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Characterization and Justification of Trees on an Inner-City Golf Course in halifax, Canada: An Investigation into the Ecological Integrity of Institutional Greenspace

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Abstract: Institutional greenspaces such as golf courses, cemeteries, military bases, hospitals, and university campuses are not generally revered for their ecological integrity. The existence of golf courses in particular has been heavily debated due to widespread perceptions of these spaces as environmentally degrading. Though much of the total area of golf courses is occupied by heavily manicured lawns, Canadian golf courses tend to be well treed and thus show significant potential to enhance forest coverage and contribute to the conservation of native tree species when established on previously unforested land. To explore this potential, a tree inventory was carried out on an inner-city golf course in halifax, Nova Scotia, and findings compared to an earlier inventory of more naturalized (i.e., ingrowth) forest areas in the same city. Based in the Acadian Forest Region, this case study used the characteristics of a healthy and mature Acadian Forest as a model for ecological integrity. It was found that both the golf course and the ingrowth populations were largely representative of a mixedwood Acadian forest. Likewise, both populations were in a similar stage of regeneration and exhibited similar stresses. These results suggest that if improved forest management approaches are employed, golf courses will effectively strengthen the ecological integrity of urban forests. This is an especially important finding in the climate change era when tree populations are likely to be subjected to new environmental stressors which may be alleviated via the human intervention that is available on managed lands such as institutional greenspaces.

Keywords: institutional greenspace; urban forest; improved forest management; golf course management; conservation; Acadian forest; ecological integrity; climate change

1. Introduction

In an increasingly urbanizing world, it seems reasonable to question whether urban forests can be characterized as functionally healthy ecosystems. Although street trees are clearly botanic lifeforms, their neat planting in intentional rows distinguishes them from their cousins growing in more natural settings. Urban ecologists and foresters, however, argue that urban ecosystems can hold great ecological value, especially due to globally increasing urbanism [1–3]. While the ecosystem services provided by street trees, such as rainwater filtration, storm-water management, carbon capture, and amelioration of the urban-heat-island effect [4] have been well documented, it is less clear how trees growing on institutional greenspaces—which in this context is defined as mixed-use, managed greenspaces that exist for purposes other than conservation, such as golf courses, cemeteries, military bases, hospitals, university campuses, etc.—might contribute to the health and vitality of urban ecosystems. Rather,

such landscapes, golf courses especially, are often condemned as being environmentally degraded, and are rarely perceived as ecologically valuable spaces [5].

This condemnation is not without cause, as common management regimes on golf courses have been found to have detrimental effects on vital ecosystem components, particularly water and soil [6]. For example, studies have shown that the chemical use required to sustain uniform, green turf, particularly nitrogen-rich fertilizers, often results in decreased surface and groundwater quality, eutrophic downstream waters, and contaminated soils [7–9]. Furthermore, the use of heavy machinery and the common practice of "top dressing" in course maintenance can significantly alter soil structure [6,10]. It is important to note, however, that these criticisms are typically not of golf courses themselves, but of the management practices used to maintain them. This is representative of a need for more research investigating and promoting improved golf course management regimes prioritizing enhanced ecological function [6].

Thus, this case study used an urban forestry lens to investigate the ecological value of golf courses in urban environments and the management conditions required to maximize this value. It compared the ecological integrity of forest patches found on an inner-city golf course, Old Ashburn Golf Course (OAGC), in halifax Regional Municipality (HRM), Nova Scotia (NS) (see Figure 1), to that of more-naturalized areas in the same city. Ecological integrity has been defined by Ordóñez and Duinker [2] as a combination of ecosystem health, native species representation, resilience to stressors, and capacity for self-maintenance, while naturalized is defined by Toni and Duinker [4] as the ecosystem condition brought about through a transition of an urban space toward compositional, structural, and functional similarity to local non-urban forests considered 'natural'.
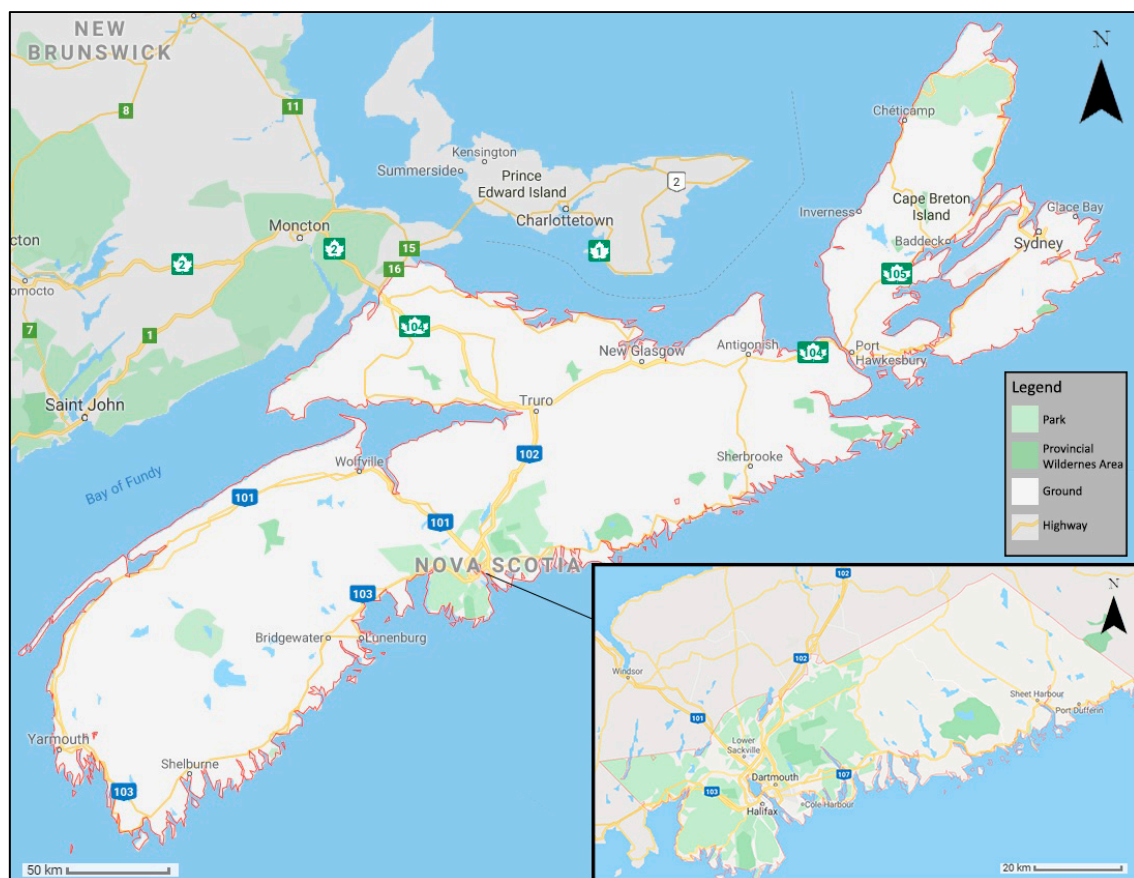

