spatial+ approach involves creating spatial models for each individual covariate and then feeding the residuals of these models into the final model in place of raw data. In addition to building density we included log(income), log(distance to a major road), and canopy cover as covariates in the model. When modelling access to private green space, log(median household income) was included as a covariate in addition to building density. As above, borough was included as a random effect. For both NO2 and private green space, model residuals were visually inspected for normality and homoscedasticity using plots produced by the gam.check function in MGCV. Covariates were checked for concurvity using the concurvity function in MGCV. The NO_2 model used 29 basis functions, which was the minimum number of basis functions that would allow for the model residuals to be normally distributed and homoscedastic. Co-variates that were included in preliminary models that were eventually excluded from the final model were the distance to the petrochemical facility in Northeast Montreal, because it was not a significant predictor of NO_2 concentrations in the study area and removing it improved the model according to likelihood ratio tests and Akaike weight ratios. The private green space access model contained nine basis functions and was verified using the same techniques as described for the NO_2 model. Here, all co-variates were retained in the final model.

To determine the effects of population density on ecosystem service indicators, we used general additive models (using the gam function in MGCV in R). We modelled each of LST, NO_2, area of private green space and public green space access as functions of the number of households per site with borough included as a random effect. Once again, the models of public green space access included one model using distance as a metric and four using surface area as a metric. The residuals for each model were visually inspected for normality and homoscedasticity using the gam.check function in MGCV. For each of the LST model and the green space access models, there were nine basis functions. The NO_2 model had 29 basis functions. We determined the number of basis functions for each model by selecting the lowest possible number of basis functions that would allow the model residuals to be normally distributed and homoscedastic. Each model was run once on its own, and then again with building density included as a co-variate to determine whether household density significantly influenced the provision of ecosystem services beyond the influence of building density. Determining whether household density was a strong predictor separate from building density was necessary because household density and building density were not closely correlated at our study sites. These models were validated using the same methods as described above and were compared using likelihood ratio tests as well as AIC comparisons.

As a form of validation, all collected data was initially split into a training data set (90% of data) and a test, or validation, data set (10% of the data) (Appendix IV). To ensure our models were representative and to avoid overfitting, all models were initially run using the training dataset that excluded 10% of the collected data, with the 10% test data set visually inspected for fit to the training model (Fig A4). The training data was also used to predict the values of the test data. The pearson correlation coefficients for predicted vs. observed vales of test data are 0.69 for LST, 0.43 for NO_2, 0.52 for distance to public green space, and 0.82 for amount of private green space. Residuals (observed – predicted values) were also inspected visually to assess model fit (Fig A5). The models presented in the final analysis are complete models using all collected data. For all analyses, the p-value threshold used for statistical significance was alpha = 0.05, with alpha = 0.1 considered as marginally significant.

Results

The effect of building density on ecosystem service provision was different for each of the indicators measured (Fig. 4). Land surface temperature (LST) is strongly, positively associated with building density ($F_{5,200}$=107, p<0.001) with every 15% increment in building density corresponding to ~1°C increase in LST. LST is also associated with canopy cover, impervious cover, income, and distance to the river (Table 1). Canopy cover and log(income) are negatively associated with LST values with every 15% increase in canopy cover leading to ~1°C decrease in LST and every 10% increase in income equating to ~0.1°C decrease in LST($F_{5,200}$=107, p<0.001) (Fig. 5). While building density and canopy cover were somewhat negatively correlated, there were many examples of study sites at equal densities with very different levels of canopy cover (Fig. 5). Distance to the river is positively associated with LST ($F_{5,200}$=107, p=0.03), as is impervious cover ($F_{5,200}$=107, p<0.001).

Study sites with fewer households showed higher variability in LST values (Fig. 6), while at sites with high numbers of households LST values were consistently high, although the relationship between number of households and LST was not significant once building density was accounted for in the model (p=0.11). The majority of households measured were living in areas of high LST (>32°C).

Concentrations of NO_2 were highly influenced by proximity to the Suncor Petrochemical Refinery at all study sites. When modelled as a non-spatial GAM, this effect was the strongest predictor of NO_2 concentrations out of all sampled co-variates (p<0.001). However, as substantial spatial autocorrelation was present in the model as measured by a semiovariogram, the Spatial+ approach was applied, which meant that the distance to the petrochemical facility was accounted for by the use of the spatial model. When modelled using the Spatial+ approach, NO_2 was significantly, negatively associated with distance to the nearest major road (p=0.003) and median household income (p=0.006) (Table 2). Study sites that are further from the nearest major road have lower concentration of NO_2, with every additional increase of 1% distance from the nearest major road equating to ~$3.8x10^{-9}$ mol/m^2 lower NO_2 concentration. Sites with higher income also have lower levels of NO_2 with every 1% increase in income equating to ~$1.2x10^{-8}$ mol/m^2 lower NO_2 concentrations. Neither building density nor canopy cover were significant predictors in this model ($p>0.05$). There was no statistically significant relationship between household density and NO_2 concentrations ($p=0.12$).

We found a marginally significant positive relationship between road network distance to the nearest public green space and building density ($p=0.06$). Twenty-four percent of all sampled residential sites did not meet the WHO guideline that people should live within a five minute walk to the nearest public green space (WHO Regional Office for Europe, 2016) (Fig. 7). The area of public green space surrounding a site was negatively correlated with building density for buffer areas of 300 m ($p<0.001$), 500 m ($p=0.002$), 800 m ($p=0.009$), and 1000 m ($p=0.02$) radii. However, for all four buffer sizes the strength of this trend was very weak (Fig. 8). Although high building density sites had slightly less public green space within a surrounding buffer on average, there were still many examples of high-density sites that contained a high surface area of public green space within a buffer.

Study sites with high numbers of households tended to have less public green space within buffer areas of all sizes measured than sites with fewer households although this trend was not statistically significant at $p<0.05$ or $p<0.1$ for any of the buffer sizes (Appendix II).

We found that area of private residential green space was significantly reduced with increasing building density ($p<0.001$) (Fig. 4). Private green space is only available to some residents. Out of all the households that were counted in this study, only 18% had access to a backyard (Fig. 10). Study sites of low building density had more private residential green space available per household than sites with high building density. We also found a statistically significant non-linear relationship between household density and area of private residential green space per household in which the more households that were contained in a site, the less private green space was available to each household ($p<0.001$) (Fig.6).

Discussion

As urban areas continue to grow around the world, it is increasingly important to consider how cities can be densified safely while maintaining environmental quality. The goal of this study was to determine how the provision of health-related ecosystem services varies along a gradient of building density, and also, to assess how this might influence access to urban ecosystem services. We found that while the indicators of some ecosystem services (i.e., temperature regulation, area of private green space per household) are strongly influenced by building density, the provision of others (distance to public green space, air quality regulation) are only marginally, or not at all affected by building density, providing insight into how urban densification may alter provision of multiple ecosystem services (Fig. 4). This finding was contrary to our initial hypothesis that as building density increased, the provision of all three ecosystem services investigated would decrease. Similarly, some ecosystem services are better provided at low population densities (i.e., private green space area per household), limiting access to these services for urban dwellers living in heavily populated areas of the city, while other services can be provided at high levels across a range of population densities (i.e., air quality regulation).

Temperature

Our findings of higher land surface temperatures in areas of the city with higher building densities and more impervious cover are unsurprising, given that impervious landscape features are known to contribute to the UHI (Arnfield, 2003). Our results suggest that in order to keep LST values below ~35°C (a value drawn by comparison to the results of similar studies (Estoque et al., 2017; Li et al., 2018; Parastatidis et al., 2017)), we would need to keep total impervious cover (including buildings) below ~60%. While the relationship between total impervious cover and increased temperature within cities has been recorded by similar studies (Estoque et al., 2017; Osborne & Alvares-Sanches, 2019; Tran et al., 2017), the effects of building density on LST separate from other impervious surfaces has rarely been researched.

Our finding that not just impervious surfaces in general, but buildings in particular, are positively associated with LST is particularly important in that it speaks to a portion of impervious cover that is much less easily modified. While other impervious cover types (i.e., parking lots, school grounds) can be converted to green areas provided enough community support, or may be greatly reduced in some compact-city scenarios, it is much more difficult to eliminate existing buildings. People need places to live and work indoors, and the reality is that we have not yet found a way to build a home out of non-impervious materials that is suitable to modern living. By addressing buildings as their own urban category that significantly influences temperature, we can better address the core of the land-sharing vs. land-sparing question because we address the main part of the built environment that a land-sparing scenario would require us to densify. From a more practical standpoint, our results also aid in understanding how to plan this fundamental aspect of a city in a way that better accommodates future climatic change. Our results suggest that pursuing a land-sparing city while maintaining temperature regulation would be very difficult in our study system using conventional architecture, but as architecture improves and more climate friendly building strategies are implemented (e.g, green or cool roofs), future studies should continue to assess the influence of locally-appropriate interventions on urban heat.

While buildings and additional impervious cover significantly increase urban temperature, this effect is moderated by canopy cover. At the same building density, sites with higher canopy cover remain cooler, with every 15% increase in canopy cover equating to a ~1°C cooler LST (Table 1). Increasing canopy cover decreases urban heat through several mechanisms. First, trees have higher surface reflectance compared to impervious materials (Wang et al., 2016), reabsorbing and re-emitting less energy as heat than pavement does. Trees also mitigate urban heat through the process of evapotranspiration, converting the sun's energy into latent heat rather than sensible heat, which reduces thermal stress on the surrounding environment (Spronken-Smith & Oke, 1999). Relatively dense canopy cover can also be an important source of shade (Leuzinger et al., 2010). These combined benefits of canopy cover help reduce heat stress in the urban environment. Thus, our results confirm an increasingly well documented relationship between canopy cover and reduced air and land surface temperatures within urban areas (Brown et al., 2015; Zhou et al., 2017; Ziter et al., 2019). For example, Ziter at al. (2019) found that at similar scales as our study, the ecosystem service of air temperature regulation is better provided in areas with over 40% canopy cover. While this non-linear relationship differs from the linear relationship found here, this could be due in part to the fact that they were measuring air temperature rather than land surface temperature. Air temperature and land surface temperature do not necessarily follow a one to one ratio (Mildrexler et al., 2011). Because of this, air temperature and LST may respond slightly differently to changes in their surroundings, although the general patterns are similar.

Our results highlight that in a temperate city such as Montreal, maximizing canopy in areas where conventional buildings are already in place can help offset the increase in LST caused by high building density. This type of change will require initiative on the part of both government stakeholders and private landowners, as privately owned green space makes up a large portion of the total urban green space in many cities (Goddard et al., 2010). If landowners took the initiative to increase the number of trees on their properties while government leaders took the initiative to increase street trees and trees in parks, this combined effect could be very beneficial, not just in terms of UHI reduction but for multiple ecosystem services. A 2018 report on ecosystem services provided by trees in Montreal estimated that the annual monetary benefit of the Montreal urban forest equates to $4,349, 803.89 (Fondation David Suzuki, 2018). Similar results have been calculated for other cities as well. In the Los Angeles million trees project for example, it was estimated that each tree planted would provide up to $56.00USD worth of benefits in ecosystem services per year (McPherson et al., 2011). Benefits of canopy can be further increased in high building areas through replacement of other impervious surfaces with trees; for example, replacing a portion of each parking lot, as was modelled by Onishi et al. (2010) in Japan. Large scale tree planting in the urban core would not, of course, be without challenges. There are real barriers to planting in dense urban areas such as conflicting infrastructure, poor soil quality, and even political pressure that would need to be addressed (McPherson et al., 2011). At very high building densities, it also may be impossible to plant many trees, as was observed in our data from Montreal (Fig. 5). Future research on how to increase the growth and survival of urban trees despite these challenges will be essential for moving forward. A potential positive finding in our data on that front, is that at density levels up to ~30%, we see a significant number of households benefitting from cool temperatures (Fig. 10). This observation indicates that if densification were pursued up to ~30%, we could see an increase in the number of people benefitting from cool temperatures, in part due to canopy cover.

