ÉCOSCIENCE

Écoscience

Taylor & Francis
Taylor & Francis Group

ISSN: 1195-6860 (Print) 2376-7626 (Online) Journal homepage: www.tandfonline.com/journals/teco20

# Forest structure but not tree diversity differs among urban woodlands with differing conservation status

Erica Padvaiskas, Isabella C Richmond & Carly D Ziter

**To cite this article:** Erica Padvaiskas, Isabella C Richmond & Carly D Ziter (2025) Forest structure but not tree diversity differs among urban woodlands with differing conservation status, Écoscience, 32:1, 35-46, DOI: 10.1080/11956860.2025.2495427

**To link to this article:** https://doi.org/10.1080/11956860.2025.2495427

ÉCOSCIENCE
2025, VOL. 32, NO. 1, 35–46
https://doi.org/10.1080/11956860.2025.2495427



# Forest structure but not tree diversity differs among urban woodlands with differing conservation status

Erica Padvaiskas, Isabella C Richmond and Carly D Ziter

Department of Biology, Concordia University, Montreal, Canada

**ABSTRACT**

While biodiversity conservation in urban areas is a topic of great interest, few studies have focused on the role that urban conservation areas have for preserving biodiversity. Urban woodlands, which are patches of forest habitat confined within the city's boundaries, offer a promising approach to evaluate the importance of conservation areas within cities. Here, we examined the relationship between conservation status, forest structure and composition across 11 urban woodlands in Montréal, Canada. We used field surveys to assess biodiversity, canopy cover, and structural complexity for urban woodlands with a conservation status and those without. We found that Montréal's urban woodlands fostered similar levels of biodiversity regardless of conservation status. Similarly, all urban woodlands supported high proportions of native tree species despite differences in conservation status and associated management. Our results suggest that both conservation areas and non-status woodlands play an important role in safeguarding urban biodiversity. Woodlands with a conservation status, however, had higher canopy cover and vegetative complexity, but also contained higher average proportions of invasive trees, particularly *Rhamnus cathartica* (Common Buckthorn). While the high complexity in vegetation layers observed may provide habitat to native wildlife, these benefits may be limited by the high proportion of invasive trees.

**RÉSUMÉ**

La conservation de la biodiversité dans les zones urbaines est un sujet important, mais peu d'études se sont penchées sur le rôle des aires protégées urbaines à cet égard. Les boisés urbains, qui sont des parcelles forestières à l'intérieur des limites d'une ville, offrent l'occasion d'évaluer l'importance des aires protégées urbaines. Nous avons étudié la relation entre le statut de conservation et la structure et la composition forestières dans 11 boisés urbains à Montréal, Canada. Nous avons utilisé des relevés terrain pour évaluer la biodiversité, le couvert forestier et la complexité structurelle de boisés urbains avec ou sans statut de conservation. Nous avons trouvé que les boisés urbains de Montréal avaient des niveaux de biodiversité similaires sans égard à leur statut de conservation. De plus, tous les boisés urbains avaient des proportions élevées d'espèces indigènes, et ce, peu importe leur statut de conservation ou leur historique d'aménagement. Nos résultats suggèrent que les deux types de boisés (protégés ou pas) jouent un rôle important dans la conservation de la biodiversité urbaine. Toutefois, les boisés avec statut de conservation avaient un couvert forestier plus important et une plus grande complexité végétale. Ils avaient aussi une proportion moyenne d'espèces arborescentes envahissantes plus élevée, particulièrement *Rhamnus cathartica* (nerprun cathartique). Ainsi, alors que la complexité végétale élevée peut fournir des habitats à la faune indigène, ces bénéfices pourraient être limités par la proportion élevée d'espèces envahissantes.

**ARTICLE HISTORY**
Received 5 August 2024
Accepted 13 April 2025

**KEYWORDS**
Forest management; nature parks; urban invasive species; urban trees; vegetative complexity

**MOTS CLÉS**
parcs naturels; arbres urbains; aménagement forestier; complexité végétale; espèces envahissantes urbaines

## Introduction

Institutions and communities alike have recognized the need for nature-based solutions to build more sustainable cities, for the long-term benefit of both people and biodiversity (United Nations Sustainable Development 2015; Kabisch et al. 2022). One management strategy for increasing the species richness and abundance of many taxa in cities is the establishment of urban protected areas via the designation of a formal conservation status (Trzyna et al. 2014). Protected areas are an important strategy for safeguarding or conserving biological diversity (Leroux et al. 2010) and are rooted in many global conservation strategies such as the Convention on Biological Diversity and the United Nations Sustainable Development Goals. However, the effectiveness of protected areas for biodiversity

**CONTACT** Isabella C Richmond isabella.richmond@mail.concordia.ca Department of Biology, Concordia University, 7141 Sherbrooke St. W. Montréal QC, H4B 1R6

Supplemental data for this article can be accessed online at https://doi.org/10.1080/11956860.2025.2495427.

conservation is predominately examined in the rural landscape (Gaston et al. 2008; Rayner et al. 2014) resulting in a lack of information regarding the ecological success of urban protected areas (Miller and Hobbs 2002; Ioja and Breuste 2020).

Urban and urban-adjacent protected areas are anticipated to increase in the coming decades, both through increased proximity of existing protected lands to cities due to urbanization (McDonald et al. 2020), as well as dedicated programs to protect valuable natural habitats in cities, such as Canada's National Urban Parks Plan (Parks Canada Agency 2023) or Singapore's Green Plan 2030 (Singapore Government 2024). Urban protected areas differ significantly from their non-urban counterparts in terms of frequency and density of visitors, number of involved partners and stakeholders, and exposure to multiple anthropogenic stressors (Trzyna et al. 2014; Ioja and Breuste 2020). Thus, studies that evaluate the effectiveness of urban protected areas for sustaining specific aspects of biodiversity are crucial for making relevant decisions and achieving urban conservation goals.

While urban nature is highly diverse in composition and structure, urban woodlands are often targeted as potential protected areas due to their ecological, social, and recreational importance in many parts of the world (Kowarik and Körner 2005; Trentanovi et al. 2021). Urban woodlands are patches of habitat that vary in size and connectedness, preserved through the heterogeneous development of cities (Kowarik and Körner 2005). Urban woodlands are structurally similar to natural forest stands, with continuous canopy cover and an unmanaged ground layer (Hedblom and Söderström 2008; Pn et al. 2017). However, unlike natural forest stands, urban woodlands are surrounded by an urban matrix. Although forested, an urban woodland is not a synonym of urban forest, which the Canadian Urban Forest Strategy defines as the 'trees, forests, greenspace and related abiotic, biotic and cultural components in areas extending from the urban core to the urban-rural fringe' (Tree Canada 2018, p. 6). Instead, urban woodlands are an important component of the urban forest and are the most common type of green space for many cities in terms of area cover, including in densely populated areas (Pauleit et al. 2005).

The vegetation complexity and stem density of urban woodlands allows for provision of otherwise underrepresented and complementary ecosystem services not produced by other components of the urban forest such as manicured parks or street trees (Vieira et al. 2018). For example, air purification and climate regulation are more prominent in unmanaged urban woodlands compared to intensely managed green spaces such as plantations and lawns (Vieira et al. 2018). Urban woodlands are also sought after for their contribution to cultural or non-material ecosystem services, including recreation and stress reduction (Lupp et al. 2016; Yao et al. 2021).