Figure 1. Location of NS on the eastern coast of North America. Land or ground is defined by white, while the darker green indicates a provincial wilderness area and the lighter green represents parkland. Inset Map: Detail map indicating location of the HRM within the province of NS. Source: ©2019 Google.

To perform this comparison, data collected during a near 7000-stem tree inventory on OAGC in 2017 was analyzed against unpublished HRM urban forest data collected by Foster and Witherspoon in 2016 [11]. Since NS is in the Acadian Forest Region [12,13], forested landscapes across the province are mostly characterized by native Acadian species. Thus, the characteristics of a healthy, mature, multi-successional Acadian forest, as described by Rowe in 1972 [12] and Loo and Ives in 2003 [13], have been used to model ecological integrity for the purposes of this study. Both the forest patches on the golf course and more-naturalized patches found elsewhere in the HRM were measured against this model based on species composition, tree condition, and age-class to determine whether patches growing on managed landscapes were in better, worse, or similar condition to those growing in areas with no human intervention. This study was conducted to demonstrate that if the forests growing on institutional landscapes are in similar or better condition than those growing without human intervention, then these managed landscapes have potential to make a significant contribution to the ecological integrity of urban forests. Recognizing the significant ecological value trees bring to the "golfscape" and the larger urban ecosystem, this study focused exclusively on golf course trees. It seeks to contribute to the growing canon of literature [14–17] pointing to the ecosystem services rendered from institutional greenspaces in urban settings with the aim of encouraging improvement in the existing management regimes on golf courses. It also endeavors to fill a gap in the existing literature investigating the role of trees in enhancing ecological integrity of golf courses in the Canadian context.

2. Literature Review

It is estimated that there are about 25,000 golf courses worldwide [18], which occupy approximately 1.5 million hectares of land [19]. In Canada, as of 2015, a land area of 110,166 ha has been occupied by golf course infrastructure extending from the Pacific coast to the Atlantic [20]. Although its exact history is unknown, it is hypothesized that the inaugural game of golf was played in the 13th century, though its current incarnation, as a game played across sprawling swaths of greenspace, did not appear until 1774 [18]. Such as it is, the four playing-area types—greens and tees, fairways, driving range/practice areas, and roughs—occupy an approximate average area of 2.4 ha (3.9%), 12 ha (20%), 2.8 ha (4.6%), and 21 ha (34%) of land, respectively, per 18-hole course [21]. As the former three play areas are far more intensely managed than the latter, and the average land area of a golf course is roughly 61 ha [19,21], only 28% of an individual golf course is subject to intensive human intervention [21]. This means that once pathways, waterbodies, and gray infrastructure (i.e., parking spaces and buildings) are considered, within the boundaries of most golf courses it is reasonable to estimate that at least half of the land is greenspace subject to low intensities of management intervention, such as forest. While there are exceptions to these approximations, such as courses constructed in desert landscapes [22,23], the value of well-treed courses as ecologically significant spaces has been well documented over the past two decades [21,24–26].