The presence of higher temperatures in low-income areas has been observed in numerous studies (Hoffman et al., 2020; Jenerette et al., 2016; Norton et al., 2015) often as a result of wealthier neighbourhoods having greater access to resources that lead to cooling (ie: parks, trees) (Jenerette et al., 2016; Norton et al., 2015). Here, we find that income remains a significant driver of LST even when accounting for the effects of impervious and canopy cover, perhaps indicating that income is associated with the physical characteristics of an urban area beyond what we were able to account for in our study. This could be due to other physical characteristics of the built environment, such as building heights and biodiversity of urban vegetation. For example, Alexander (2021) found that the height of buildings is positively correlated with LST, so that areas with taller buildings are hotter. It could be that lower income sites were more likely to have taller buildings, however including data on building heights was beyond the scope of this particular study. If this were true it would also explain why the most population dense sites had such consistently high temperatures. Sites with over 100 households tend to be made up of high-rise apartment buildings rather than single family detached housing. There could also be effects related to the composition of vegetation, which was not considered in the present work. Deciduous trees may be better mitigators of heat than coniferous trees due in part to their higher rates of evapotranspiration, for example (Zhao et al., 2020). Of particular relevance to the negative association we find between income and LST, wealthier urban areas often have higher species diversity than poorer areas due to the luxury effect (Leong et al., 2018), which could in turn influence temperature. For example, recent research in green spaces in China showed that higher Shannon-Weiner diversity resulted in greater cooling benefits when total tree cover was held constant (Wang et al., 2021).

Air Quality

In contrast to temperature, our results indicate that building density is not related to NO_2 concentrations at the scale of our investigation (Fig. 4). This provides reason to be optimistic about the implementation of 'land-sparing' or compact city strategies. If densification can be achieved without sacrificing air quality, then it would make sense to do so and spare natural habitat in the process. However, this result speaks specifically to the densification of *buildings*, rather than increased cover of other built features, including roads.

Unlike buildings, major roads were important predictors of NO_2 concentrations in our study, consistent with other work in this field (Apparicio et al., 2016; Lee, 2019; Yli-Pelkonen et al., 2017). Cars, trucks and other road vehicles emit NO_2 as they travel, exposing the areas surrounding them to the effects of this pollutant (Apparicio et al., 2016). This result aligns with previous studies of Montreal, with researchers finding that sites near major roads displayed higher levels of NO_2 than other urban 'background' sites (Smargiassi et al. 2005; Crouse et al. 2009). Major roads can of course also be used by public transportation or other alternative transportation types, but it is the presence of cars and similar vehicles that make up the majority of traffic congestion on these roads (Duranton & Turner, 2011).

The sensitivity of air quality to particular built features, rather than building density indicates that in the land-sparing vs. land-sharing debate, it is not sufficient to simply discuss cities in terms of 'dense' or 'sprawling'. Rather, we must tease out which parts of a city we could safely densify in their current form, and which parts of the urban design might need to be reimagined to reduce issues like automobile dependency. Dominant forms of transportation may also change in future cities (e.g., electric vehicles, increased public transit), and future studies should take into account how changes in urban transportation will intersect with the ecology of our cities.

The link between poverty and higher exposure to air pollutants is also well noted throughout the scientific literature (Carrier et al., 2014; Crouse et al., 2009; Pinault et al., 2016). A house located beside a busy road may be considered lower in value because of pollutants, noise, lack of privacy, or a general distaste for living along a busy road (Boehmer et al., 2013). This leads to lower income families moving into these homes (Boehmer et al., 2013). Lower income areas also tend to have less vegetation cover (Schell et al., 2020). Although in our analyses we did not find vegetation to be a statistically significant factor in our model at the scale of investigation, previous work at different spatial scales has found correlations between vegetative cover and lower pollutant concentrations (Nowak et al., 2014; Selmi et al., 2016; Yang et al., 2015).

Here, we see limited evidence of air pollution mitigation by vegetation. Results of other studies regarding the provision of air pollution mitigation in urban landscapes are highly variable depending on study methods and scale (Yli-Pelkonen et al., 2017). Studies that address whether increased tree cover reduces air pollution have found that the presence of trees can result in decreases (Nowak et al., 2018), increases (Fantozzi et al., 2015), or no change in levels of NO_2 (Yli-Pelkonen et al., 2017). While vegetation can remove NO_2 from the air through stomatic uptake, whether these effects are measurable often depends on the scale of observation. On a broad spatial scale, trees reduce air pollutants present in the overall landscape (Nowak et al., 2018). However, tall structures, like trees, can also block pollutants from dispersing, increasing pollution at the very local scale (i.e., street canyon) (Gromke & Ruck, 2007). At medium scales, both of these effects might be occurring, which could lead to inconclusive results. This may have contributed to our own inconclusive findings when investigating a medium spatial scale. The grain size at which sampling occurred could also have influenced our results because variation within a pixel is not detected. It is quite possible that if we had used a satellite image with smaller pixel sizes that our results would have differed. Unfortunately, we were not able to obtain any smaller pixel size NO_2 satellite images, but it could be interesting in future research to conduct a pixel size comparison or to install a network of sensors to compare the measured air quality values to the values detected by satellite imagery.

Proximity to intensive manufacturing facilities in Eastern Montreal was an important predictor of NO_2 concentrations across the island. A 2018 report on disparities in the city of Montreal found that people living in certain parts of Eastern Montreal have up to a ten year shorter life expectancy than those living in the rest of the city due in part to having less access to a clean, green environment (Westgate, 2018). Similarly, individuals living in these parts of the city are more likely to experience obesity, psychological distress and attempted suicide (Westgate, 2018). While this effect is not caused solely by any particular factor, the combination of air pollution, lack of high-quality green space, and lower socio-economic status contributes to negative outcomes. The negative effects of having a nearby factory on air quality are not specific to Montreal but have been observed by studies in a variety of cities around the world (Cansaran-Duman et al., 2011; Kushkbaghi et al., 2017). These cumulative effects must be accounted for when decisions are made about infrastructure in affected neighbourhoods, so as not to compound existing environmental injustices.

It is difficult to determine the magnitude of the biological implications indicated by the differences in NO_2 concentration measured by our own study due to the units of measurement involved, which resulting from Sentinel imagery, do not easily convert into the units commonly used in the health literature (i.e., ppm, ppb, etc.). However, the general patterns observed were very close to those observed by other air quality studies in Montreal (Crouse et al., 2009; Westgate, 2018) that found biologically relevant results. This suggests that our own observed patterns would also hold biological significance.

Green Space Access

We did not find a statistically significant relationship between household density and distance to public green space, and although marginally significant, the relationship between building density and distance to public green space was weak (Fig. 4). When using surface area of green space within a buffer as a metric, we found no relationship with housing density and a weakly negative relationship with building density. This indicates that in Montreal, building density does not greatly impact an individual's distance from the nearest green space, but may impact the sizes of those green spaces available to them. However, the difference between surface area of green space between high and low building density sites was small, and we found many examples of building dense sites with high surface area of public green space (Fig. 8). This general pattern held true when looking at green space availability between boroughs as well. There was relatively consistent public green space access in all boroughs apart from one that is located on the far periphery of the island (Baie-D'Urfé). This demonstrates that it is possible to have dense development with access to ample public green space, and that such areas already exist. This result is contrary to similar research that has been conducted in other cities (Koprowska et al., 2020). The fact that Montreal only sees minimal differences in public green space access between low and high density areas implies that in cities where dense areas do have lower access to public green space, this is due to inequitable urban planning rather than an innate feature of densely built environments. Montreal's relatively equal access to public green space across building densities provides reason to be optimistic. However, when other real-world factors related to green space are considered, the lack of a trend here points to a different form of inequitable distribution. A major factor to consider here is that some households have access to private green space while others do not.

Our results revealed that building dense and densely populated study sites have significantly less available private green space per household than their sprawling counterparts. This was also true between different boroughs, with highly populated and building dense boroughs enjoying less private residential green space per household on average than less dense boroughs. This situation is likely unavoidable to a certain extent. As more people share the same amount of land, each person will inevitably get access to less of it. Therefore, this specific finding does not necessarily point to a problem in private green space distribution that needs to be solved so much as it serves as a reminder that equality does not equal equity when it comes to ecosystem service provision. Given that some neighbourhoods have significantly less private green space available to them, it follows that those same neighbourhoods should be given higher access to public green space in order to level the playing field. Our results reveal that in Montreal, this is not occurring. Part of this could be related to the way in which green spaces are divided up into private and public in the North American context. We should not rule out the possibility that in the future, new ways of sharing green space could come into vogue. It should also be noted that due to the way this metric was calculated, it cannot be guaranteed that every household that *could* have access to private green space truly does in practise. For example, in a case where a duplex existed on a lot with a private yard, we considered both households in the duplex to have some access to private green space. However, it is perfectly conceivable that in practise, only one of these households would be allowed to use the yard. Therefore, our results regarding private green space access should be interpreted as potential access to green space rather than real-life access.

Another important factor to consider is the number of people who must share the same public green space. Although not statistically significant, we found that study sites with a greater number of households tend to have less public green space within a buffer of 300-1000 m radius (Fig 8). This creates a disparity in green space access in which those living in denser areas enjoy less green space per person than those living in less dense areas in addition to having less available private green space. Given that our study sites were selected primarily along a gradient of building density, future studies should be explicitly designed to further examine the relationship between green space access and population density to better understand the nuances of this relationship. The issue of disparities in access to green space has been acknowledged by the scientific literature numerous times but has yet to be resolved in most cities (Haaland & van den Bosch, 2015).

Disparities in access to green space can have significant consequences, given the importance of green space for physical and mental health (Demoury et al., 2017; Dennis et al., 2020; Sarkar et al., 2018). People who enjoy high access to green space are less likely to develop several health conditions including certain cancers and psychiatric illnesses (Demoury et al., 2017; Engemann et al., 2019; Frumkin et al., 2017). Among socio-economically deprived neighbourhoods,

neighbourhoods with high levels of public green space have disproportionately good health (Dennis et al., 2020). Especially over the course of the COVID-19 pandemic, the benefits of green space are being recognized by a broader audience (Park People, 2020). While it is unsafe to gather indoors, parks have become increasingly important for maintaining mental health (Park People, 2020). Because of this, Canadians are now valuing green spaces more highly than in previous years, which is manifesting in greater demand for urban green spaces (Park People, 2020). This could become an issue in a city like Montreal where we found that nearly one quarter of all sampled residential sites were not meeting the WHO recommendations for access to public green space and nearly one in ten did not meet the WHO recommendations or have access to private green space. This reality increases the urgency of the call from previous studies for a greater total number of green spaces to be implemented in cities, which would increase overall mental and physical health (Barbosa et al., 2007).

Our finding that private green space access is decreased in building-dense areas does not indicate that overall green space access as an ecosystem service would be lost in a land-sparing type scenario because public green spaces can be implemented in even the densest urban areas. Rather, these findings show that there is a trade-off between the capacity of a city to provide private vs. public green space. If public green space were prioritized during the process of densification, this would provide greater access to ecosystem services to more people.

Implications

The findings of our research support the idea that a land-sparing type city can be maintained without sacrificing access to certain ecosystem services. However, not every ecosystem service can be maintained in a densified city that is mainly impervious. It may be very feasible to densify a conventional North American city while maintaining adequate air quality and access to public green spaces, but much more difficult to densify while maintaining cool daytime temperatures or access to private green spaces. An encouraging example from our study of an area that did well in maintaining decent temperature despite high building density was study site 12, located in Côte-des-Neiges neighbourhood (Fig. 9). Despite having over 50% building density, this site maintained an average LST below 35°C. This may have been in part due to the relatively high canopy cover at this site compared to other sites of similar building density. This example illustrates the importance of considering various types of densification. If densification is done in such a way that adequate greenery is maintained, this could be much more beneficial for preserving ecosystem services than if densification is done without such a consideration for the overall landscape structure.

In regard to the concern that there may be a trade-off between conserving biodiversity and maintaining ecosystem service provision in urban environments (the land-sharing vs. land-sparing debate), our results do not provide a perfectly straight forward answer. In order to maintain a variety of ecosystem services under scenarios of urban densification, careful consideration should be given to where, and how, green infrastructure can be integrated (ie: green roofs, green facades) (Ionescu et al., 2015; Jiachuan Yang & Bou-zeid, 2019). Therefore, our findings do not suggest that land-sparing type cities are unfeasible, or undesirable. Rather, our results highlight potential barriers that would have to be addressed in order to densify a city safely. Additionally, while we considered the impacts of densification on ecosystem service provision within the city itself, further studies should also consider the additional ecosystem services that may be afforded to city dwellers by maintaining high quality habitat outside of the city. Through supporting high biodiversity outside of a densified city, land-sparing approaches may provide additional ecosystem services not considered here (such as recreational services, or improvement of regulating services at a broader scale) (Lin & Fuller, 2013). It should also always be taken into consideration that local climates and biomes differ by location, therefore these differences should always be considered when implementing any sort of ecosystem service based solution (Lin & Fuller, 2013). For example, in our study, we emphasize the importance of maintaining high canopy cover, but in Nevada, evidence has been found that urban areas with high canopy cover tend to use water resources at a much higher rate than the surrounding natural biome (Imhoff et al., 2010)

Practically, our findings support two major overlapping themes: the importance of minimizing impervious cover while maximizing vegetation cover, and the need to decrease automobile dependency. The issue of minimizing impervious cover is particularly relevant to temperature regulation given the strong positive relationship between impervious features and LST. In our study sites, the most prevalent type of impervious feature other than buildings was designated parking areas. This can serve as a reasonable starting point for greening. In one study, it was found that by replacing a single parking lot with grass, summer daytime land surface temperatures could be reduced by up to 3°C in the surrounding area and up to 8°C in the immediate area (Onishi et al., 2010). When trees were added, these differences rose to 4°C for the surrounding area and 9°C for the immediate area (Onishi et al., 2010). Replacing some parking lots with green space in Montreal's high-density, high-heat neighbourhoods could reduce heat stress in the parts of the city that need it the most. Greening parking lots could also increase access to green space for local residents.