In addition to providing ecosystem services, urban woodlands offer an opportunity to investigate the effectiveness of urban-protected areas for multiple aspects of biodiversity conservation. Unlike other urban green spaces, urban woodlands contain high tree density and complex vegetation layers (trees, shrubs and herbaceous strata) (Vieira et al. 2018; Xu et al. 2021). High canopy cover and the presence of many vegetation layers provide habitat to birds, amphibians, and other small animals (Noël and Lapointe 2010; Lintott et al. 2014; Otieno and Mutati 2021). A study conducted in Rennes, France, found that urban woodlands can support more than 50% of the species present in adjacent peri-urban forests (Croci et al. 2008). Urban woodlands are also capable of maintaining high levels of native tree species. In New York City, 84% of the total canopy cover of urban woodlands is native (Pregitzer et al. 2019). In comparison, a city-scale urban canopy assessment of New York City revealed that only 55% of the overall canopy was native (Nowak 2007). Native trees have coevolved with other native biotas, resulting in the capability of native tree species to support high levels of native insects and birds (Narango et al. 2018). For example, Tallamy and Shropshire (2009) found that native plant genera supported 3-fold more caterpillars than non-native plant genera.

Notwithstanding their capacity to conserve native trees relative to more manicured urban green spaces, urban woodlands are vulnerable to degradation due to high fragmentation, frequent disturbance, and complex land use histories (Ramalho et al. 2014; Pregitzer and Bradford 2023). In the above-cited example of New York City, for example, non-native species were prevalent in groundcover layers. Similarly, non-native shrubs are also highly prevalent in urban and urban-adjacent woodlands (Johnson et al. 2020). Urban woodland management likely plays an important – but still poorly understood – role in determining biodiversity outcomes across these highly variable social-ecological systems (Pluess et al. 2012).

While the benefits of urban woodlands are increasingly acknowledged, their management outcomes remain understudied relative to street trees, manicured parks, and urban canopy over multiple land use types (Hedblom and Söderström 2008; Pn et al. 2017; Pregitzer et al. 2019). Specifically, although the establishment of protected areas through a formal conservation status designation is a management strategy used in urban woodlands (Trzyna et al. 2014), the success of this management strategy is not well understood. Our study investigated the importance of establishing conservation areas for maintaining tree biodiversity in urban woodlands open to public recreational activities. Specifically, we examined how formal status and associated management of urban woodlands targeted towards biodiversity conservation determines biodiversity outcomes. Through a case study of parks across the Island of Montreal, Quebec, Canada, we asked, how does forest composition and structure differ between urban woodlands that have an official conservation status (i.e., nature parks) and those which do not (i.e., non-status woodlands in urban parks)?

Urban woodlands are valuable contributors to urban biodiversity and provide key forested habitats within the city boundaries (Hedblom and Söderström 2008). Within this system, we hypothesized that there would be key differences in the presence of woody invasive species, tree species richness, tree Shannon diversity, canopy cover, and vegetative complexity between urban woodlands with and without an official conservation status, given that management plans in conservation sites are specifically designed to protect and enhance biodiversity and habitat quality. For example, the presence of an official protected park area has resulted in an increase in vegetative native species, species richness, alpha diversity, and gamma diversity in two Quebec parks (Bourdages and Lavoie 2011; Beauvais et al. 2016).

First, we hypothesized that conservation areas will foster fewer invasive woody species and stems than nearby non-status woodlands. We expected these differences because the Ecosystem Management Program used in Montreal nature parks includes active monitoring and removal of invasive species such as *Rhamnus cathartica* (common buckthorn) and *Rhamnus frangula* (glossy buckthorn) (de Montréal 2013), and long-term management has been found to reduce the impacts of invasion in urban woodlands (Johnson and Handel 2019). Consequently, we hypothesized that the nature parks – if subject to less invasion – would retain a higher tree species richness and Shannon diversity compared to non-status woodlands.

Second, we hypothesized that the forest structure will differ among non-status woodlands and nature parks, with nature parks having higher levels of canopy cover and more complexity in vegetation structure compared to non-status parks. We expected to see these differences in forest structure due to the contrasting management strategies in the nature parks versus non-status woodlands. The management of nature parks aims to maximize biodiversity, as such shrubs, understory, and woody debris are not removed (de Montréal 2013). In contrast, the management of non-status woodlands may focus more strongly on providing recreation opportunities which entails the removal of the shrub and understory layers. We also assume that park visitors are less likely to engage in off-trail recreational activities in nature-parks compared to non-status woodlands due to more prevalent signage and well-established trail networks, leading to reduced human pressure on plant biodiversity compared to non-status woodlands – further reinforcing our above hypotheses (Ballantyne and Pickering 2015).

## Methods

### *Study site*

We conducted this study across the Island of Montréal, located in southern Québec, Canada (45.50 °N, 73.75 °W). Montréal is the second most populous city in Canada, with over 2 million residents on the island (Government of Canada and Statistics Canada 2023). Sampling was conducted across urban woodlands that were subsequently categorized as parks: (1) with a conservation status (i.e., Montréal's nature parks) or (2) without a conservation status (i.e., non-status woodlands) (Figure 1). Montréal's nature parks are a network of conservation areas collectively covering 13.8 $km^2$ which were established in the 1980s to protect vital habitats and promote conservation and biodiversity protection (de Montréal 2013, 2018). These parks are managed through the Ecosystem and Landscape Management Program, which includes initiatives such as the removal of invasive species and directed tree planting of native tree species (de Montréal 2004). However, not all urban woodlands in Montréal are found within conservation areas. These additional woodlands, referred to here as non-status woodlands, are also key forested habitats within the city's boundaries yet lack an official conservation status. Non-status woodlands chosen for our study are part of city parks and are thus also managed by the City of Montréal and their respective boroughs. Although biodiversity conservation is a goal for all green spaces in Montréal, non-status woodlands are primarily managed for aesthetic and recreation purposes (de Montréal 2018). For example, management of these spaces may include the removal of shrubs and other understory species to make room for recreation spaces or for aesthetic reasons.

the summer of 2022, we selected 6 nature parks and 5 non-status woodlands to sample, within the constraints of a larger research project (Richmond et al. 2025). Using data obtained from the City of Montréal Open Data Portal (Ville de Montréal 2022), the boundaries of each park and the respective wooded areas were determined.

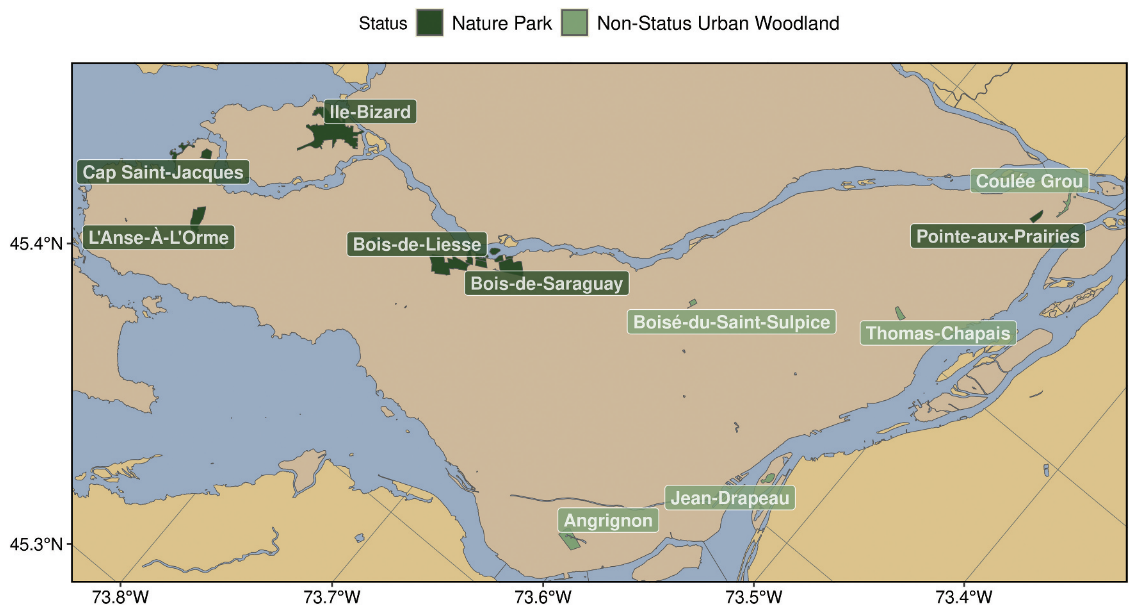


**Figure 1.** Location of the study area on the island of Montreal. The island is a beige colour, with rivers in blue surrounding it on both sides. Polygons showing the boundaries of 11 urban woodlands across the island, split into two categories. Dark green polygons represent nature parks with a conservation status, light green polygons represent non-status urban woodlands. Each park is labelled with the park's name.