Despite their water, herbicide, and pesticide usage [21], in inner-city environments, where greenspace tends to be fragmented or degraded by installations of built infrastructure, well-managed golf courses can hold significant ecological value [27]. Yet, the environmental impact of golf courses is often a polarizing debate, primarily between those who golf and those who do not. This was demonstrated in 2002, when an English research team distributed a survey to investigate public opinion about whether golf courses are beneficial to the environment. Among golfers, 80% responded in the affirmative, largely citing habitat preservation as the reason for their answer. Conversely, non-golfers expressed the exact opposite opinion, with 64% responding negatively due to perceived destruction of habitat [5]. Since that study's publication, however, extensive research has demonstrated the ecological benefits of golf courses, especially in urban settings [16,17,25,26,28–30].

Golf courses can play a vital role in enhancing biodiversity by providing a refuge for flora and fauna and improving connectivity between greenspaces [29]. Terman supports this point by identifying small naturalized ecosystems, such as those found on golf courses, as "habitat sinks" [24] (p. 192), meaning that these smaller habitats often support the mating activities of migrating populations. This is

especially true for bird species, such as the Canada goose (*Branta canadensis*), and wetland fauna for which golf courses have been identified as significant habitat [28,31]. In fact, it has been found that golf course wetlands often provide habitat for rare and threatened amphibian and macroinvertebrate species, some of which cannot be found in off-course wetlands [31]. Though arguments have been made that chemical contamination of golf course wetlands could put species at risk [32], a 2017 study showed that minor management improvements, such as adjusting fertilization timelines, would significantly reduce threats of negative impacts [33]. Indeed, swelling populations of amphibian, bird, and bat species have been correlated to minor management improvements such as supporting natural regeneration of native vegetation [29,34]. While some studies have hypothesized that strong species representation on golf courses is either due to the resilience of urban-adaptable species or their habitual behavior, it is generally conceded that, in any case, most species tend to do better on golf courses than in residential developments, such as suburbs [27,31]. Despite these assertions, it may seem unlikely that institutional greenspaces, with their heavily humanized footprints, can possess the same degree of ecological integrity as undisturbed hinterland.

Yet, increasingly, institutional urban greenspaces are being found to be at least as ecologically valuable as urban park space. For example, one Australian study found that golf courses tended to have higher proportions of native plant species than parks [15]. Furthermore, it was discovered that golf courses usually had more big trees and higher concentrations of dead wood, an important component to forest ecosystem dynamics. Other institutional greens spaces, cemeteries for example, have been found to deliver significant regulating ecosystem services, such as filtration of air and stormwater pollutants and increasing pollination through the provision of insect habitat, in addition to a litany of social services [14].

Furthermore, in a 2009 study measuring "ecological value" based on "a quantitative synthesis of studies in the scientific literature that have measured and compared biota on golf courses to that of biota in green area habitats related to other land uses" [30] (p. 191), it was found that golf courses have 64% more ecological value than other nearby land uses; furthermore, as surrounding landscapes become increasingly altered by anthropogenic activity, such as urbanization, this relative value tends to increase. Moreover, 63% of the courses studied showed an ecological integrity equal to, or exceeding, that of natural protected areas based on the levels and types of biodiversity present. Though these results are highly illuminating, it is important to emphasize the relativity associated with the ecological values. That is, if a wild forest is clear cut to allow for the construction of a golf course, the ecological value of the resulting course will be significantly lower than that of the former forest. Conversely, if a course is constructed on a former parking lot or agricultural field, it is likely to improve the ecological integrity of the site. Thus, since golf courses have historically been constructed on agricultural land as opposed to productive and ecologically diverse hinterland, their development has often led to a relatively improved degree of biodiversity [24,29]. This is indeed the case at the OAGC, which was constructed in 1922 on a Stanley Thompson design. He was a revered golf course architect, famous for his ability to work with the natural contours of the land as opposed to using invasive techniques (e.g., blasting) to achieve an artificial course layout [35]. Although the forested areas of golf courses, along with wetlands [30,31], are typically the sites that maintain ecological integrity, there is little in the existing literature that considers the health of golf course trees specifically.

In golf course culture, the game tends to be prioritized above all else. Put simply, "turf trumps trees" [36]. According to one Nova Scotian superintendent consulted during the study, due to growing pressure to make greens "faster" (i.e., shorter grass) for increased challenge of play, the use of fertilizers and herbicides on that specific course has nearly doubled in the past decade, despite industry-wide pushes to reduce chemical use [21]. Similarly, as golfers seem to desire sweeping views of greens and waterbodies, course managers tend to favor maintaining vistas over preserving a healthy tree canopy, opting to remove regenerating trees rather than allowing the development of a healthy multi-successional understory in forest patches. This is exacerbated by a golfer preference for no understory, as golf balls can easily get lost in thick vegetation [5]. This results in trees being treated

as infrastructure rather than living individual organisms contributing to a wider ecosystem. This is reflected by the tendency of maintenance crews and superintendents—those on the frontlines of course care—to vilify trees as destructive and disposable nuisances. This was demonstrated in an article published by Patrick Gross, Regional Director of the United States Golf Association, which included "Tree" as one of the seven "dirty words" of golf course maintenance [37]. A similar sentiment was reflected by another Nova Scotian golf course superintendent who, during a course visit, described one of the signature species of the Acadian Forest Region, and indeed of Canada, the sugar maple (*Acer saccharum)*, as a "weed", due to the ease with which it grows in its native habitat.