The second major theme, decreasing automobile dependency, is particularly important for air quality because we found that proximity to major roadways was a significant predictor of NO_2 concentrations. In some ways, densification is a stepping-stone to reducing automobile dependency. In a higher density city where many daily activities can be carried out by foot or by bicycle, cars become less necessary (Greene & Wegener, 1997). If cars became less necessary, that would provide an opportunity to reduce impervious infrastructure such as large roads and parking lots, which would be helpful in reducing high daytime temperatures as well. This is a strategy that could be combined with theme one, minimizing impervious cover by simultaneously decreasing car-centric infrastructure while increasing publicly accessible green space.

Making sustainable modes of transportation more attractive would also be essential if automobile dependency were to truly decrease. For example, road areas could be redesigned to assume that pedestrians and cyclists are the main users and therefore the priority (Banister, 2008). An efficient, accessible public transit system would of course be vital to accommodate individuals who cannot use active transportation (ie: disabled, elderly, pushing a stroller). This could involve strategies like increasing the number of bus stops and giving busses priority over cars as was done in Copenhagen (Ogryzek et al., 2020). Making driving less convenient than other modes of transportation either because of the cost or efficiency has been an effective strategy to reduce automobile use in multiple European cities including Zurich and London (Freudendal-Pedersen et al., 2020). These types of strategies could be adapted to Canadian cities like Montreal to reduce automobile dependency here as well.

Conclusions

The results of our research indicate that whether or not cities can be densified without sacrificing access to ecosystem services depends on the ecosystem service in question. For air quality, it seems we can densify considerably without sacrificing access to good air quality as long as we limit the number of major pollutant sources (e.g., factories, freeways). For temperature, however, increasing building density does result in higher daytime temperatures, which would put human health at risk. For the ecosystem service of green space access, there is a trade-off between the ability of a densified city to provide public vs. private green space, therefore public green space should be prioritized to maximize the provision of ecosystem services to people. Interdisciplinary collaboration among architects, designers, ecologists, and engineers will be critical to design dense cities that maintain provision of, and access to, multiple ecosystem services.

Despite differences in the shape and strength of the relationships between different ecosystem service indicators and building density, many of the potential solutions are shared among ecosystem services. For example, access to green space could be improved by creating more public green spaces. The vegetation in these new green spaces would in turn help mitigate temperature. It is therefore possible to achieve multiple benefits using a 'kill two birds with one stone' type of approach. Overall, this research provides reason to be cautiously optimistic that a land-sparing model of the city could be pursued so long as creative solutions were implemented to maintain ecosystem services like temperature regulation that are most strongly affected by densification.

Works Cited

Adams, M. A., Frank, L. D., Schipperijn, J., G., S., Chapman, J., Christiansen, L. B., … Sallis, J. F. (2014). International variation in neighborhood walkability, transit, and recreation environments using geographic information systems: the IPEN adult study Adams M.A. *International Journal of Health Geographics*, *13*, 43. Retrieved from http://0-ovidsp.ovid.com.lib.exeter.ac.uk/ovidweb.cgi?T=JS&PAGE=reference&D=emed12&NEWS=N&AN=25343966

Alexander, C. (2021). Influence of the proportion, height and proximity of vegetation and buildings on urban land surface temperature. *International Journal of Applied Earth Observation and Geoinformation*, *95*, 102265. https://doi.org/10.1016/j.jag.2020.102265

Apparicio, P., Carrier, M., Gelb, J., Séguin, A., & Kingham, S. (2016). Cyclists ' exposure to air pollution and road traffic noise in central city neighbourhoods of Montreal. *Journal of Transport Geography*, *57*, 63–69. https://doi.org/10.1016/j.jtrangeo.2016.09.014

Arnfield, A. J. (2003). Two decades of urban climate research: A review of turbulence, exchanges of energy and water, and the urban heat island. *International Journal of Climatology*, *23*(1), 1–26. https://doi.org/10.1002/joc.859

Aronson, M. F. J., Lepczyk, C. A., Evans, K. L., Goddard, M. A., Lerman, S. B., MacIvor, J. S., … Vargo, T. (2017). Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment*, *15*(4), 189–196. https://doi.org/10.1002/fee.1480

Artmann, M., Kohler, M., Meinel, G., Gan, J., & Ioja, I. C. (2019). How smart growth and green infrastructure can mutually support each other — A conceptual framework for compact and green cities. *Ecological Indicators*, *96*(July 2017), 10–22. https://doi.org/10.1016/j.ecolind.2017.07.001

Banister, D. (2008). The sustainable mobility paradigm. *Transport Policy*, *15*(2), 73–80. https://doi.org/10.1016/j.tranpol.2007.10.005

Barbosa, O., Tratalos, J. A., Armsworth, P. R., Davies, R. G., Fuller, R. A., Johnson, P., & Gaston, K. J. (2007). Who benefits from access to green space? A case study from Sheffield, UK. *Landscape and Urban Planning*, *83*(2–3), 187–195. https://doi.org/10.1016/j.landurbplan.2007.04.004

Boehmer, T., Foster, S., Henry, J., Woghiren-Akinnifesi, E., & Yip, F. (2013). Residential proximity to major highways - United States, 2010. *Morbidity and Mortality Weekly Report. Surveillance Summaries (Washington, D.C. : 2002)*, *62 Suppl 3*(3), 46–50.

Bowler, D. E., Buyung-Ali, L., Knight, T. M., & Pullin, A. S. (2010). Urban greening to cool towns and cities: A systematic review of the empirical evidence. *Landscape and Urban Planning*, *97*(3), 147–155. https://doi.org/10.1016/j.landurbplan.2010.05.006

Brown, R. D., Vanos, J., Kenny, N., & Lenzholzer, S. (2015). Designing urban parks that ameliorate the effects of climate change. *Landscape and Urban Planning*, *138*, 118–131.

https://doi.org/10.1016/j.landurbplan.2015.02.006

Cansaran-Duman, D., Atakol, O., & Aras, S. (2011). Assessment of air pollution genotoxicity by RAPD in Evernia prunastri L. Ach. from around iron-steel factory in Karabük, Turkey. *Journal of Environmental Sciences*, *23*(7), 1171–1178. https://doi.org/10.1016/S1001-0742(10)60505-0

Carrier, M., Apparicio, P., Séguin, A. M., & Crouse, D. (2014). The application of three methods to measure the statistical association between different social groups and the concentration of air pollutants in Montreal: A case of environmental equity. *Transportation Research Part D: Transport and Environment*, *30*, 38–52. https://doi.org/10.1016/j.trd.2014.05.001

Castel, E., & Farber, S. (2017). *Benchmarking the Health and Public Transit Connection in the GTHA : An Analysis of Survey Microdata*. *Metrolinx*. Toronto. Retrieved from https://tspace.library.utoronto.ca/bitstream/1807/81447/1/Castel and Farber - Health and Transit Report - 20170427.pdf

Cheng, X., Wei, B., Chen, G., Li, J., & Song, C. (2015). Influence of Park Size and Its Surrounding Urban Landscape Patterns on the Park Cooling Effect. *Journal of Urban Planning and Development*, *141*(3), 1–10. https://doi.org/10.1061/(asce)up.1943-5444.0000256

Collas, L., Green, R. E., Ross, A., Wastell, J. H., & Balmford, A. (2017). Urban development, land sharing and land sparing: the importance of considering restoration. *Journal of Applied Ecology*, *54*(6), 1865–1873. https://doi.org/10.1111/1365-2664.12908

Crouse, D. L., Ross, N. A., & Goldberg, M. S. (2009). Social Science & Medicine Double burden of deprivation and high concentrations of ambient air pollution at the neighbourhood scale in Montreal , Canada. *Social Science & Medicine*, *69*(6), 971–981. https://doi.org/10.1016/j.socscimed.2009.07.010

Crum, S. M., & Jenerette, D. (2017). Microclimate variation among urban land covers: The importance of vertical and horizontal structure in air and land surface temperature relationships. *Journal of Applied Meteorology and Climatology*, *56*(9), 2531–2543. https://doi.org/10.1175/JAMC-D-17-0054.1

Demoury, C., Thierry, B., Richard, H., Sigler, B., Kestens, Y., & Parent, M. E. (2017). Residential greenness and risk of prostate cancer: A case-control study in Montreal, Canada. *Environment International*, *98*, 129–136. https://doi.org/10.1016/j.envint.2016.10.024

Dennis, M., Cook, P. A., James, P., Wheater, C. P., & Lindley, S. J. (2020). Relationships between health outcomes in older populations and urban green infrastructure size, quality and proximity. *BMC Public Health*, *20*(1), 1–15. https://doi.org/10.1186/s12889-020-08762-x

Dennis, M., Scaletta, K. L., & James, P. (2019). Evaluating urban environmental and ecological landscape characteristics as a function of land-sharing-sparing, urbanity and scale. *PLOS ONE*, *14*(7), e0215796. https://doi.org/10.1371/journal.pone.0215796

Dockery, D., Pope, A., Xu, X., Spengler, J., Ware, J., Fay, M., … Speizer, F. (1993). An association between air pollution and mortality in six U.S. cities. *The New England Journal* *of Medicine*, *29*(328 (17)), 1230–1235.

Dominici, F., Peng, R. D., Bell, M. L., Pham, L., McDermott, A., Zeger, S. L., & Samet, J. M. (2006). Fine particulate air pollution and hospital admission for cardiovascular and respiratory diseases. *Journal of the American Medical Association*, *295*(10), 1127–1134. https://doi.org/10.1001/jama.295.10.1127

Dupont, E., Wood, S. N., & Augustin, N. (2020). Spatial+: a novel approach to spatial confounding, (2006), 1–35. Retrieved from http://arxiv.org/abs/2009.09420

Duranton, G., & Turner, M. A. (2011). The fundamental law of road congestion: Evidence from US cities. *American Economic Review*, *101*(6), 2616–2652. https://doi.org/10.1257/aer.101.6.2616

Engemann, K., Pedersen, C. B., Arge, L., Tsirogiannis, C., Mortensen, P. B., & Svenning, J. C. (2019). Residential green space in childhood is associated with lower risk of psychiatric disorders from adolescence into adulthood. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(11), 5188–5193. https://doi.org/10.1073/pnas.1807504116

Escobedo, F. J., Kroeger, T., & Wagner, J. E. (2011). Urban forests and pollution mitigation: Analyzing ecosystem services and disservices. *Environmental Pollution*, *159*(8–9), 2078–2087. https://doi.org/10.1016/j.envpol.2011.01.010

Escobedo, F. J., & Nowak, D. J. (2009). Spatial heterogeneity and air pollution removal by an urban forest. *Landscape and Urban Planning*, *90*(3–4), 102–110. https://doi.org/10.1016/j.landurbplan.2008.10.021

Estoque, R. C., Murayama, Y., & Myint, S. W. (2017). Effects of landscape composition and pattern on land surface temperature: An urban heat island study in the megacities of Southeast Asia. *Science of the Total Environment*, *577*, 349–359. https://doi.org/10.1016/j.scitotenv.2016.10.195

Fantozzi, F., Monaci, F., Blanusa, T., & Bargagli, R. (2015). Spatio-temporal variations of ozone and nitrogen dioxide concentrations under urban trees and in a nearby open area. *Urban Climate*, *12*(2), 119–127. https://doi.org/10.1016/j.uclim.2015.02.001

Fondation David Suzuki. (2018). *Le rôle des infrastructures naturelles dans la prévention des inondations dans la communauté métropolitaine de Montréal*.