All sites were previously forested and were not used for agricultural or industrial purposes immediately prior to being designated as a park – confirmed by City of Montréal archives – ensuring similarity in land-use history across sites. Nature parks were generally found in the suburban or peri-urban parts of the island, whereas non-status woodlands were typically located in more densely populated areas near the city center, although there are exceptions to this (Figure 1). We focused on differences caused by management in this study, as the soil conditions across the selected sites are similar (Paulauskas, *unpublished data*).

### *Sampling plot selection*

To capture variation across sites, we selected nine sampling plots per site in areas spanning three canopy classes: high canopy (>80%, $n = 3$), medium canopy (31–79%; $n = 3$), and low canopy (10–30%, $n = 3$) (see Richmond et al. for details). For some parks that were smaller or had homogeneous canopy cover, we could not obtain three sampling points in each canopy class. In those cases, we sampled the maximum number of plots per canopy class possible. If we arrived at points in the field and they were not accessible or did not represent their canopy class anymore (i.e., trees had been cut or added), we did not use that point and instead found an appropriate sampling plot in the field.

### *Sampling plot evaluation*

Once in the field, we constructed a 0.08-hectare square plot (800 m$^2$), which is larger than the standard 0.04 ha plot often used in forest inventories (i.e., iTree protocol). Multiple plot sizes are considered acceptable for urban forestry work, although it has been demonstrated that after 0.04 ha the sampling effort may outweigh the number of new species found (Nowak et al. 2008). Given that our parks were highly variable and we had a relatively small sample size, we opted for a larger plot to gain as much sampling coverage as possible. At the center of the plot, we recorded GPS location, structural complexity, and tree species and size. Here, we define structural complexity as the sum of the number of vegetation layers present in each plot. This includes the presence of an overstory layer, an understory layer, and a shrub layer (Canadian Forestry Inventory Committee 2008) (i.e., plots were assigned a '1' if one vegetation layer is present; '2' if two of the vegetation layers are present; or '3' if all vegetation layers are present). An overstory layer was defined as woody stems with foliage at the top of the canopy, while the understory layer was defined as woody stems with foliage underneath the top level of the canopy but over 2 m (i.e., larger than shrubs). Within each plot, each tree was identified to the species level, and the diameter at breast height (DBH, at 1.3 m) and canopy layer (i.e., overstory or understory) were recorded. We examined all woody stems over 2 m in height and with a DBH of ≥5.0 centimeters (cm).

### *Species composition*

Species composition was evaluated by categorizing tree species as native, non-native, and invasive. A native status was classified as a species whose native species range includes Quebec (Brouillet et al. 2024). Non-native species included trees that were not native to Quebec but were also not invasive. Finally, invasive species were defined as species that rapidly colonize new sites and who dominate the area once established (Lavoie et al. 2022). Invasive species classification was defined using Quebec-specific resources (Plant Dynamics Laboratory and Plant Biology Research Institute 2008; Lavoie and Ayotte 2019; Lavoie et al. 2022). Following the classification of species origins, we determined the proportion of invasive species per park. We did this by summing the number of invasive species across all sampling plots in a park and then dividing each site by its respective total species richness.

Due to the notably high abundance of particular invasive species, such as *Rhamnus cathartica* (Common Buckthorn), we also calculated the proportion of invasive stems (i.e., the proportion of individual trees that were invasive to Quebec) per park by summing the number of invasive trees per park then dividing each park by its respective total number of trees.

### *Canopy cover*

We calculated canopy cover for each site using 2021 LiDAR data (métropolitaine de Montréal 2021). Proportions of pixels containing canopy cover were calculated across the park boundary using the 'stars' and 'sf' packages (Pebesma et al. 2023a, 2023b) and then used to calculate the overall percent canopy cover.

### *Statistical analysis*

Although the number and size of plots were consistent across all woodlands, the parks varied significantly in their total area. As such, smaller parks were sampled more extensively than larger ones. To account for differences in sampling coverage, we used the iNEXT package in R to characterize species indices (species richness and Shannon diversity) for each park (Tc et al. 2016). Species rarefaction curves were created to account for the different sampling efforts across parks. Curves were visually analyzed to verify the urban woodland with the lowest sampling cover. Biodiversity indices were then calculated based on the lowest sampling coverage (Chao et al. 2014).

We used general linear models and beta regression to test whether forest composition and structure are associated with conservation status in urban woodlands. We made a separate model for each dependent variable; proportion of invasive species, proportion of invasive stems, species richness, Shannon diversity, structural complexity, and canopy cover. For species richness, Shannon diversity, and structural complexity, we used general linear models with independent variables including the park conservation status, size of the woodland, and proportion of invasive stems. For canopy cover, we used a beta regression model (`betareg` package; Zeileis et al. 2024) with independent variables including the park conservation status, size of the woodland, and proportion of invasive stems. For the proportion of invasive species and the proportion of invasive stem models, we used beta regression models with independent variables including park conservation status and size of the woodland. All numeric independent variables were scaled and centered before modeling. After building the models, we assessed residuals to ensure that assumptions of normality, independence, and heterogeneity were met. Given our small sample size, we use a p-value of 0.10 to indicate statistical significance. We performed all statistical analyses using R version 4.4.2 (The R Core Team and Contributors Worldwide 2024).

## Results

We sampled a total of 2,834 trees from 29 genera and 57 species across both nature parks and non-status woodlands. Of the 57 species we surveyed, 39 tree species were native to Quebec, 12 species were non-native but not invasive, and 6 species were invasive to the area (Table S1). Although invasive species accounted for only a small fraction of the total number of species (10%), the second most common species overall in terms of stem abundance was the invasive *Rhamnus cathartica* (Common Buckthorn). Other common species included the native *Acer saccharum* (Sugar Maple), *Ostrya virginiana* (American Hophornbeam), *Ulmus americana* (White Elm), and *Fraxinus pennsylvanica* (Green Ash).

### *Proportion of invasive species*

We examined the proportion of invasive species for each park and found that both types of woodlands fostered relatively low levels of invasive species, with the proportion of invasive species under 25% in all parks. Invasive species were present in all urban woodlands, regardless of conservation status. The proportion of invasive species was not statistically significantly related to conservation status (estimate = $-0.13$, SE = 0.27, $p = 0.63$, Table S6) or woodland size (estimate = 0.0, SE = $3 \times 10^{-3}$, $p = 0.27$, Table S6).