Ecologically speaking, monoculture fields of maintained grass are of low value, likely contributing more to environmental problems through their upkeep than anything else [6–10,18,32]. That said, grass is a permeable surface, and therefore, may arguably be better than an impermeable urban development such as a rooftop or parking lot. For example, turfgrass has been found to be an effective storm-water filter due to its extensive root network [21]. However, the canopy, bark, and roots of trees are more effective stormwater processors due to their higher capacity to retain excess water and filter nutrients [38,39]. Likewise, although golf course grass (inclusive of all four play spaces noted above) has been found to have a notable capacity for carbon sequestration, with an estimated potential to absorb 28.7 Tg per year in the United States alone, this positive effect is mitigated by its short-term storage, and the inefficient maintenance practices employed for its upkeep [40].Comparatively, though golf course trees absorb an average of 2.8 times more carbon than turf [41]—which doubtlessly will be increasingly important in the climate-change context—they are consistently advocated against due to their effects on turf: shading, limiting air flow, interfering with play, and engaging in root competition [36,42,43]. This can result in overmanaged treed landscapes, which Jim and Chen describe as being "regular and standardized, if not sanitized" [26] (p. 1720). Therefore, in the pursuit of long-term ecological integrity, improved golf course management is of the utmost importance. Yet, management practices—improved or otherwise—will cease to matter if golf courses begin disappearing altogether from urban environments.

As urban areas expand to accommodate a perpetually growing human population, it is hypothesized that community buy-in will be increasingly important to the preservation of urban greenspaces [44–46]. However, perceptions of golf courses as environmentally detrimental and elitist [47] seem unlikely to elicit widespread interest in their preservation. If not illustrated by the chain-link fences characteristic of most inner-city courses, the exclusivity of these spaces is often demonstrated by restrictive membership policies. For example, the 260-year-old Royal and Ancient Golf Club of St. Andrew's in Scotland refused admission of women until legally forced to do so in 2014 [48]. Furthermore, there is a dramatically disproportionate representation—approximately 70%—of white, male, high-income earners within the golfing population [39]. Compared to the mid-1990s, the proportion of golfers aged 18–34 has decreased by roughly 30% in the United States [47]. Since most millennials have been found to have less disposable income, more debt, and less free time than preceding generations, club memberships are largely unaffordable, while the game itself is perceived to be too time-consuming [47,49]. In an age where five-hour forays across fairways is a luxury few can afford, especially at $50 to $100 CAD per round [50], many millennials find the exclusivity of golf to be unappealing [47]. This disinterest has resulted in declining memberships and increasing course closures across the United States, the United Kingdom, and Canada [29,47,48]. These perceptions are detrimental to the longevity of golf courses; without community support, these spaces are likely to be redeveloped into something with less ecological potential: a suburban neighborhood, for example [51]. Further, these redevelopments are not always seen as negative occurrences, especially in highly populated cities such as Toronto, where space for residential development only exists at a premium cost [52]. However, it is difficult to determine which is least financially accessible: an inner-city golf course or a new suburban development in Canada's most expensive city [53]? Moreover, which supports a greater or better biodiversity: a golf course or a residential development? With a significant amount of global land occupied by golf courses, it is important to understand the environmental

implications of these developments in terms of ecological integrity and long-term sustainability. Thus, in addition to improved course management, increased community support is essential to transforming antiquated "golfscapes" into more modern iterations of recreational and ecologically functional landscapes.

The Acadian Forest Context

Nova Scotia is dominated by the Acadian forest, a hybrid between the northern boreal and the more temperate Great Lakes-St. Lawrence forest region [54,55]. The Acadian forest is less prone to fire than the boreal forest and boasts over 40 native tree species [13,55]. However, due to a history of extensive harvesting, the Acadian forest is currently largely dominated by early-successional species such as Balsam fir (*Abies balsamea)* and White birch (*Betula papyrifera*) [56], as opposed to its more characteristic, shade-tolerant, mid-to-late successional species such as the iconic Sugar maple, Eastern hemlock (*Tsuga canadensis*), Eastern white pine (*Pinus strobus*), Yellow birch (*Betula alleghaniensis*), American beech (*Fagus grandifolia*), and Red spruce (*Picea rubens*) [54,55]—a complete list of Acadian species binomials can be found at the end of this article as Appendix A. This has resulted in the Acadian forest region being listed as an endangered ecosystem by the World Wildlife Fund [57].

While urban forests present an underused opportunity for promoting Acadian species, it is challenging to maintain Acadian forest structure in urban landscapes due to harsh growing conditions such as poor soil quality and growth restrictions presented by the built environment [58] to which some species are ill-suited. Thus, institutional greenspaces present a unique opportunity to maintain and enhance tree populations representative of the natural Acadian forest structure within the urban landscape. To demonstrate the ecological potential of golf courses in NS, a comparison was conducted between tree data collected on OAGC and data collected in naturalized (i.e., "ingrowth") forest patches around the HRM.