Freudendal-Pedersen, M., Hartmann-Petersen, K., Friis, F., Lindberg, M. R., & Grindsted, T. S. (2020). Sustainable mobility in the mobile risk society-designing innovative mobility solutions in copenhagen. *Sustainability (Switzerland)*, *12*(17). https://doi.org/10.3390/su12177218

Frumkin, H., Bratman, G. N., Breslow, S. J., Cochran, B., Kahn, P. H., Lawler, J. J., … Wood, S. A. (2017). Nature contact and human health: A research agenda. *Environmental Health Perspectives*, *125*(7), 1–18. https://doi.org/10.1289/EHP1663

Geschke, A., James, S., Bennett, A. F., & Nimmo, D. G. (2018). Compact cities or sprawling suburbs? Optimising the distribution of people in cities to maximise species diversity.

Journal of Applied Ecology, *55*(5), 2320–2331. https://doi.org/10.1111/1365-2664.13183

Goddard, M. A., Dougill, A. J., & Benton, T. G. (2010). Scaling up from gardens: biodiversity conservation in urban environments. *Trends in Ecology and Evolution*, *25*(2), 90–98. https://doi.org/10.1016/j.tree.2009.07.016

Government of Canada. (2021). Historical Data. Retrieved June 17, 2021, from https://climate.weather.gc.ca/historical_data/search_historic_data_e.html

Greene, D., & Wegener, M. (1997). Sustainable Transport. *Journal of Transport Geography*, *5*(3), 177–190. https://doi.org/10.1016/C2013-0-17820-8

Gromke, C., & Ruck, B. (2007). Influence of trees on the dispersion of pollutants in an urban street canyon-Experimental investigation of the flow and concentration field. *Atmospheric Environment*, *41*(16), 3287–3302. https://doi.org/10.1016/j.atmosenv.2006.12.043

Haaland, C., & van den Bosch, C. K. (2015). Challenges and strategies for urban green-space planning in cities undergoing densification: A review. *Urban Forestry and Urban Greening*, *14*(4), 760–771. https://doi.org/10.1016/j.ufug.2015.07.009

Heynaen, N., Perkins, H. A., & Roy, P. (2006). The Political Ecology of Uneven Urban Green Space: The Impact of Political Economy on Race and Ethnicity in Producing Environmental Inequality in Milwaukee. *Urban Affairs Review*, *42*(1), 3–25.

Hoffman, J., Shandas, V., & Pendleton, N. (2020). The Effects of Historical Housing Policies on Resident Exposure to Intra-Urban Heat: A Study of 108 US Urban Areas. *MDPI*, *8*(12), 1–15. https://doi.org/10.3390/cli8010012

Hogendorf, M., Oude Groeniger, J., Noordzij, J. M., Beenackers, M. A., & van Lenthe, F. J. (2020). Longitudinal effects of urban green space on walking and cycling: A fixed effects analysis. *Health and Place*, *61*(December 2019). https://doi.org/10.1016/j.healthplace.2019.102264

Imhoff, M. L., Zhang, P., Wolfe, R. E., & Bounoua, L. (2010). Remote sensing of the urban heat island effect across biomes in the continental USA. *Remote Sensing of Environment*, *114*(3), 504–513. https://doi.org/10.1016/j.rse.2009.10.008

Ionescu, C., Baracu, T., Vlad, G. E., Necula, H., & Badea, A. (2015). The historical evolution of the energy efficient buildings. *Renewable and Sustainable Energy Reviews*, *49*, 243–253. https://doi.org/10.1016/j.rser.2015.04.062

Jenerette, G. D., Harlan, S. L., Buyantuev, A., Stefanov, W. L., Declet-Barreto, J., Ruddell, B. L., … Li, X. (2016). Micro-scale urban surface temperatures are related to land-cover features and residential heat related health impacts in Phoenix, AZ USA. *Landscape Ecology*, *31*(4), 745–760. https://doi.org/10.1007/s10980-015-0284-3

Knapp, S. (2020). A Research Agenda for Urban Biodiversity in the Global Extinction Crisis. *BioScience*, *XX*(X), 1–12. https://doi.org/10.1093/biosci/biaa141

Koprowska, K., Łaszkiewicz, E., & Kronenberg, J. (2020). Is urban sprawl linked to green space availability? *Ecological Indicators*, *108*(July 2018), 105723. https://doi.org/10.1016/j.ecolind.2019.105723

Kushkbaghi, S., Ehrampoush, M. H., & Mirhosseinidehabadi, S. A. (2017). Assessment of role of concrete factories in particulate matter emissions, 2015-2016, using the AQI index and zoning by GIS software (Case study: Nasr Kashan Concrete Factory). *Environmental Health Engineering And Management Journal*, *4*(3), 149–155. https://doi.org/10.15171/EHEM.2017.21

Lee, C. (2019). Impacts of urban form on air quality: Emissions on the road and concentrations in the US metropolitan areas. *Journal of Environmental Management*, *246*(February), 192–202. https://doi.org/10.1016/j.jenvman.2019.05.146

Leong, M., Dunn, R. R., & Trautwein, M. D. (2018). Biodiversity and socioeconomics in the city: a review of the luxury effect. *Biology Letters*, *14*(5). https://doi.org/10.1098/rsbl.2018.0082

Leuzinger, S., Vogt, R., & Körner, C. (2010). Tree surface temperature in an urban environment. *Agricultural and Forest Meteorology*, *150*(1), 56–62. https://doi.org/10.1016/j.agrformet.2009.08.006

Li, H., Zhou, Y., Li, X., Meng, L., Wang, X., Wu, S., & Sodoudi, S. (2018). A new method to quantify surface urban heat island intensity. *Science of the Total Environment*, *624*, 262–272. https://doi.org/10.1016/j.scitotenv.2017.11.360

Lin, B. B., & Fuller, R. A. (2013). FORUM: Sharing or sparing? How should we grow the world's cities? *Journal of Applied Ecology*, *50*(5), 1161–1168. https://doi.org/10.1111/1365-2664.12118

Lin, B., Meyers, J., & Barnett, G. (2015). Understanding the potential loss and inequities of green space distribution with urban densification. *Urban Forestry and Urban Greening*, *14*(4), 952–958. https://doi.org/10.1016/j.ufug.2015.09.003

Mayer, H. (1999). Air pollution in cities. *Atmospheric Environment*, *33*(24), 4029–4037.

Mcdonald, R. I., Kareiva, P., & Forman, R. T. T. (2008). The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biological Conservation*, *141*(6), 1695–1703. https://doi.org/10.1016/J.BIOCON.2008.04.025

McDonald, R. I., Mansur, A. V., Ascensão, F., Colbert, M., Crossman, K., Elmqvist, T., … Ziter, C. (2019). Research gaps in knowledge of the impact of urban growth on biodiversity. *Nature Sustainability*. https://doi.org/10.1038/s41893-019-0436-6

McPherson, E. G., Simpson, J. R., Xiao, Q., & Wu, C. (2011). Million trees Los Angeles canopy cover and benefit assessment. *Landscape and Urban Planning*, *99*(1), 40–50. https://doi.org/10.1016/j.landurbplan.2010.08.011

MEA. (2005). Ecosystems and Their Services. In *Millenium Ecosystem Assessment* (pp. 49–69).

Microsoft. (2019). CanadianBuildingFootprints. Retrieved from https://github.com/microsoft/CanadianBuildingFootprints

Mildrexler, D. J., Zhao, M., & Running, S. W. (2011). A global comparison between station air temperatures and MODIS land surface temperatures reveals the cooling role of forests. *Journal of Geophysical Research: Biogeosciences*, *116*(3), 1–15.

https://doi.org/10.1029/2010JG001486

Nazarnia, N., Schwick, C., & Jaeger, J. A. G. (2016). Accelerated urban sprawl in Montreal, Quebec City, and Zurich: Investigating the differences using time series 1951-2011. *Ecological Indicators*, *60*, 1229–1251. https://doi.org/10.1016/j.ecolind.2015.09.020

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D. R., … Shaw, M. R. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment*, *7*(1), 4–11. https://doi.org/10.1890/080023

Norton, B. A., Coutts, A. M., Livesley, S. J., Harris, R. J., Hunter, A. M., & Williams, N. S. G. (2015). Planning for cooler cities: A framework to prioritise green infrastructure to mitigate high temperatures in urban landscapes. *Landscape and Urban Planning*, *134*, 127–138. https://doi.org/10.1016/j.landurbplan.2014.10.018

Nowak, D. J., Hirabayashi, S., Bodine, A., & Greenfield, E. (2014). Tree and forest effects on air quality and human health in the United States. *Environmental Pollution*, *193*, 119–129. https://doi.org/10.1016/j.envpol.2014.05.028

Nowak, D. J., Hirabayashi, S., Doyle, M., McGovern, M., & Pasher, J. (2018). Air pollution removal by urban forests in Canada and its effect on air quality and human health. *Urban Forestry and Urban Greening*, *29*(September 2017), 40–48. https://doi.org/10.1016/j.ufug.2017.10.019

Ogryzek, M., Adamska-Kmieć, D., & Klimach, A. (2020). Sustainable transport: An efficient transportation network-case study. *Sustainability (Switzerland)*, *12*(19), 1–14. https://doi.org/10.3390/su12198274

Oke, T. R. (1982). The energetic basis of the urban heat island. *QJR Meteorol. Soc.*, *108*, 1–24.

Onishi, A., Cao, X., Ito, T., Shi, F., & Imura, H. (2010). Evaluating the potential for urban heat-island mitigation by greening parking lots. *Urban Forestry and Urban Greening*, *9*(4), 323–332. https://doi.org/10.1016/j.ufug.2010.06.002

Opdam, P., & Steingröver, E. (2018). How could companies engage in sustainable landscape management? An exploratory perspective. *Sustainability (Switzerland)*, *10*(1), 1–15. https://doi.org/10.3390/su10010220

Osborne, P. E., & Alvares-Sanches, T. (2019). Quantifying how landscape composition and configuration affect urban land surface temperatures using machine learning and neutral landscapes. *Computers, Environment and Urban Systems*, *76*(August 2018), 80–90. https://doi.org/10.1016/j.compenvurbsys.2019.04.003

Parastatidis, D., Mitraka, Z., Chrysoulakis, N., & Abrams, M. (2017). Online global land surface temperature estimation from landsat. *Remote Sensing*, *9*(12), 1–16. https://doi.org/10.3390/rs9121208

Park People. (2020). The Canadian City Parks Report. Retrieved February 16, 2021, from https://cityparksreport.parkpeople.ca/

Park People. (2021). COVID-19 and Parks: Highlights from our national surveys. Retrieved June

1, 2021, from https://parkpeople.ca/2020/07/16/covid-19-and-parks-highlights-from-our-national-surveys/

Patz, J. A., Campbell-Lendrum, D., Holloway, T., & Foley, J. A. (2005). Impact of regional climate change on human health. *Nature*, *438*(7066), 310–317. https://doi.org/10.1038/nature04188

Phalan, B., Onial, M., Balmford, A., & Green, R. E. (2011). Reconciling Food Production and Biodiversity Conservation: Land Sharing and Land Sparing Compared. *Science*, *333*(September), 1289–1291. https://doi.org/10.1126/science.1208742

Phalan, B. T. (2018). What have we learned from the land sparing-sharing model? *Sustainability (Switzerland)*, *10*(6), 1–24. https://doi.org/10.3390/su10061760

Pham, T. T. H., Apparicio, P., Landry, S., Séguin, A. M., & Gagnon, M. (2013). Predictors of the distribution of street and backyard vegetation in Montreal, Canada. *Urban Forestry and Urban Greening*, *12*(1), 18–27. https://doi.org/10.1016/j.ufug.2012.09.002

Pinault, L., Crouse, D., Jerrett, M., Brauer, M., & Tjepkema, M. (2016). Socioeconomic differences in nitrogen dioxide ambient air pollution exposure among children in the three largest Canadian cities. *Health Reports*, *27*(7), 3–9.

Pinheiro, J., Bates, D., DebRoy, S., & Sarkar, D. (2020). nlme. CRAN. Retrieved from https://cran.r-project.org/package=nlme

QGIS Development Team. (2020). QGIS. Open Source Geospatial Foundation. Retrieved from http://qgis.org

Rustgi, Di, Q., Dai, L., Wang, Y., Zanobetti, A., Choirat, C., … Dominici, F. (2018). Association of Short-Term Exposure to Air Pollution with Mortality in Older Adults. *JAMA*, *318*(24), 2446–2456. https://doi.org/10.1001/jama.2017.17923.Association

Samet, J., Dominici, F., Curriero, F., Coursac, I., & Zeger, S. (2000). Fine particulate air pollution and mortality in 20 U.S. cities, 1987-1994. *The New England Journal of Medicine*, *343*(24), 1742–1749.