### *Proportion of invasive stems*

Overall, we identified 556 invasive individuals across both types of urban woodlands. Invasive species were found in all woodlands regardless of conservation status. However, the mean proportion of invasive stems varied between the nature parks and non-status woodlands (Figure 4a). Proportions of invasive stems ranged from 1% to 46% (mean = 13%) in urban woodlands and from 1% to 52% (mean = 21%) in nature parks. The proportion of invasive stems is not statistically significantly related to conservation status (estimate = 0.34, SE = 0.61, $p = 0.58$, Table S7); however, there is a higher average of invasive stems seen in the nature parks (Figure 4b). Woodland size is also not statistically significantly related to the proportion of invasive stems (estimate = 0.0, SE = 0.01, $p = 0.69$, Table S7)

### *Diversity indices*

After rarifying to account for sampling coverage, species richness per location ranged from 11.4 to 22.2 (mean = 17.2) in non-status woodlands, versus 13.1–29.7 (mean = 18.7) in nature parks. The mean Shannon diversity ranged from 9.9 to 19.6 species in non-status woodlands (mean = 15.0) and from 11.6 to 24.2 (mean = 15.6) in nature parks. Neither species richness nor Shannon diversity was statistically significantly different across conservation status (estimate = 2.7/1.9, SE = 4.4/3.5, $p = 0.56/0.59$, Tables S2-S3). Similarly, species richness and Shannon diversity were not significantly affected by woodland size (estimate = 0.2/-0.4, SE = 2.1/1.6, $p = 0.94/0.83$, Tables S2-S3), or the proportion of invasive species (estimate = −1.4/−1.1, SE = 2.2/1.8, $p = 0.56/0.53$, Tables S2-S3).

### *Canopy cover*

Canopy cover ranged from 36% to 86% (mean = 74%) in non-status woodlands and ranged from 75% to 98% (mean = 89%) in nature parks. Nature parks had statistically significantly higher levels of canopy cover (estimate = 1.4, SE = 0.57, $p = 0.01$, Figure 2, Table S4). The woodland size (estimate = −0.3, SE = 0.2, $p = 0.27$, Table S4) and proportion of invasive species (estimate = −0.2, SE = 0.3, $p = 0.38$, Table S4) did not have statistically significant relationships with canopy cover.

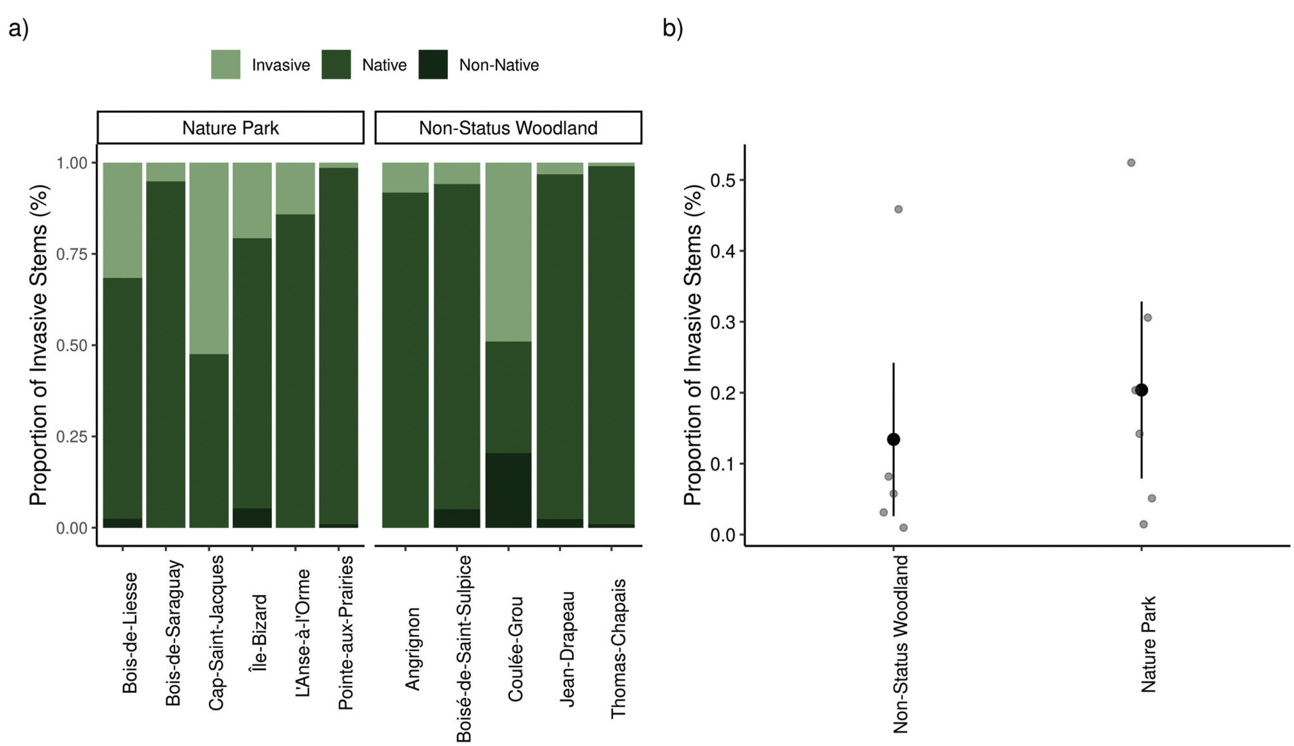


**Figure 2.** (a). The proportion of invasive stems for nature parks and non-status woodlands. Native is defined based on if the species' range includes Quebec (to see each species' designation see table S1). (b) Model predictive values for proportion of invasive stems as a function of conservation status (black dots indicate mean with 95% confidence intervals). Grey dots indicate raw data values for each park measured (see table S7 for model summary).

### Vegetative complexity

We found that structural complexity (i.e., the number of vegetation layers) was significantly positively related to conservation status (estimate = 0.5, SE = 0.2, $p = 0.09$, Figure 3, Table S5), but not to the woodland size (estimate = −0.03, SE = 0.1, $p = 0.79$, Table S5), or proportion of invasive species (estimate = 0.08, SE = 0.1, $p = 0.6$, Table S5).

## Discussion

We assessed whether a formal conservation status and associated management of Montréal's urban woodlands is related to biodiversity outcomes. Specifically, we examined the difference in forest structure and composition of urban woodlands designated as nature parks compared to those within the regular park system – such that all sites are open and accessible to the public and under the jurisdiction of the City of Montreal. We found no differences between park types with respect to proportion of invasive species, proportion of invasive stems, species richness, or Shannon diversity. Our hypotheses were partially supported, as nature parks had significantly higher levels of vegetative complexity and canopy cover compared to non-status woodlands.