3. Materials and Methods

3.1. Study Area

High-resolution ortho-imagery (Pictometry® [59]) of the course was obtained courtesy of the HRM. As the imagery was slightly dated, having been captured in 2014, this process was complemented using publicly available Google Satellite imagery circa 2016. The resulting maps included the delineation of parcel ownership to illustrate study area boundaries, and a gridline overlay to break the extent of the course into manageable and systematic pieces. These finalized maps were used to distinguish between those treed areas that were to be measured using plot-based sampling and those that were to be included in a single-tree census. The maps were updated manually, and adjustments to canopy coverage were made as needed. That is, if significant dieback had occurred since the Google Satellite imagery was captured, this was manually recorded, and later maps were digitally edited to reflect these changes. At the end of the field season, all maps were updated in ArcGIS and amalgamated into a single course overview map (Figure 2).

3.2. Inventory Overview

Two approaches were employed to inventory the tree population on OAGC: an individual inventory of stand-alone trees, and plot-based samples of forested patches. To maintain consistency, stand-alone trees were identified as those that were mowed around by the course maintenance crew or occurred in patches with little to no understory vegetation (e.g., shrubs, wild grasses). This census yielded data from over 5000 stems. Conversely, forest patches were defined by their resemblance to naturalized forest due to the proliferation of unmaintained understory vegetation. Nearly 2000 trees were measured in 22 plots. Vital tree data included species and diameter at breast height (DBH, measured 1.3 m above ground [60]). Any tree encountered with a DBH lower than 2.5 cm was excluded from the inventory. Other data collected included crown condition (poor, fair, good, or very

good—this ranking system is described in more detail in Table 1); signs of disease (e.g., beech-bark disease); and any notes of interest (e.g., interference of tree growth due to infrastructure). Additionally, any standing dead trees (snags) were included in the inventory. Species was recorded for snags when possible. This sampling method was adapted from the "Neighbourwoods" protocol [61].
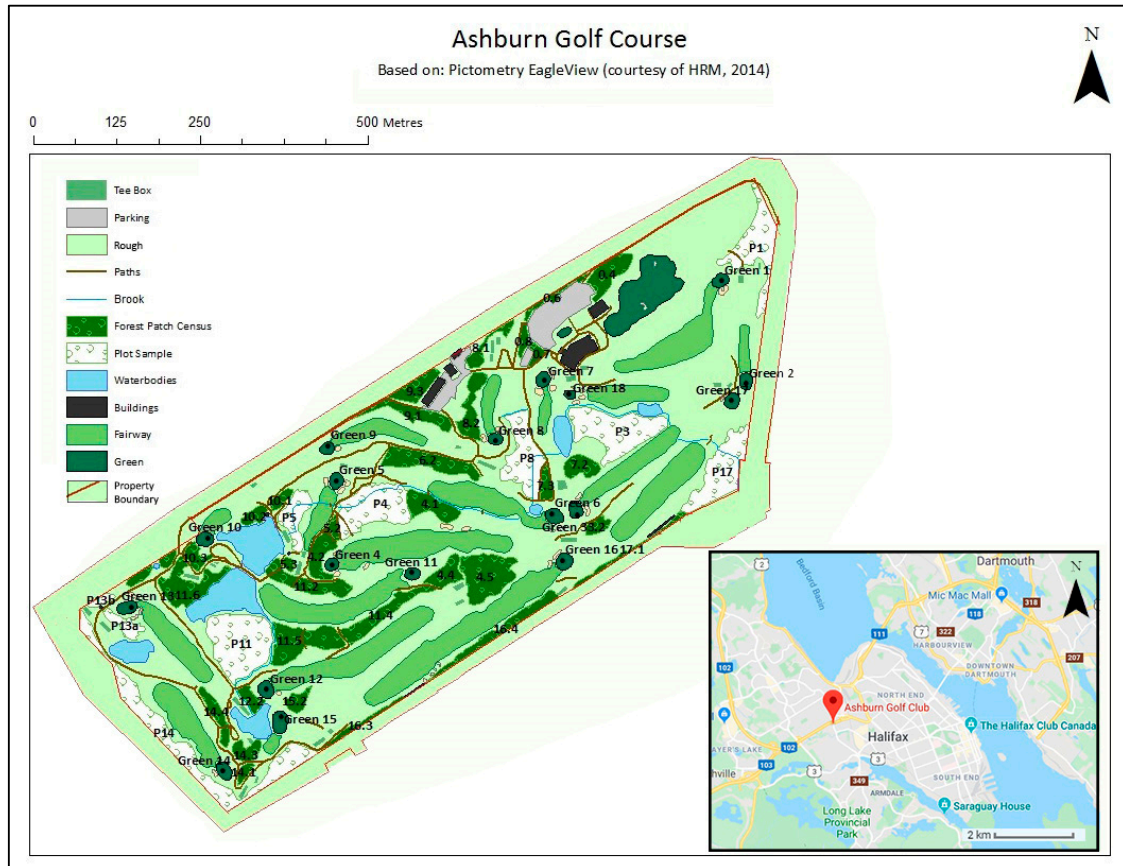

Figure 2. Overview map of OAGC; red outline reflects property boundaries. Inset map provided to illustrate the course's location within the HRM. Source: ©2019 Google.

Table 1. Description of crown condition ranking system. Adapted from Neighbourwoods protocol [61].