Sarkar, C., Webster, C., & Gallacher, J. (2018). Residential greenness and prevalence of major depressive disorders: a cross-sectional, observational, associational study of 94 879 adult UK Biobank participants. *The Lancet Planetary Health*, *2*(4), e162–e173. https://doi.org/10.1016/S2542-5196(18)30051-2

Schell, C. J., Dyson, K., Fuentes, T. L., Roches, S. Des, Harris, N. C., Miller, D. S., … Lambert, M. R. (2020). The ecological and evolutionary consequences of systemic racism in urban environments. *Science*, *369*(6509). https://doi.org/10.1126/SCIENCE.AAY4497

Schwick, C., Jaeger, J., Bertiller, R., & Kienast, F. (2012). *Urban Sprawl in Switzerland - Unstoppable?* (R. Landolt, Ed.) (1st ed.). Zurich: Bristol-Stiftung.

Selmi, W., Weber, C., Rivière, E., Blond, N., Mehdi, L., & Nowak, D. (2016). Air pollution removal by trees in public green spaces in Strasbourg city, France. *Urban Forestry and Urban Greening*, *17*(2), 192–201. https://doi.org/10.1016/j.ufug.2016.04.010

Smargiassi, A., Baldwin, M., Pilger, C., Dugandzic, R., & Brauer, M. (2005). Small-scale spatial variability of particle concentrations and traffic levels in Montreal: A pilot study. *Science of the Total Environment*, *338*(3), 243–251. https://doi.org/10.1016/j.scitotenv.2004.07.013

Soga, M., Yamaura, Y., Koike, S., & Gaston, K. J. (2014). Land sharing vs. land sparing: Does the compact city reconcile urban development and biodiversity conservation? *Journal of Applied Ecology*, *51*(5), 1378–1386. https://doi.org/10.1111/1365-2664.12280

Spotswood, E. N., Beller, E. E., Grossinger, R., Grenier, J. L., Heller, N. E., & Aronson, M. F. J. (2021). The Biological Deserts Fallacy: Cities in Their Landscapes Contribute More than We Think to Regional Biodiversity. *BioScience*, *XX*(X), 1–13. https://doi.org/10.1093/biosci/biaa155

Spronken-Smith, R. A., & Oke, T. R. (1999). Scale modelling of nocturnal cooling in urban parks. *Boundary-Layer Meteorology*, *93*(2), 287–312. https://doi.org/10.1023/A:1002001408973

Statistics Canada. (2016). Census Profile, 2016 Census. Retrieved January 1, 2021, from https://www12.statcan.gc.ca/census-recensement/2016/dp-pd/prof/details/page.cfm?Lang=E&Geo1=CMACA&Code1=462&Geo2=PR&Code2=01&Data=Count&SearchText=montreal&SearchType=Begins&SearchPR=01&B1=All&TABID=1

Statistics Canada. (2020). Proximity to public transportation in Canada's metropolitan areas. Retrieved January 23, 2021, from https://www150.statcan.gc.ca/n1/daily-quotidien/200602/dq200602a-eng.htm

Tran, D. X., Pla, F., Latorre-Carmona, P., Myint, S. W., Caetano, M., & Kieu, H. V. (2017). Characterizing the relationship between land use land cover change and land surface temperature. *ISPRS Journal of Photogrammetry and Remote Sensing*, *124*, 119–132. https://doi.org/10.1016/j.isprsjprs.2017.01.001

Tzoulas, K., Korpela, K., Venn, S., Yli-Pelkonen, V., Kaźmierczak, A., Niemela, J., & James, P. (2007). Promoting ecosystem and human health in urban areas using Green Infrastructure: A literature review. *Landscape and Urban Planning*, *81*(3), 167–178. https://doi.org/10.1016/J.LANDURBPLAN.2007.02.001

United Nations, Department of Economic and Social Affairs, P. D. (2019). *World Urbanization Prospects 2018: Highlights*.

van Vliet, J. (2019). Direct and indirect loss of natural area from urban expansion. *Nature Sustainability*, *2*(8), 755–763. https://doi.org/10.1038/s41893-019-0340-0

Villamagna, A. M., Angermeier, P. L., & Bennett, E. M. (2013). Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. *Ecological Complexity*, *15*, 114–121. https://doi.org/10.1016/j.ecocom.2013.07.004

Ville de Montreal. (2020). Unités d'évaluation foncière. Retrieved from https://donnees.montreal.ca/ville-de-montreal/unites-evaluation-fonciere

Wang, X., Dallimer, M., Scott, C. E., Shi, W., & Gao, J. (2021). Tree species richness and diversity predicts the magnitude of urban heat island mitigation effects of greenspaces. *Science of The Total Environment*, *770*, 145211. https://doi.org/10.1016/j.scitotenv.2021.145211

Wang, Y., Berardi, U., & Akbari, H. (2016). Comparing the effects of urban heat island mitigation strategies for Toronto, Canada. *Energy and Buildings*, *114*, 2–19. https://doi.org/10.1016/j.enbuild.2015.06.046

Westgate, G. (2018). Equity and Green Space in Montreal: Potential for Healthier and More Inclusive Communities, (August). Retrieved from www.socialconnectedness.org

WHO Regional Office for Europe. (2016). Urban green spaces and health, 92.

Wolf, K. L., Lam, S. T., McKeen, J. K., Richardson, G. R. A., Bosch, M. van den, & Bardekjian, A. C. (2020). Urban trees and human health: A scoping review. *International Journal of Environmental Research and Public Health*, *17*(12), 1–30. https://doi.org/10.3390/ijerph17124371

Wood, S. N. (2011). mgcv. Retrieved from https://rdocumentation.org/packages/mgcv/versions/1.8-36

Yang, Jiachuan, & Bou-zeid, E. (2019). Scale dependence of the benefits and efficiency of green and cool roofs. *Landscape and Urban Planning*, *185*(February), 127–140. https://doi.org/10.1016/j.landurbplan.2019.02.004

Yang, Jun, Chang, Y., & Yan, P. (2015). Ranking the suitability of common urban tree species for controlling PM2.5pollution. *Atmospheric Pollution Research*, *6*(2), 267–277. https://doi.org/10.5094/APR.2015.031

Yli-Pelkonen, V., Scott, A. A., Viippola, V., & Setälä, H. (2017). Trees in urban parks and forests reduce O3, but not NO2 concentrations in Baltimore, MD, USA. *Atmospheric Environment*, *167*(2), 73–80. https://doi.org/10.1016/j.atmosenv.2017.08.020

Yli-Pelkonen, V., Setälä, H., & Viippola, V. (2017). Urban forests near roads do not reduce gaseous air pollutant concentrations but have an impact on particles levels. *Landscape and Urban Planning*, *158*(2), 39–47. https://doi.org/10.1016/j.landurbplan.2016.09.014

Zhao, D., Lei, Q., Shi, Y., Wang, M., Chen, S., Shah, K., & Ji, W. (2020). Role of species and planting configuration on transpiration and microclimate for urban trees. *Forests*, *11*(8). https://doi.org/10.3390/f11080825

Zhou, W., Wang, J., & Cadenasso, M. L. (2017). Effects of the spatial configuration of trees on urban heat mitigation: A comparative study. *Remote Sensing of Environment*, *195*, 1–12. https://doi.org/10.1016/j.rse.2017.03.043

Ziter, C. D., Pedersen, E. J., Kucharik, C. J., & Turner, M. G. (2019). Scale-dependent interactions between tree canopy cover and impervious surfaces reduce daytime urban heat during summer. *Proceedings of the National Academy of Sciences*, *116*(15), 7575–7580. https://doi.org/10.1073/pnas.1817561116

Tables and Figures

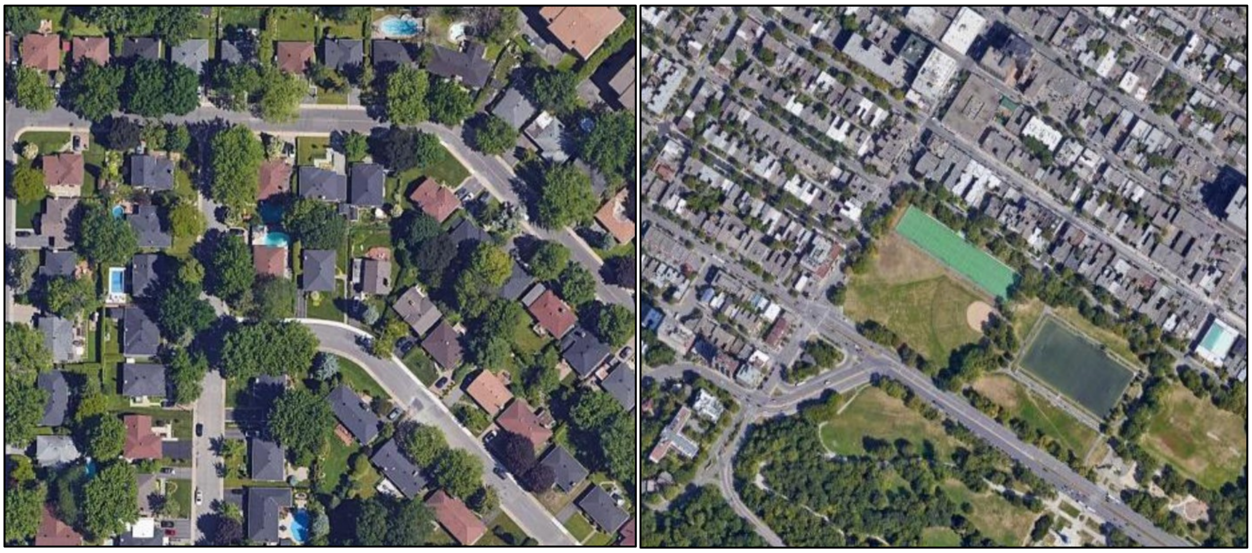


Fig. 1: ‘Land-sharing’ refers to spread out urban development, like the suburban areas in the photo on the left. ‘Land-sparing’ refers to compact development that leaves natural area outside the developed area, similar to the photo on the right. (Google satellite image).

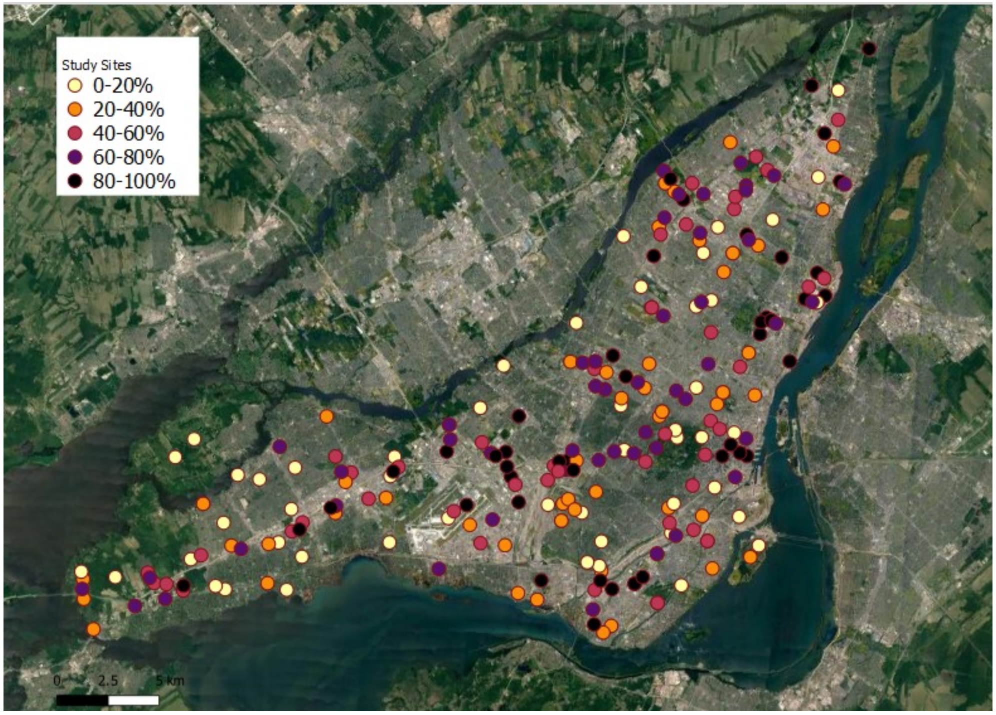


Fig 2: Image of the Montreal area with dots indicating locations of 250 study sites (Google satellite image). The colour of the dot represents the building density at that particular site (yellow=low, purple=medium, black=high).