We hypothesized that nature parks would have higher proportions of native trees, an important criterion for assessing urban biodiversity (Threlfall et al. 2017; Liu and Slik 2022). Native trees provide resources and habitat for native fauna and tend to foster higher levels of native species compared to non-native trees (Tallamy and Shropshire 2009, Narango et al. 2018). We expected

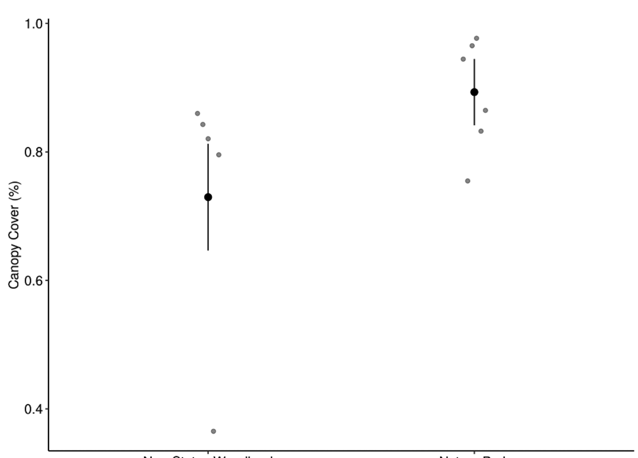


Figure 3. Model predictive values for percent canopy cover as a function of conservation status (black dots indicate mean with 95% confidence intervals). Grey dots indicate raw data values for each park measured (see table S4 for model summary).

that nature parks would have a lower proportion of invasive species, and in turn, a higher percentage of native species. Our results show, however, that the proportion of tree species native to Quebec was high (above 65%) in all but two urban woodlands (one nature park at 47% and one non-status woodland at 30%), regardless of conservation status. Our findings that even non-status urban woodlands are valuable green spaces for maintaining native tree species are consistent with other studies, which demonstrate that urban woodlands maintain a high percentage of native trees (Bouchard 1979; Domon et al. 1986; DeCandido et al. 2004; Pregitzer et al. 2019). Considering that the City of Montréal, like many cities, aims to increase the proportions of native tree species (de Montréal 2013), planning policies should consider the establishment of urban woodlands or other more natural green spaces to increase native biota (Pregitzer et al. 2019; Trentanovi et al. 2021). This aligns with an increasing trend towards urban conservation strategies such as Miyawaki forests (or 'tiny forests', which prioritize dense plantings of native tree species) in global cities (Lewis 2022).

We hypothesized that non-status woodlands would foster a higher proportion of invasive species and stems. We expected to see these differences due to differences in management priorities, as well as an assumption of greater human impact (e.g., through a higher density of visitors and/or more off-trail activity in the more centrally located and less-monitored non-status woodlands) (Hedblom and Söderström 2008). Specifically, the Ecosystem Management Program used in the nature parks is meant to actively monitor and remove invasive species (de Montréal 2013) thus presumably resulting in a lower prevalence of invasives. However, our results found (1) no noticeable difference between the proportion of invasive species in the nature parks and non-status woodlands (Table S6) and (2) a greater (although not statistically significantly so) proportion of invasive stems in the nature parks (Figure 4, Table S7).

The majority of invasive species were a minor component of the species composition; however, *Rhamnus cathartica* (Common Buckthorn) was found at high proportions across the nature parks. For example, at the study site with the highest proportion of *R. cathartica* stems (Cap-Saint-Jacques Nature Park), 52% of all individual trees sampled were *R. cathartica*, with other nature parks boasting high proportions of *R. cathartica* as well (e.g., 20% of stems at Île-Bizard Nature Park and 30% of stems at Bois-de-Leisse Nature Park; Table S1). This high proportion of *R. cathartica* was not observed in the non-status woodlands. For example, the site with the highest proportion of *R. cathartica* for the non-status woodlands,

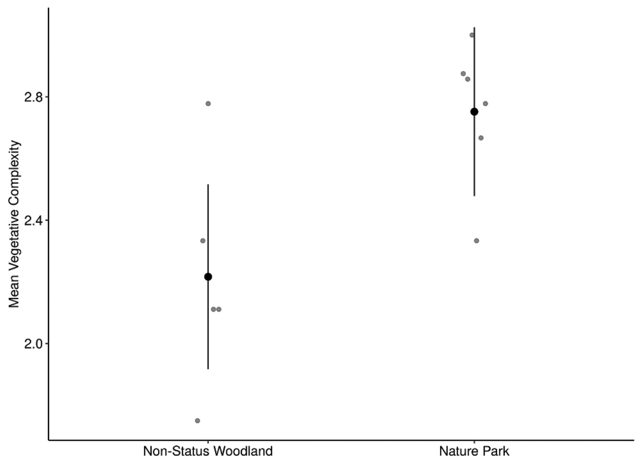


Figure 4. Model predictive values for mean vegetative complexity as a function of conservation status (black dots indicate mean with 95% confidence intervals). Grey dots indicate raw data values for each park measured (see table S5 for model summary).

reached only 6.9% of the total stems (Parc Angrignon, Figure 4a, Table S1).

The high proportion of invasive stems observed in nature parks may be reflective of the landscape surrounding the woodlands (Johnson et al. 2020). While all sites were forested prior to park establishment, the woodlands of the nature parks are often surrounded by abandoned agricultural fields which are frequently dominated by common buckthorn. This was historically used as a hedge species (Stover and Marks 1998; McCay et al. 2009) and may be facilitating *R. cathartica's* spread into nature parks. Alongside this historical influence, it may also be that the Ecosystem Management Program is insufficient – or insufficiently implemented – for tackling the removal of invasive trees given the large size of most nature parks and limited resources for invasive species management (Pluess et al. 2012). An underappreciated aspect of smaller, neighbourhood woodlands may be the feasibility of controlling invasive species and improving ecological quality through more intensive management of small areas (Johnson and Handel 2019) – particularly in cases where residents are actively involved in their care, as is often the case with local stewardship groups (e.g., 'friends-of park' groups) in more densely populated urban areas (Fors et al. 2018). A limitation of this study was that we were unable to measure the implementation of management plans. Future work incorporating both implementation and efficacy of management plans for a variety of greenspace types could improve our understanding of the role of conservation designations in urban spaces.

Our results suggest that the conservation potential of nature parks may be limited by the well-documented negative effects of *R. cathartica* in urban woodlands (Heneghan et al. 2004; Grady et al. 2012). The rapid growth of *R. cathartica* allows it to outcompete native tree species and establish dominance across woodlands (Knight et al. 2007). Once established, *R. cathartica* creates a dense canopy and has allelopathic abilities which further inhibits the establishment of potential native plant competitors (Seltzner and Eddy 2003; Warren et al. 2017). *R. cathartica* can also facilitate the establishment of additional invasive species (i.e., invasional meltdown, Heimpel et al. 2010). Our findings suggest that without significant intervention, the high prevalence of *R. cathartica* may shift local plant community composition and ultimately limit biodiversity conservation in parks where common buckthorn is abundant, i.e., in nature parks).

Our results showed no significant differences in mean species richness and Shannon diversity. These results highlight that urban woodlands support similar levels of woody plant richness regardless of conservation status. These results are supported by previous studies that have examined the relationship between urban woodlands and tree species richness. Non-status urban woodlands have been established as key spots for woody plant species richness across the globe, regardless of conservation status (DeCandido et al. 2004; Cornelis and Hermy 2004; Korhonen et al. 2020). Furthermore, in Quebec riparian areas, the presence of intact forest rather than a protected area designation influences the quantity and quality of multiple ecosystem services and tree biodiversity (Hanna et al. 2020). Such findings suggest that while protection can be an important mechanism to conserve a particular land-cover class (i.e., forest), good conservation outcomes are not necessarily limited to formally protected areas. Non-status spaces, under appropriate management, can also foster high levels of species richness (Cox et al. 2011; Rayner et al. 2014). This presents an opportunity for urban planners and decision-makers to prioritize and value non-status woodlands for biodiversity planning alongside conservation-oriented parks.

In addition to differences in species composition, species richness, and Shannon diversity, we expected to see differences in forest structure, with nature parks having higher canopy cover and structural complexity. Our hypotheses were supported, with nature parks on average having both higher canopy cover and higher structural complexity. Differences in forest structure are likely reflective of management strategies between nature parks and non-status woodlands. The Ecosystem Management Program used in these parks involves minimal intervention to maximize biodiversity conservation (de Montréal

2013. Consequently, dead wood and shrub and understory layers are retained unless they present a safety concern to visitors or overgrow paths (de Montréal 2004). In contrast, non-status woodlands are primarily managed for recreation. As such, shrubs, understory layers, and trees contributing to the canopy are often removed for either aesthetic or recreational purposes (de Montréal 2018). We intentionally selected urban woodlands that had similar historical legacies (i.e., preserved forest remnants as opposed to agricultural land), further emphasizing these differences as a result of current management practices, not historical differences.