Crown Condition	
Snag (0)	**Dead Tree**
Poor (1)	Significant defoliation (0%–50% remaining canopy), weak/discolored foliage
Fair (2)	Some defoliation (50%–74% remaining canopy), some weak/discoloring foliage
Good (3)	Minor defoliation (75%–84% remaining canopy), mostly healthy foliage
Very Good (4)	No defoliation (85%–100%), healthy foliage

For the inventory of forest patches, plot-based samples were used to reduce plot edge per unit area [62,63]. While random plot placement is generally considered best practice to avoid bias [62], due to the course's small and irregularly shaped patches, plot centers were located as near to the center of the patch as possible to capture as many stems as possible. If a patch was big enough (i.e., over 0.5 ha) to warrant the use of two plots, the plot centers were located on either end of the patch to avoid overlap. Also included in this inventory was a census of all large trees in plot-sample patches. This included any trees with a DBH of 40 cm and above, so as not to miss any representative specimens and major seed sources within sampled areas. A total of 6954 trees were measured as part of the tree inventory. However, based on extrapolation of plot data, there are well over 20,000 trees growing on the OAGC. These projections were found using the area of forested patches on the OAGC and have

been used throughout data analysis to emphasize the significant contributions institutional greenspace can make to the urban forest. Regarding plot size, Avery and Burkhart [60] (p. 223) suggest using a fixed plot-sample size. However, Van Laar and Akça [62] (pp. 233–234) assert that a plot-sample size should be determined relational to the size of the forest patch. Due to the irregularly shaped and sometimes small patch sizes, a combination of both methods was used. A 400 m^2 area was determined to be an adequate plot size for most forested patches on the course, in accordance with the United States Department of Agriculture Forest Service [64] p. 11. Likewise, the standard of sampling error of approximately 10% per patch was adhered to, though in a few instances this was not possible due to the irregular shape of the patches [63]. One exception was made, however, for the long, narrow forested patch along the south-western edge of OAGC. This patch was too dense to census, but too narrow to accommodate the standard 400 m^2 plots. Thus, the plot size was adjusted to 100 m^2. These nine plots were laid out at regular intervals from the northern-most to the southern-most corners of the patch, resulting in a 10% standard sampling error. Data from both sampling methods were analyzed to determine signature, dominant, and solitary species; size and species distribution; dominant species distribution of snags; and crown condition across species.

3.3. Comparative Analysis: Data Selection

To execute a comparison between the trees found on OAGC and nearby, more naturalized forest areas, data were obtained from an i-Tree Eco assessment conducted in treed areas in the HRM by Foster and Witherspoon in 2016 [11]. Using plot-based methods highly similar to those employed on the OAGC, Foster and Witherspoon randomly dropped 20 plots within each of the 10 neighborhoods identified by the HRM's Urban Forest Master Plan (UFMP) [65], including the Ashburn/Armdale neighborhood in which the OAGC is located. This resulted in the inventory of 3670 trees growing in 200 plots [66]. While Foster and Witherspoon [11] measured a host of tree variables (see [66] for a full list), we extracted from their database only the variables that matched our measurements of OAGC trees. For the purpose of this comparative study, all data collected in plots labelled "Ingrowth" growing on sites with the land-use label "Vacant" (defined as "no discernable land use" [66] (p. 12)), as opposed to "Residential" or "Commercial", were used from the i-Tree Eco Assessment, totaling 2927 stems in 54 plots, as is shown in Figure 3. Every neighborhood is represented in this sample, with the exception of the halifax Peninsula due to no "Ingrowth" plots being sampled there. Some neighborhoods such as Beaver Bank, Sackville, and Spryfield are more heavily represented, likely due to less development and more tree coverage. To clarify, plots with these labels were used because these areas have, presumably, been left to naturalize, relatively undisturbed by human intervention. None of the plots included in the comparative analysis occurred on the golf course itself, though two were located within the Ashburn/Armdale neighborhood. Compared with the golf course data, these two plots provide an interesting glimpse into the differences between the institutional golfscape and the naturalized woodland in this HRM neighborhood. A summary flowchart of these methods can be seen in Figure 4 at the end of this section.