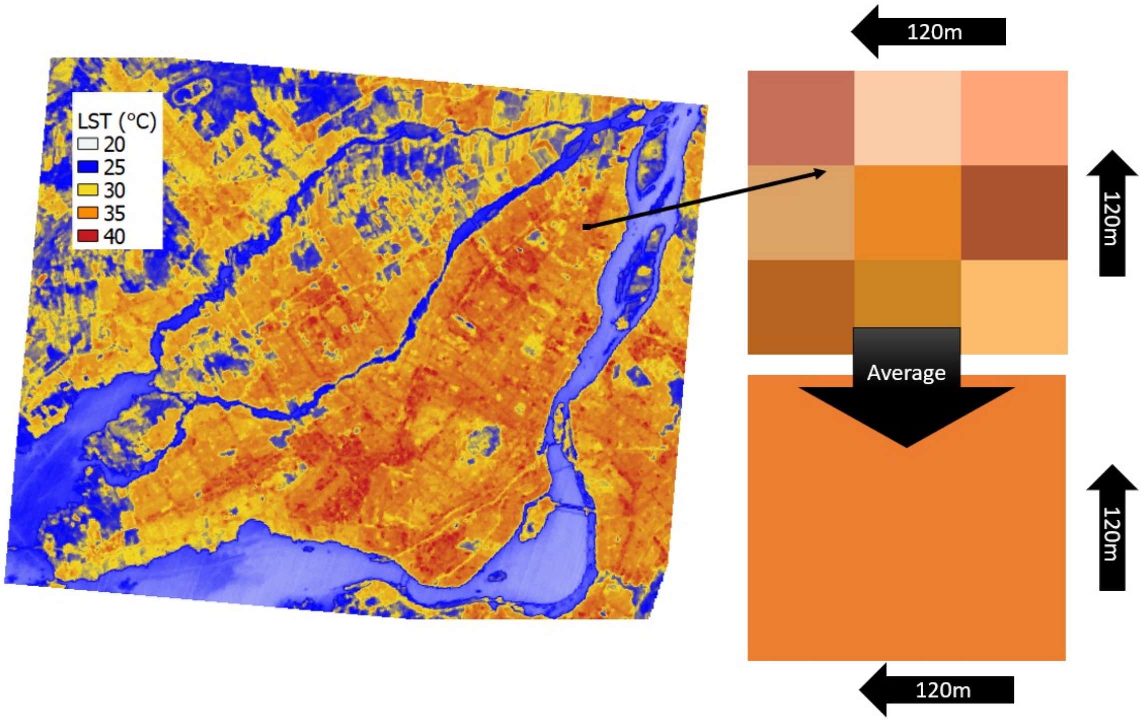


Fig 3: Land surface temperature (LST) values (°C) were derived from a Landsat 8 satellite image as shown on the left. The average value of the pixels contained in each study site was taken to be the value used in analysis as shown on the right.

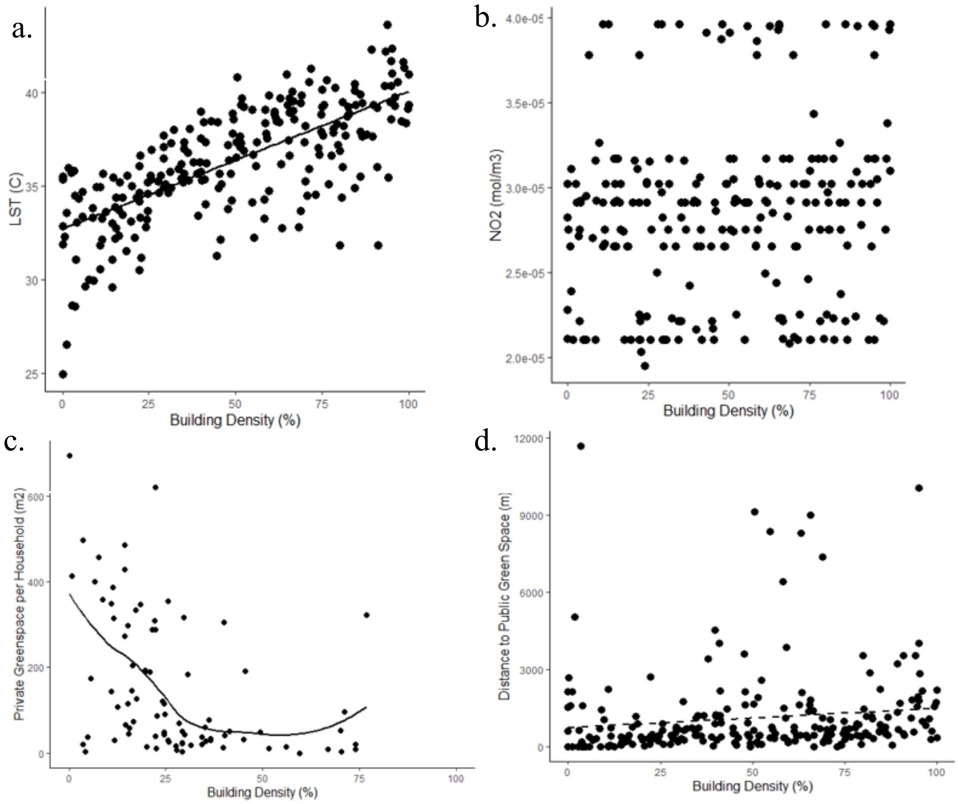


Fig 4: (a) Land surface temperature (LST), (b) NO_2 concentration, (c) area of private residential green space per household, and (d) distance to the nearest public green space as functions of building density (%) for 250 study sites. The plot of area of private green space has fewer points because private residential green space only exists on residential land parcels. Trend lines indicate statistical significance and dashed trend line indicates marginal significance.

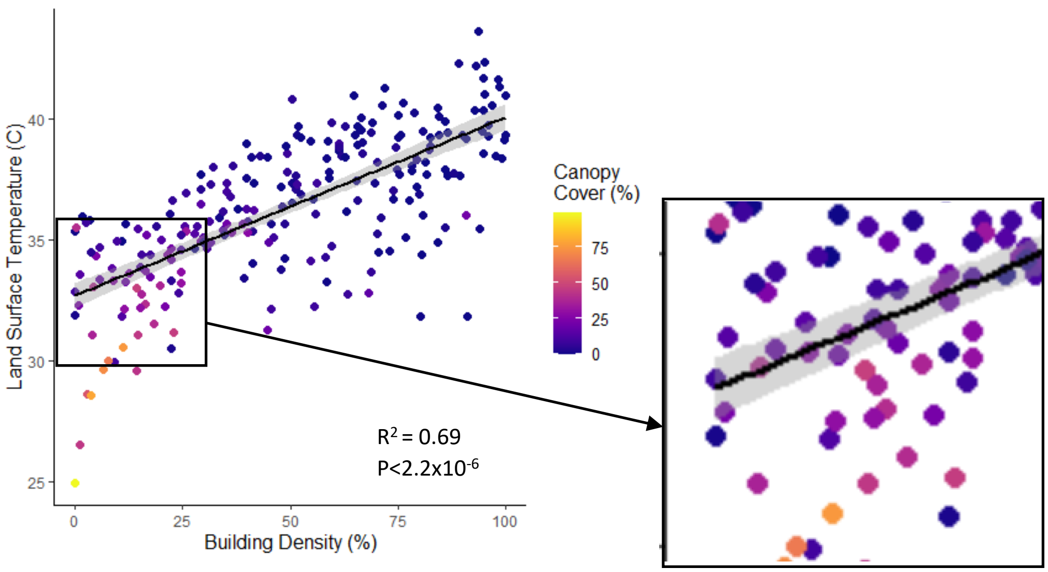


Fig 5: Daytime land surface temperature (LST) as a function of building density for 250 sites across the Island of Montreal. Shaded area represents +/-2 SE. Colour represents percent canopy cover within each site. Some of the spread of the points in the building density – LST relationship can be explained by differences in canopy cover. When we zoom into the low building density section of the plot (inset figure) we can see that sites with lower LST tend to have higher canopy cover, even at the same building density.

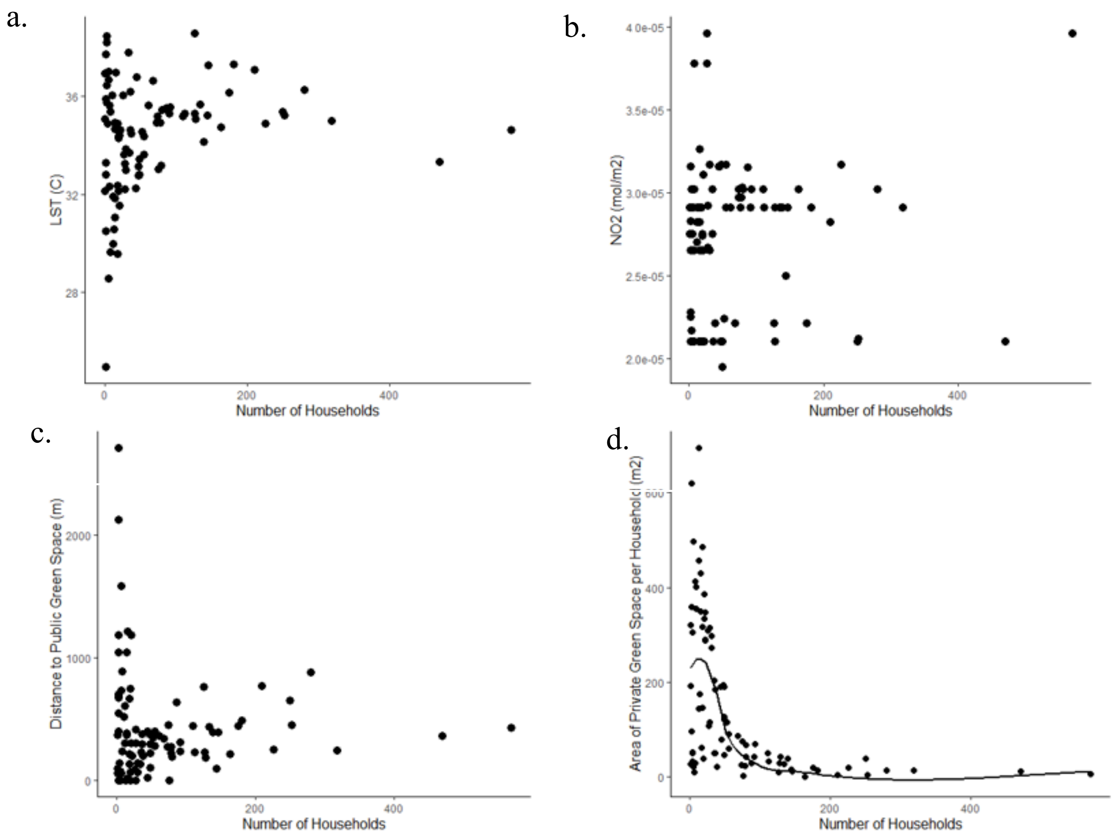


Fig 6: (a) Land surface temperature (LST), (b) NO_2 concentration, (c) distance to the nearest public green space, and (d) area of private residential green space per household as functions of household density for study sites containing residential buildings. Trend lines indicate statistical significance.