Maintaining high structural complexity (i.e., the presence of a shrub, understory and overstory layer) is ecologically important. Structurally complex vegetation layers serve as habitat and an important refuge for specialist fauna (Hyde et al. 2006; Remeš et al. 2022). For example, Threlfall et al. (2017) showed that increasing understory volume in urban green spaces by 10–30% resulted in a 30–120% increase in bird, bat, and beetle proportions. Thus, although we found that nature parks and non-status woodlands support similar levels of tree species richness and Shannon diversity, these results may not hold for herbaceous vegetation (DeCandido et al. 2004; Bergeron et al. 2019). Given the increased structural complexity, further investigation is warranted to see if nature parks are supporting higher levels of native species, species richness, and Shannon diversity in the herbaceous understory. If one of the goals of non-status woodlands is to maintain structural complexity, adopting strategies from the Ecosystem Management Program, which promotes the retention of the shrub and understory layers, may help to achieve that goal.

However, the increased structural complexity in nature parks is most likely being created in part by the high abundance of *R. cathartica* stems. Given this connection, further investigation is needed to determine if the high proportion *R. cathartica* are limiting the benefits provided by high structural complexity in nature parks. For example, while the negative effects of *R. cathartica* have been well documented (Knight et al. 2007; Warren et al. 2017), less is known on the potential benefits of common buckthorn. For instance, studies conducted in woodlands in Illinois and Montreal found that many bird species, including important native species such as the *Cardinalis cardinalis* (northern cardinal) and *Turdus migratorius* (American robin), nested in *R. cathartica* (Whelan and Dilger 1992; Cull et al. 2024). The high prevalence of birds nesting in invasive woody species was due to the lack of native shrub alternatives (Whelan and Dilger 1992).

As urban parks continue to tackle the removal of invasive species, it is crucial that management considers the holistic implications of removing such abundant species, which may potentially be providing habitat – albeit habitat that could potentially be improved by replacement with native shrub species (Gaertner et al. 2016). Although replacing invasive shrubs with native alternatives does not necessarily improve structural complexity, it will increase habitat quality by offering more required resources such as food and result in higher abundance and species richness of native fauna (Tallamy and Shropshire 2009, Narango et al. 2018). Future studies should also assess the abundance and species richness of multiple taxa and trophic levels to determine the extent to which differences in both the presence and composition of a shrub layer drives urban biodiversity more broadly.

Our study helps bridge the gap in the lack of information regarding urban woodland management and the role that urban conservation areas have for safeguarding biodiversity. The results of our study provide evidence that urban woodlands foster high levels of native tree species, regardless of conservation status. While a conservation status may not be necessary to preserve tree species richness or proportion of native species, the less intensive management strategy used in the nature parks may be important for fostering high levels of canopy cover and structural complexity. Despite the known benefits that structural complexity has for supporting fauna species, our conclusions are limited because we did not identify which species are making up the shrub and understory layer. Based on the high proportion of *R. cathartica* (common buckthorn) found in the nature parks, however, we can assume that this invasive species contributes to the high structural complexity and at times, canopy cover. Future research is needed to investigate if the potential benefits received from common buckthorn, in terms of providing structural complexity and wildlife habitat, outweigh the known negative effects.

## Acknowledgments

We gratefully acknowledge Michael Paulauskas, Laura Carolina Gonzalez Sinisterra, Adela Bartova, Angelina Teichrib, and Gabrielle Gingras for their field support. We would also like to thank the cities of Montreal, Beaconsfield, and Bai-d'Urfé for their logistical support. Thank you to two anonymous reviewers and members of the Ziter Urban Landscape Ecology lab for their editing and comments that greatly improved the manuscript.

## Disclosure statement

No potential conflict of interest was reported by the author(s).

## Funding

This research was funded by the Natural Sciences and Engineering Research Council of Canada (CGSD3-559539-2021 to ICR and Discovery Grant RGPIN-2019-07010 to CDZ); UFOR NSERC CREATE scholarship awarded to EP.

## ORCID

Isabella C Richmond http://orcid.org/0000-0001-5079-8442

## Data availability statement

All data and code for sampling, data cleaning, and analysis can be found at https://github.com/zule-lab/UrbanWoodlandConservation. archived at https://zenodo.org/records/12774017.

## References

Ballantyne M, Pickering CM. 2015. Differences in the impacts of formal and informal recreational trails on urban forest loss and tree structure. J Environ Manag. 159:94–105. doi: 10.1016/j.jenvman.2015.05.007.

Beauvais M-P, Pellerin S, Lavoie C. 2016. Beta diversity declines while native plant species richness triples over 35 years in a suburban protected area. Biol Conserv. 195:73–81. doi: 10.1016/j.biocon.2015.12.040.

Bergeron A, Lavoie C, Domon G, Pellerin S, Hermy M. 2019. Changes in spatial structures of plant communities lead to functional homogenization in an urban forest park. Appl Veg Sci. 22(2):256–268. doi: 10.1111/avsc.12417.

Bouchard A. 1979. La végétation forestière du cap Saint-Jacques, île de Montréal, Québec. Can J Bot. 57 (11):1191–1202. doi: 10.1139/b79-143.

Bourdages M, Lavoie C. 2011. Plant introduction and extirpation in a small island park: natural and anthropogenic rates. Écoscience. 18(2):89–97. doi: 10.2980/18-2-3388.

Brouillet L, Coursol F, Meades S, Favreau M, Anions M. 2024. Database of vascular plants of Canada (VASCAN). https://data.canadensys.net/vascan/search.

Canadian Forestry Inventory Committee. 2008. Canada's national forest inventory ground sampling guidelines. Ottawa: Canadian Forestry Inventory Committee.

Chao A, Gotelli NJ, Hsieh TC, Sander EL, Ma KH, Colwell RK, Ellison AM. 2014. Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies. Ecol Monogr. 84(1):45–67. doi: 10.1890/13-0133.1.

Communauté métropolitaine de Montréal. 2021. Indice canopée métropolitain. Montreal: Communauté métropolitaine de Montréal.

Cornelis J, Hermy M. 2004. Biodiversity relationships in urban and suburban parks in flanders. Landscape Urban Plan. 69 (4):385–401. doi: 10.1016/j.landurbplan.2003.10.038.

Cox RL, Underwood EC, Fenton B. 2011. The importance of conserving biodiversity outside of protected areas in Mediterranean ecosystems. PLoS One. 6(1):e14508. doi: 10.1371/journal.pone.0014508.

Croci S, Butet A, Georges A, Aguejdad R, Clergeau P. 2008. Small urban woodlands as biodiversity conservation hot-spot: a multi-taxon approach. Landscape Ecol. 23:1171–1186.

Cull CA, Guest MJ, Frei B, et al. 2025. Human recreational activity does not influence open cup avian nest survival in urban green spaces. Urban Ecosyst. 28(24). doi: https://doi.org/10.1007/s11252-024-01669-0.

DeCandido R, Muir AA, Gargiullo MB. 2004. A first approximation of the historical and extant vascular flora of New York City: implications for native plant species conservation. J Torrey Bot Soc. 131(3):243–251. doi: 10.2307/4126954.