The comparison between OAGC and areas of ingrowth in the HRM is meant to illustrate the similarities in forest composition between an institutional greenspace and more naturalized areas, thereby demonstrating the potential for ecological integrity within the former, especially in urban areas where greenspace is limited. As both tree populations grow in the Acadian Forest Region, the Acadian Forest has been used as the ideal definition of forest composition and structure. Since demonstrating the potential for ecological integrity on institutional greenspaces is the aim of this study, emphasis has been placed on assessing the OAGC forest structure.
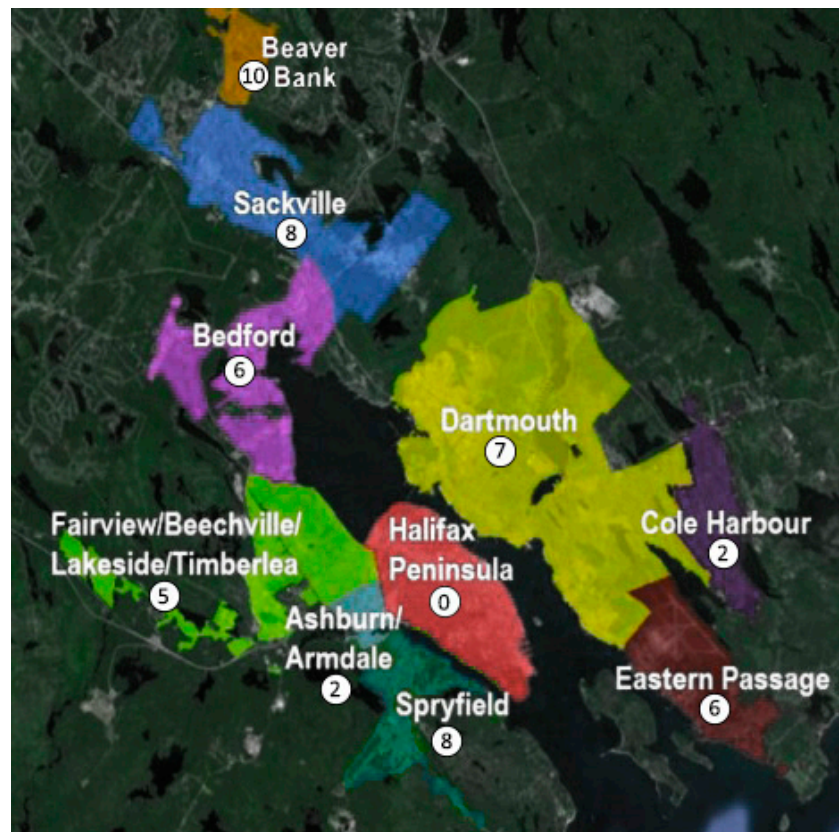
Figure 3. Overview of the 10 neighborhoods identified by, and adapted from, the halifax Urban Master Plan [66]. Labels added to indicate number of plots from each neighborhood used in comparative study.
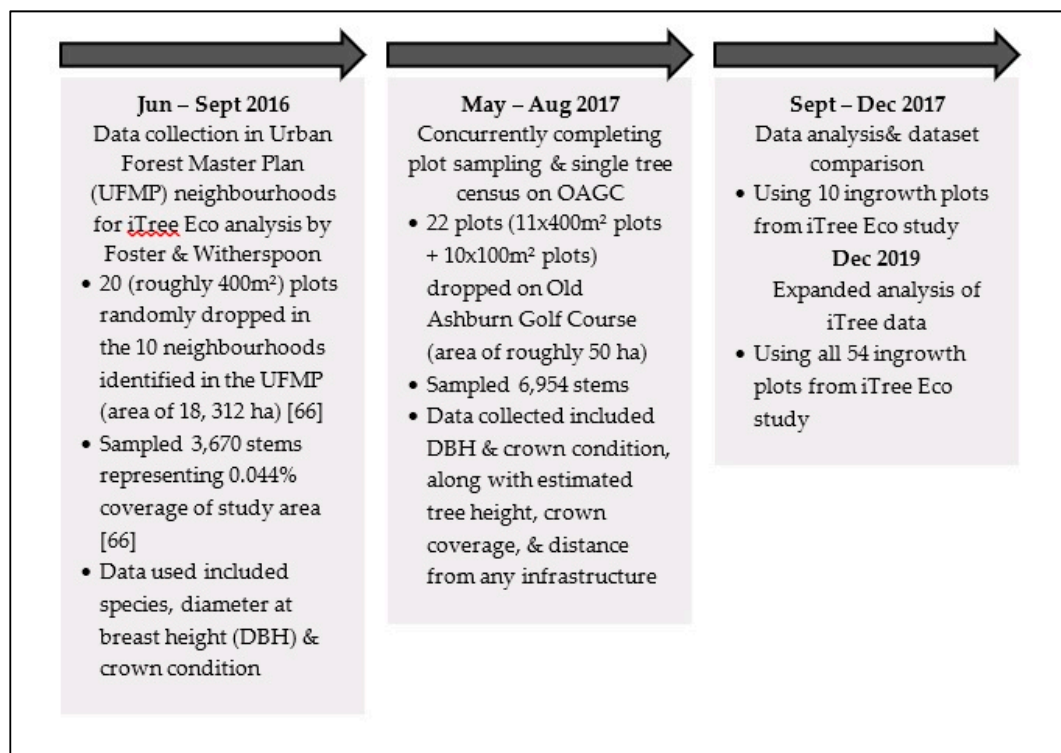

Figure 4. Flow chart of methods.

4. Results and Discussion

4.1. Species Composition: OAGC versus Ingrowth

With a tree population of 72% hardwood species, it was found that the tree population on OAGC is largely representative of a mixedwood Acadian forest [67], with large amounts of Red spruce, Eastern white pine, Yellow birch, and Red maple (*Acer rubrum*) [12,13]. The Forest Ecosystem Classification of Nova Scotia (FECNS) defines sites with this species distribution as either *AC10* or *AC11* ecosites, which often include a blend of early-successional and mid-to-late successional species with nutrient medium, highly moist soil [67] (p. 375). These forests are long-lasting. Though sometimes susceptible to windthrow, they can grow for up to 1000 years without disturbance [67].