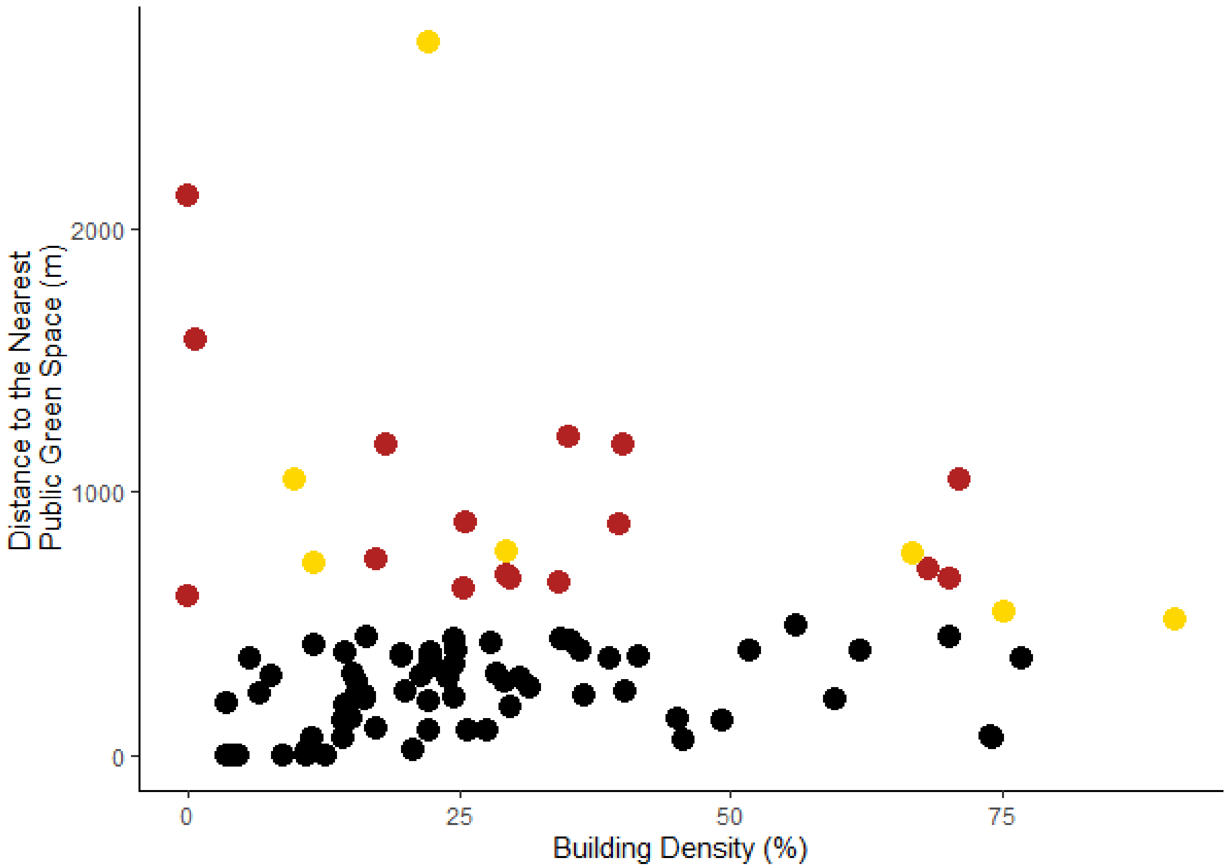


Fig 7: Distance to the nearest public green space as a function of building density for residential study sites. Red points indicate study sites that did not meet World Health Organization (WHO) guidelines of public green space access in terms of distance. Yellow points indicate study sites that did not meet the WHO guidelines of distance to public green space and also do not have access to private green space (backyards).

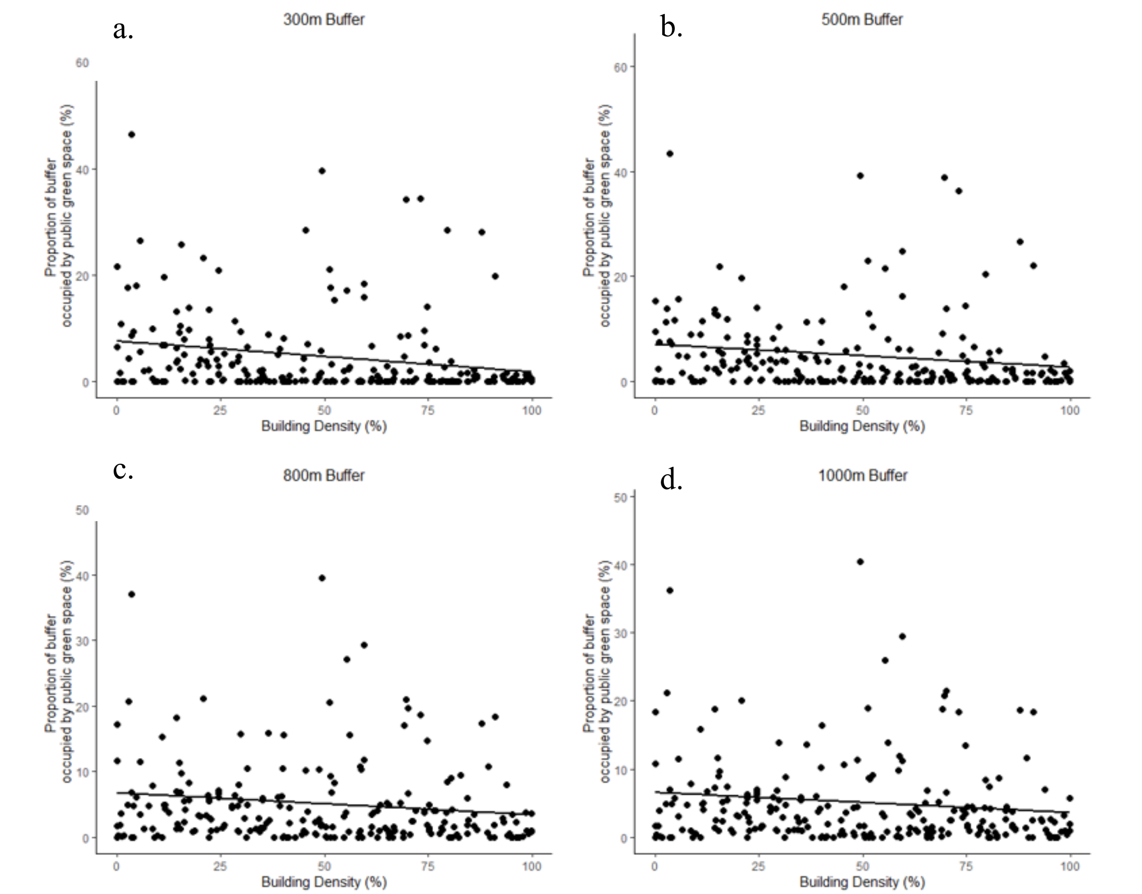


Fig 8: Proportion (%) of publicly accessible green space within a buffer of (a) 300 m, (b) 500 m, (c) 800 m, and (d) 1000 m plotted against building density for 250 study sites. Trend lines indicate statistical significance. For all buffer sizes, there was a weakly negative relationship between building density and the amount of green space within the buffer.

Fig 9: Aerial view of study site 12 located in Côte-des-Neiges neighbourhood (Google satellite image). This study site maintained land surface temperature (LST) below 35°C despite having a high building density of over 50%. This may have been in part due to the relatively high canopy cover at this site compared to other sites of similar density (16% at site 12 vs. ~3% at similarly dense sites)

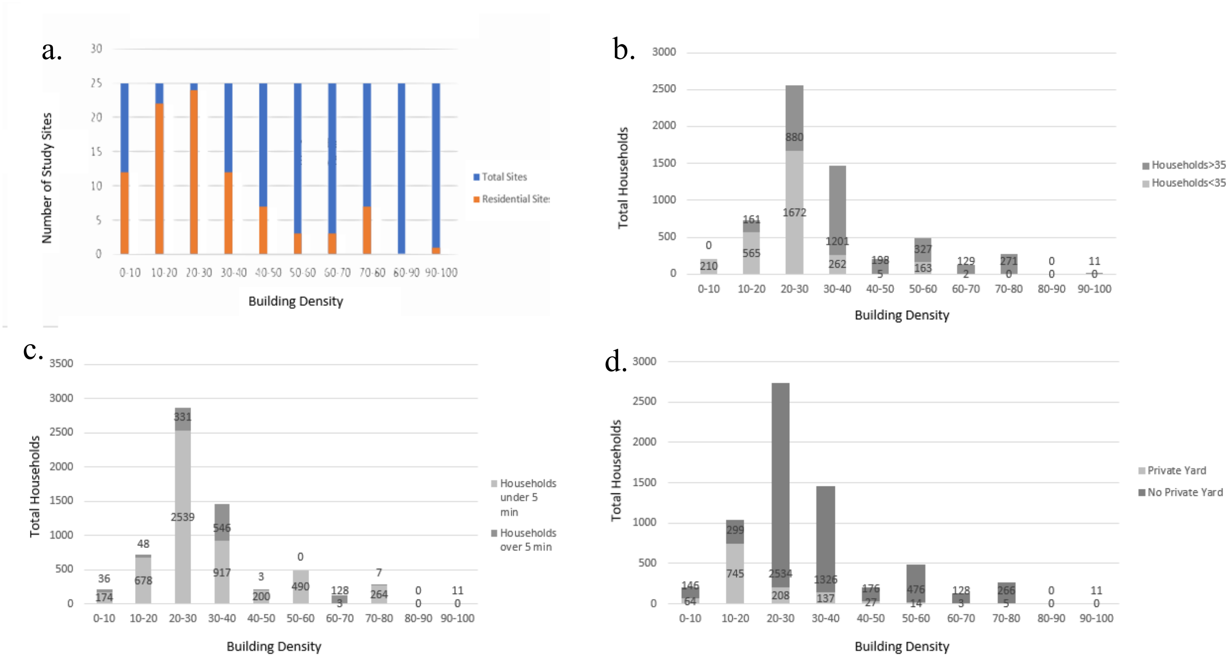


Fig. 10: (a) The proportion of study sites that were residential for each building density category (10% increment in building density). (b) The proportion of households benefitting from temperatures under 35°C (the median temperature at our sites) for each density category. (c) The proportion of households benefitting from the WHO recommended access to public green space and (d) the proportion of households benefitting from access to a private green space

Table 1: Fixed and random effects for linear mixed model predicting mean urban land surface temperature (LST) (°C) at site level (n=250), based on building density, canopy cover, impervious cover, log(median household income), and log(distance to the river). Table represents only what was included in the final version of the model.

	Estimate	Standard Error	t-value	p-value
Intercept	41.84	3.33	12.57	<0.001
Building Density	0.06	0.006	9.623	<0.001
% Canopy	-0.06	0.01	-5.993	<0.001
Log(Income)	-1.11	0.29	-3.769	<0.001
Log(Distance to the River)	0.453	0.129	3.522	0.03
% Impervious	0.023	0.007	3.099	<0.001
Random Effect				
Borough	1.03			

Table 2: Fixed effects for a general additive model predicting mean concentration of nitrogen dioxide (mol/m^2) at site level (n=250), based on building density, canopy cover, log(distance to the nearest major road), and log(median household income). Table represents only what was included in the final version of the model.

	Estimate	Standard Error	t-value	p-value
Intercept	$2.8e^{-05}$	$1.1e^{-07}$	249.71	< 0.001
Building Density	$-7.8e^{-10}$	$5.6e^{-09}$	-0.14	0.89
Log(Income)	$1.2e^{-06}$	$4.5e^{-07}$	2.76	0.006
Log(Distance to Road)	$3.8e^{-07}$	$1.3e^{-07}$	3.03	0.003
% Canopy	$6.0e^{-09}$	$1.1e^{-08}$	0.56	0.58
Smooth Term	*edf*	*Ref.df*	*F*	*p-value*
S(Lat,Long)	27.41	28.84	63.77	<0.001

Appendix I: Comparisons of satellite imagery across different days

To ensure that my results in regards to land surface temperature were not simply an artefact of the day I had chosen for analysis, I repeated my analysis using LST from multiple different days during different times of the year. I selected the least cloudy image for each of May, July and October (8th, 27th, 15th, respectively). My results for the relationship between building density and LST are robust to differences in day of the year (Fig. A1). The relationship remained consistent using data from May, July, and October.

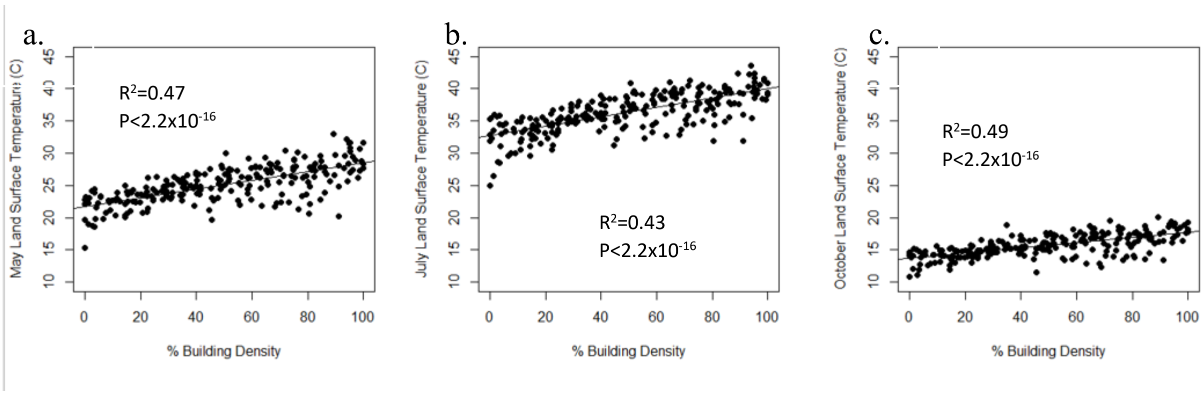


Fig. A1: The relationship between LST and building density is consistent across multiple times of year that have leaves on trees. From left to right, these plots represent this relationship on the least cloudy day in May (a), July (b), and October (c).