Domon G, Bouchard A, Bergeron Y, Gauvin C. 1986. La répartition et la dynamique des principales espèces arborescentes du Bois-de-Saraguay, Montréal (Québec). Can J Bot. 64(5):1027–1038. doi: 10.1139/b86-140.

Fors H, Jansson M, Nielsen AB. 2018. The impact of resident participation on urban woodland quality—A case study of Sletten, Denmark. Forests. 9(11):670. doi: 10.3390/f9110670 .

Gaertner M, Larson BMH, Irlich UM, Holmes PM, Stafford L, van Wilgen Bw, Richardson DM, van Wilgen BW. 2016. Managing invasive species in cities: a framework from Cape Town, South Africa. Landscape Urban Plan. 151:1–9. doi: 10.1016/j.landurbplan.2016.03.010.

Gaston KJ, Jackson SF, Cantú-Salazar L, Cruz-Piñón G. 2008. The ecological performance of protected areas. Annu Rev Ecol Evol Syst. 39(1):93–113. doi: 10.1146/annurev.ecolsys.39.110707.173529.

Government of Canada and Statistics Canada. 2023. Census of population - Montréal, territoire équivalent (TÉ) [Census division]. In: Quebec. Profile table, statistics canada catalogue no. 98-316-X2021001. Ottawa.

Grady KC, Hennegha L, Jabon D. 2012. Reduced herbivory on exotic buckthorn (*rhamnus cathartica*) compared to native tree species in an Illinois woodland. Biological notes no. 143. Illinois natural history survey. Prairie Research Institute, University of Illinois at Urbana-
Champaign. Champaign, Illinois: Illinois Natural History Survey.

Hanna DEL, Roux DJ, Currie B, Bennett EM. 2020. Identifying pathways to reduce discrepancies between desired and provided ecosystem services. Ecosystem Serv. 43:101119. doi: 10.1016/j.ecoser.2020.101119.

Hedblom M, Söderström B. 2008. Woodlands across Swedish urban gradients: status, structure and management implications. Landscape Urban Plan. 84(1):62–73. doi: 10.1016/j.landurbplan.2007.06.007.

Heimpel GE, Frelich LE, Landis DA, Hopper KR, Hoelmer KA, Sezen Z, Asplen MK, Wu K. 2010. European buckthorn and Asian soybean aphid as components of an extensive invasional meltdown in North America. Biol Invasions. 12 (9):2913–2931. doi: 10.1007/s10530-010-9736-5.

Heneghan L, Rauschenberg C, Fatemi F, Workman M. 2004. European buckthorn (*Rhamnus cathartica*) and its effects on some ecosystem properties in an urban woodland. Ecol Restor. 22(4):275–280. doi: 10.3368/er.22.4.275.

Hsieh TC, Ma KH, Chao A. 2016. iNEXT: an R package for rarefaction and extrapolation of species diversity (Hill numbers). Methods Ecol Evol. 7(12):1451–1456. doi: 10.1111/2041-210X.12613.

Hyde P, Dubayah R, Walker W, Blair JB, Hofton M, Hunsaker C. 2006. Mapping forest structure for wildlife habitat analysis using multi-sensor (LiDAR, SAR/InSAR, ETM+, quickbird) synergy. Remote Sens Of Environ. 102(1–2):63–73. doi: 10.1016/j.rse.2006.01.021.

Ioja C, Breuste J. 2020. Urban protected areas and urban biodiversity. In: Breuste J, Artmann M, Ioja C Qureshi S, editors. Making green cities: concepts, challenges and practice. (NY): Springer; p. 289–398.

Johnson LR, Handel SN. 2019. Management intensity steers the long-term fate of ecological restoration in urban woodlands. Urban Urban Gree. 41:85–92. doi: 10.1016/j.ufug.2019.02.008.

Johnson LR, Trammell TLE, Bishop TJ, Barth J, Drzyzga S, Jantz C. 2020. Squeezed from all sides: urbanization, invasive species, and climate change threaten riparian forest buffers. Sustainability. 12(4):1448. doi: 10.3390/su12041448.

Kabisch N, Frantzeskaki N, Hansen R. 2022. Principles for urban nature-based solutions. AMBIO: A J Hum Environ. 51 (6):1388–1401. doi: 10.1007/s13280-021-01685-w.

Knight KS, Kurylo JS, Endress AG, Stewart JR, Reich PB. 2007. Ecology and ecosystem impacts of common buckthorn (*rhamnus cathartica*): a review. Biol Invasions. 9(8):925–937. doi: 10.1007/s10530-007-9091-3.

Korhonen A, Siitonen J, Kotze DJ, Immonen A, Hamberg L. 2020. Stand characteristics and dead wood in urban forests: potential biodiversity hotspots in managed boreal landscapes. Landscape Urban Plan. 201:103855. doi: 10.1016/j.landurbplan.2020.103855.

Kowarik I, Körner S, eds. 2005. Wild urban woodlands: new perspectives for urban forestry. (NY): Springer.

Lavoie C, Ayotte G. 2019. 50 plantes envahissantes : protéger la nature et l'agriculture. Québec: Publications du Québec.

Lavoie C, Groeneveld E, Brisson J, Dumont B, Pellerin S, Proulx R, Villard M-A. 2022. 40 autres plantes envahissantes: protéger la nature aujourd'hui et demain. Québec: Publications du Québec.

Leroux SJ, Krawchuk MA, Schmiegelow F, Cumming SG, Lisgo K, Anderson LG, Petkova M. 2010. Global protected areas and IUCN designations: do the categories match the conditions? Biol Conserv. 143(3):609–616. doi: 10.1016/j.biocon.2009.11.018.

Lewis H. 2022. Mini-forest revolution: using the Miyawaki method to rapidly rewild the world. White River Junction: Chelsea Green Publishing.

Lintott PR, Bunnefeld N, Fuentes-Montemayor E, Minderman J, Blackmore LM, Goulson D, Park KJ. 2014. Moth species richness, abundance and diversity in fragmented urban woodlands: implications for conservation and management strategies. Biodivers Conserv. 23(11):2875–2901. doi: 10.1007/s10531-014-0753-z.

Liu J, Slik F. 2022. Are street trees friendly to biodiversity? Landscape Urban Plan. 218:104304. doi: 10.1016/j.landurbplan.2021.104304.

Lupp G, Forster B, Kantelberg V, Markmann T, Naumann J, Honert C, Koch M, Pauleit S. 2016. Assessing the recreation value of urban woodland using the ecosystem service approach in two forests in the Munich metropolitan region. Sustainability. 8(11):1156. doi: 10.3390/su8111156.

McCay TS, McCay DH, Caragiulo AV, Mandel TL. 2009. Demography and distribution of the invasive Rhamnus cathartica in habitats of a fragmented landscape 1. J Torrey Bot Soc. 136(1):110–121. doi: 10.3159/08-RA-041R.1.

McDonald RI, Mansur AV, Ascensão F, Colbert M, Crossman K, Elmqvist T, Gonzalez A, Güneralp B, Haase D, Hamann M, et al. 2020. Research gaps in knowledge of the impact of urban growth on biodiversity. Nat Sust. 3(1):16–24. doi: 10.1038/s41893-019-0436-6.

Miller JR, Hobbs RJ. 2002. Conservation where people live and work. Conserv Biol. 16(2):330–337. doi: 10.1046/j.1523-1739.2002.00420.x.

Narango DL, Tallamy DW, Marra PP. 2018. Nonnative plants reduce population growth of an insectivorous bird. Proc Natl Acad Sci USA. 115(45):11549–11554. doi: 10.1073/pnas.1809259115.