To ensure that my results in regards to NO_2 concentrations were not simply an artefact of the day I had chosen for analysis, I repeated this analysis using NO_2 from multiple different days during different times of the year. I selected the images from the same days I used of May, July and October as I selected for the LST analysis. The relationship between building density and NO_2 concentrations was consistently uniform among days, with no trend in either direction $R^2<0.02$ (Fig. A2).

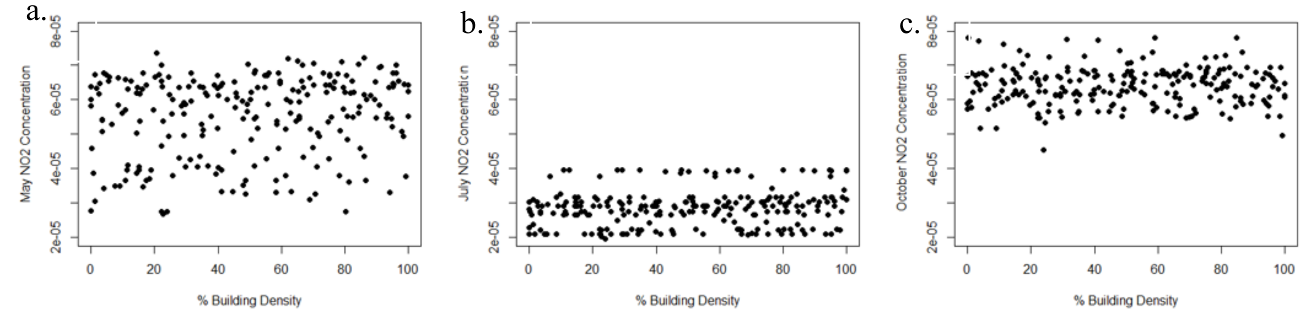


Fig. A2: The relationship between building density and NO_2 concentrations (mol/m^2) is consistently uniform across all times of the year that have leaves on trees. From left to right this pattern is shown here for the least cloudy day in each of July (b), May (a), and October (c). All $R^2<0.02$. While the general amount of NO_2 in the air differed between seasons, the relationship to building density remained constant.

Appendix II: Multiple metrics of green space access

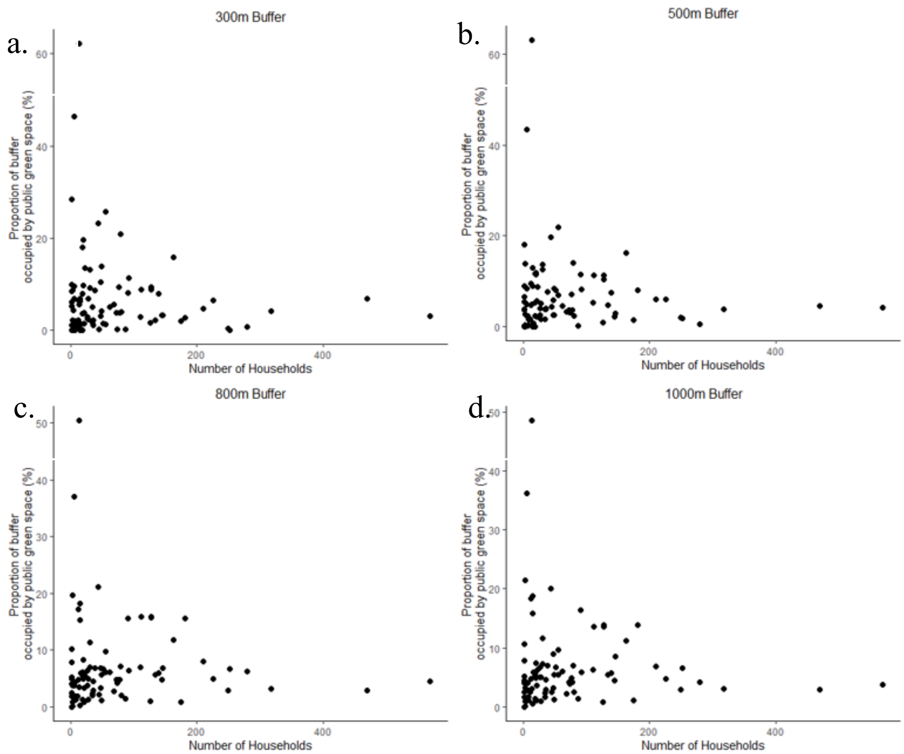


Fig. A3: Surface area of publicly accessible green space within a buffer of (a) 300 m, (b) 500 m, (c) 800 m, and (d) 1000 m plotted against the number of households at a study site. For all buffer sizes, there was a non-statistically significant negative relationship between the number of households at a site and the amount of green space within the buffer. All p-values >0.1.

Appendix III: Creating the impervious cover layer

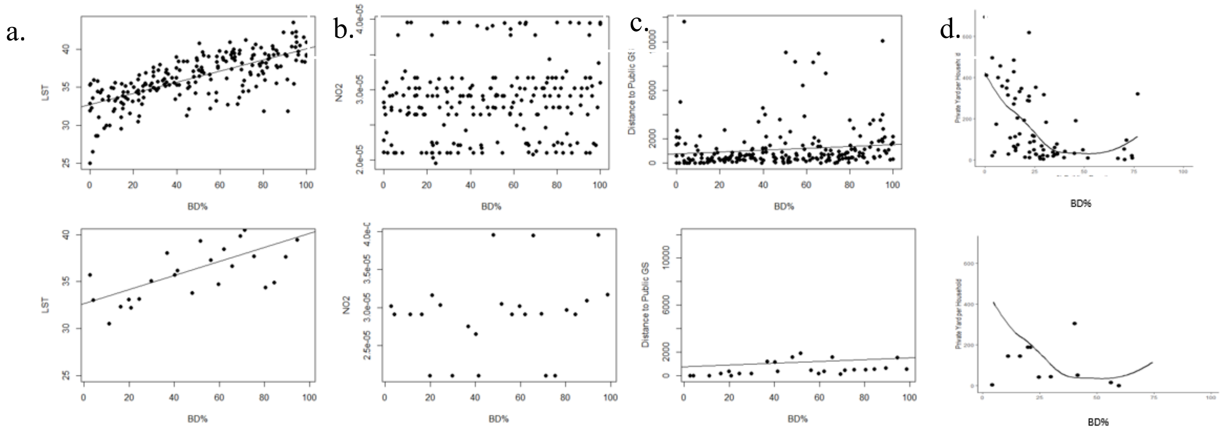
Prior to this project, there was no existing layer of all impervious surfaces for Montreal. The layer of impervious cover was created using a two-step process. The first step was to combine multiple existing GIS layers mapping specific portions of impervious cover. This included a layer of building footprints, the road network, sidewalks, bicycle paths, and intersections. The sources of these layers are listed and linked in the table below.

Resource	Description	Link
CMM Util-Sol	A layer classifying every land parcel in Montreal into land use types. We used classifications: 700, 710, 720, 725, 750, 760.	http://observatoire.cmm.qc.ca/observatoire-grand-montreal/produits-cartographiques/donnees-georeferencees/
Montreal Open Data Portal Voirie-actif	A layer of all roadways, sidewalks and intersections	http://donnees.ville.montreal.qc.ca/dataset/voirie-actif
Montreal Open Data Portal Pistes-cyclables	A layer of all bicycle paths in Montreal	http://donnees.ville.montreal.qc.ca/dataset/pistes-cyclables
Microsoft Building Footprints	A layer rendering the building footprints of every building in Quebec	https://github.com/Microsoft/CanadianBuildingFootprints

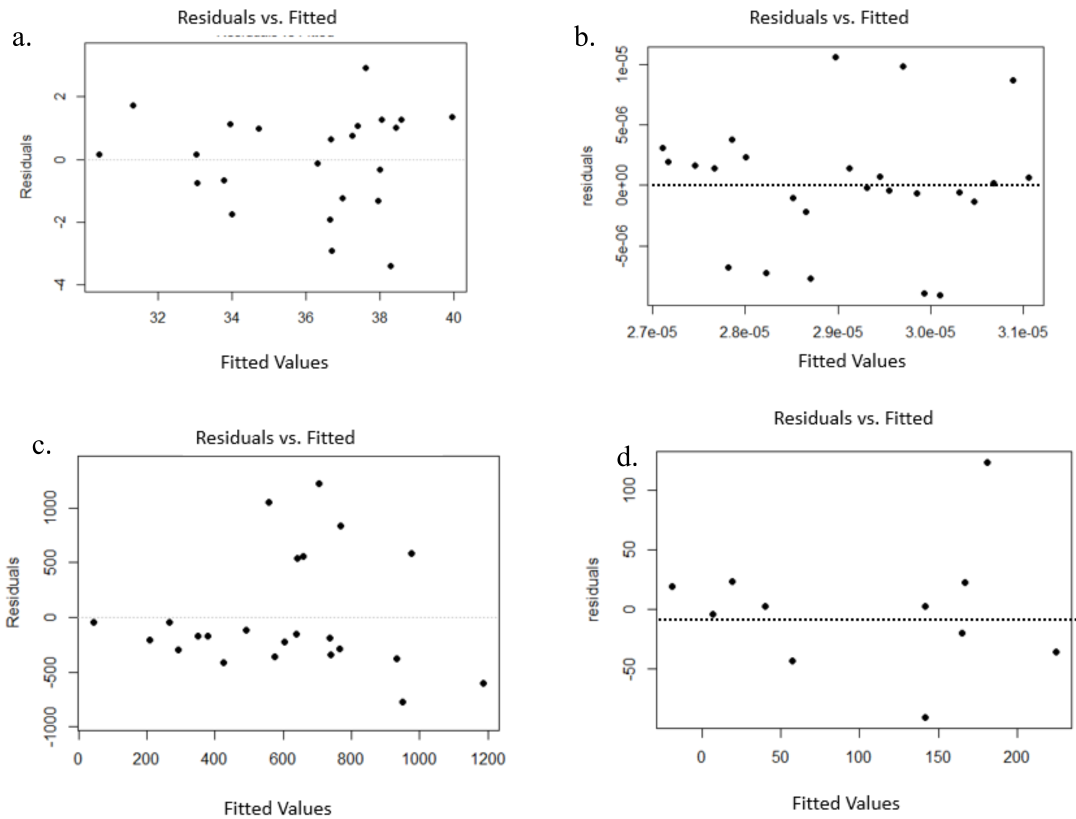
Once these layers were combined, all remaining impervious surfaces (including buildings, parking areas, recreational areas, roads, and misc. impervious) were hand digitized onto the map. This was completed by tracing features from the Google StreetView Satellite images available in QGIS. We divided the island into a 1 km x 1 km grid and moved grid cell by grid cell to ensure no area was missed. The final layer produced included all impervious surfaces in Montreal with the exception of minor impervious features such as driveways at private homes, which we were unable to digitize due to time constraints within the project. A unique feature of this layer, is that due to the hand digitizing process involved, impervious features that are under vegetation are largely included, even though they are often missed in remote sensing based impervious datasets.

This layer has a few caveats that users should be aware of. Curvy roads and pathways may not be perfectly shaped like their real-life counterparts because they were digitized by hand. Since the layer was created by referencing satellite imagery, where parts of a concrete feature were visually blocked by the canopy cover there may be some errors. In some cases, corners of features were difficult to digitize so that they may be slightly off-centre of the location of the real-life corner in the GIS layer. Finally, it was sometimes unclear if a small parking area was a public parking lot that should be digitized, or a private driveway, so as a result, some private driveways may have been included (we included these if we were unsure). Thus, while I am confident the layer is sufficient for the purposes of the broad patterns investigated in the current study, it may be a slight underrepresentation of total impervious cover.

Appendix IV



A4: Test vs. Training data for building density models for a) LST, b) NO_2, c) distance to public green space, and d) amount of private green space. For each model, the data was split into a "training" dataset of 90% of the data, and a "test" dataset of 10% of the data. Each model was first run using the training data, then again using the test data, and finally as a complete model which was used in analysis. The top and bottom panels of each figure represent the training data (top) and the test data (bottom). The regression lines represent the regression line for the training model.



A5: Plots of residuals (observed – predicted values) vs. fitted values for models for the 10% test data set of a) LST, b) NO2, c) distance to public green space and, d) amount of private green space. The correlation coefficients for predicted vs. observed vales of test data are as follows: a) 0.69, b) 0.43, c) 0.52, and d) 0.82.