Noël S, Lapointe F-J. 2010. Urban conservation genetics: study of a terrestrial salamander in the city. Biol Conserv. 143 (11):2823–2831. doi: 10.1016/j.biocon.2010.07.033.

Nowak DJ. 2007. Assessing urban forest effects and values: new York City's urban forest. Forest Service, Northern Research Station: Madison: U.S. Department of Agriculture.

Nowak D, Walton J, Stevens J, Crane D, Hoehn R. 2008. Effect of plot and sample size on timing and precision of urban forest assessments. Arboriculture Urban For. 34(6):386–390. doi: 10.48044/jauf.2008.052.

Otieno NE, Mutati A. 2021. Bird alpha, beta and functional diversities across three peri-urban woodland stands along an anthropogenic disturbance gradient: is formal protection a guarantee for ecological integrity? Global Ecol Conserv. 25: e01410. doi: 10.1016/j.gecco.2020.e01410.

Parks Canada Agency. 2023 Apr 6. Creating new national urban parks in Canada. Parks Canada: Ottawa: Government of Canada.

Pauleit S, Jones N, Nyhuus S, Pirnat J, Salbitano F. 2005. Urban forest resources in European cities. In: Konijnendijk C, Nilsson K, Randrup T Schipperijn J, editors. Urban forests and trees. (NY): Springer; p. 49–80.

Pebesma E, Bivand R, Racine E, Sumner M, Cook I, Keitt T, Lovelace R, Wickham H, Ooms J, Müller K, et al. 2023a July 11. sf: Simple Features for R.

Pebesma E, Sumner M, Racine E, Fantini A, Blodgett D. 2023b Sep 11. Stars: Spatiotemporal Arrays, Raster and Vector Data Cubes.

Plant Dynamics Laboratory and Plant Biology Research Institute. 2008. The management of invasive alien plants on Mount Royal. Montreal: Université de Montréal.

Pluess T, Jarošík V, Pyšek P, Cannon R, Pergl J, Breukers A, Bacher S, Adler FR. 2012. Which factors affect the success or failure of eradication campaigns against alien species? PLoS One. 7(10):e48157. doi: 10.1371/journal.pone.0048157.

Pn D, Lehvävirta S, Ab N, Sa T. 2017. Urban woodlands and their management. In: Ferrini F, Konijnendijk van den Bosch C Fini A, editors. Routledge handbook of urban forestry. (NY): Routledge; p. 515–228.

Pregitzer CC, Bradford MA. 2023. Associations between recent land use history and urban forest composition. Urban Urban Gree. 90:128134. doi: 10.1016/j.ufug.2023.128134.

Pregitzer CC, Charlop-Powers S, Bibbo S, Forgione HM, Gunther B, Hallett RA, Bradford MA. 2019. A city-scale assessment reveals that native forest types and overstory species dominate New York city forests. Ecol Appl. 29(1):e01819. doi: 10.1002/eap.1819.

Ramalho CE, Laliberté E, Poot P, Hobbs RJ. 2014. Complex effects of fragmentation on remnant woodland plant communities of a rapidly urbanizing biodiversity hotspot. Ecology. 95(9):2466–2478. doi: 10.1890/13-1239.1.

Rayner L, Lindenmayer DB, Wood JT, Gibbons P, Manning AD. 2014. Are protected areas maintaining bird diversity? Ecography. 37(1):43–53. doi: 10.1111/j.1600-0587.2013.00388.x.

The R Core Team & Contributors Worldwide. 2024. R: The R Stats Package.

Remeš V, L H, Matysioková B, L R, Remešová E. 2022. Vegetation complexity and pool size predict species richness of forest birds. Front Ecol Evol. 10:964180. doi: 10.3389/fevo.2022.964180.

Richmond IC, Paulauskas MA, Padvaiskas E, Sinisterra LCG, Hutt-Taylor K, Robitaille AL, Ziter CD 2025. Land-use history causes differences in park nighttime cooling capacity and forest structure bioRxiv. 637130. doi: 10.1101/2025.02.13.637130.

Seltzner S, Eddy T. 2003. Allelopathy in *Rhamnus Cathartica*. Eur Buckthorn Mich Botanist. 42:51–61.

Singapore Government. 2024. Singapore green plan 2030. Singapore: Singapore Government.

Stover ME, Marks PL. 1998. Successional vegetation on abandoned cultivated and pastured land in Tompkins County, New York. J Torrey Bot Soc. 125(2):150–164. doi: 10.2307/2997302.

Tallamy DW, Shropshire KJ. 2009. Ranking lepidopteran use of native versus introduced plants. Conserv Biol. 23 (4):941–947. doi: 10.1111/j.1523-1739.2009.01202.x.

Threlfall CG, Mata L, Mackie JA, Hahs AK, Stork NE, Williams NSG, Livesley SJ, Beggs J. 2017. Increasing biodiversity in urban green spaces through simple vegetation interventions. J Appl Ecol. 54(6):1874–1883. doi: 10.1111/1365-2664.12876.

Tree Canada. 2018. Canadian urban forest strategy 2019-2024. Ottawa: Tree Canada.

Trentanovi G, Campagnaro T, Kowarik I, Munafò M, Semenzato P, Sitzia T. 2021. Integrating spontaneous urban woodlands into the green infrastructure: unexploited opportunities for urban regeneration. Land Use Pol. 102:105221. doi: 10.1016/j.landusepol.2020.105221.

Trzyna T, Jt E, Hyman G, McNeely JA, da Cunhae Menezes P, Myrdal B, Phillips A. 2014. Urban protected areas : profiles and best practice guidelines. Gland: IUCN.

United Nations Sustainable Development. 2015. Cities - united nations sustainable development action 2015.

Vieira J, Matos P, Mexia T, Silva P, Lopes N, Freitas C, Correia O, Santos-Reis M, Branquinho C, Pinho P. 2018. Green spaces are not all the same for the provision of air purification and climate regulation services: the case of urban parks. Environ Res. 160:306–313. doi: 10.1016/j.envres.2017.10.006.

Ville de Montréal. 2004. Politique de protection et de mise en valeur des milieux naturels. Montreal: Ville de Montréal.

Ville de Montréal. 2013. Rapport sur la biodiversité. Montreal: Ville de Montréal.

Ville de Montréal. 2018. Plan directeur du sport er du plein air urbains. Montreal: Ville de Montréal.

Ville de Montréal. 2022. Grands parcs, parcs d'arrondissements et espaces publics. Montreal: Ville de Montréal, Données Ouvertes Montréal.

Warren RJ, Labatore A, Candeias M. 2017. Allelopathic invasive tree (*rhamnus cathartica*) alters native plant communities. Plant Ecol. 218(10):1233–1241. doi: 10.1007/s11258-017-0766-2.

Whelan CJ, Dilger ML. 1992. Invasive, exotic shrubs: a paradox for natural area managers? Nat Areas J. 12:109–110.

Xu S, Xu K, Zou G, Yan J, Peng Z, Zhang W, Zhang Y, Han Y, Wang J, Chang J. 2021. Density management strategy for overstory and understory of urban woodland based on ecological size-density allometry. Urban Urban Gree. 66:127379. doi: 10.1016/j.ufug.2021.127379.

Yao W, Zhang X, Gong Q. 2021. The effect of exposure to the natural environment on stress reduction: a meta-analysis. Urban Urban Gree. 57:126932. doi: 10.1016/j.ufug.2020.126932.

Zeileis A, Cribari-Neto F, Grün B, Kosmidis I, A. B. S. (earlier version by) & A. V. R. (earlier version by). Betareg: beta regression. [2024 July 7].