Tree populations dominated by such species as Red maple, White ash (*Fraxinus americana*), Eastern larch (*Larix laricina*), and Eastern white pine are in a state of "edaphic climax", as opposed to the "climatic climax" of later-successional species due to "local extremes in site conditions" (p. 239). Though some foresters dispute the term "climax" due to the implication that forests in this stage are somehow inherently better than those in earlier stages—despite a multi-successional forest being demonstrative of a healthy forest structure [68]—these later-successional forests are indeed valuable, if only due to their rarity among Nova Scotian forest ecosystems [69]. Although it has been theorized that mid-to-late-successional tree species are characteristic of a healthy Acadian Forest, as alluded to above, due to a history of rampant harvesting across the province, less than 5% of forest cover in the Canadian Maritimes is older than 100 years old [69]. As a result, NS forests tend to be in much earlier successional stages than they would be had they been unaffected by disturbance by settler populations. In the context of this study, while it has been found that forests in the HRM may currently be in a phase of edaphic climax, ample opportunities exist for forest revitalization in managed spaces.

Interestingly, there were fewer ornamental species on the OAGC than what might be expected based on common perceptions of golf courses as ornamental rather than ecological spaces [5]. Besides a few Siberian pea trees (*Caragana arborescens*), Japanese maples (*Acer palmatum*), "King Crimson" Norway maples (*Acer platanoides*), and one Bebb willow tree (*Salix bebbiana*), most species found on the course were representative of the Acadian Forest of NS [12,13,55,67]. Although Ash (*Fraxinus*) is the namesake species of OAGC, its signature species—in this context defined as the most strongly represented species in the overall population—was Red maple. While the invasive Norway maple can also be counted among the more abundant species on the course, none was found regenerating in forest patches; rather, they were typically found in areas near fairways where they had been planted by course managers. Overall, with approximately 770 trees of non-native species sampled on the course, an estimated 90% of course trees are native, Acadian species.

An important distinguishing feature of the Acadian Forest in NS is the predominance of Red spruce, which is strongly represented in both populations [67] (see Figures 5 and 6; for detailed information on all species found on both the OAGC and Ingrowth patches, see Appendix B). Furthermore, both areas are representative of other signature Acadian species, with Red maple, Red spruce, and Yellow birch making up a significant portion of both populations. There was a comparatively small amount of characteristic longer-lived species, such as Eastern hemlock and Red oak on the golf course. This challenges the potential long-term development of Acadian old-growth forest on the OAGC—a climax forest stage widely considered to be underrepresented across the Nova Scotian landscape [69]. Yet, the balanced mix between early successional stage species such as Serviceberry, Balsam fir (*Abies balsamea*), and White birch with mid-to-late successional species such as Yellow birch and Red spruce, suggests that while both forest populations may be in earlier successional stages, both are experiencing healthy regeneration [67]. Additionally, the representation of Eastern larch within the OAGC population suggests wet, perhaps even boggy, conditions [55] characteristic of the biodiverse wetlands common to golf courses [34,35].
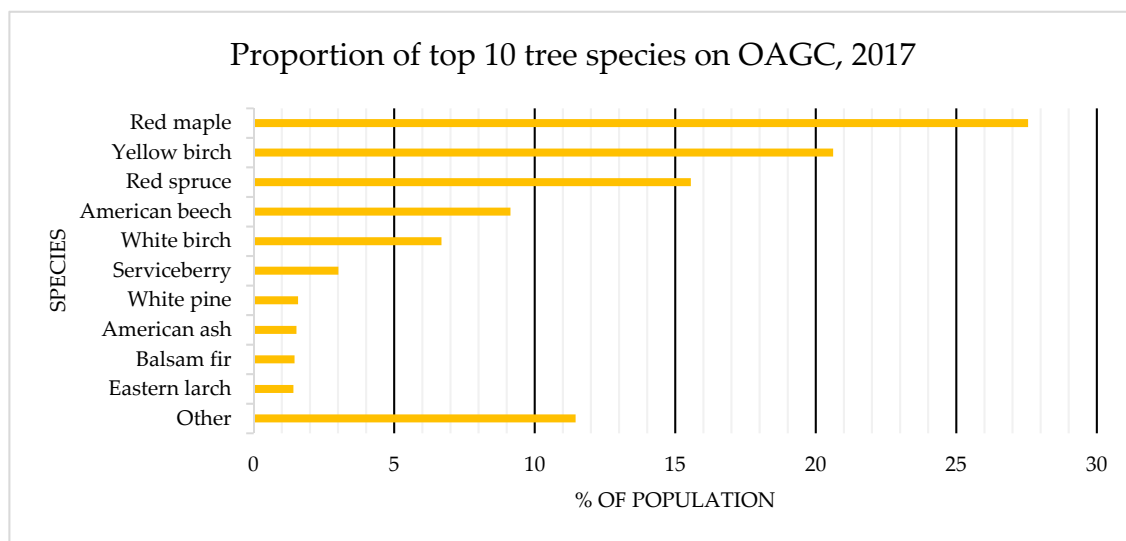
Figure 5. Ten most commonly occurring tree species found on OAGC in 2017. The high proportion of mid-successional species such as Red maple, Yellow birch, and Red spruce mixed with early successional species such as White birch, Serviceberry, and Balsam fir suggest a healthy, young forest. The representation of only one later successional species, American beech, indicate that without management intervention this forest may not have the potential to flourish into an old-growth population.
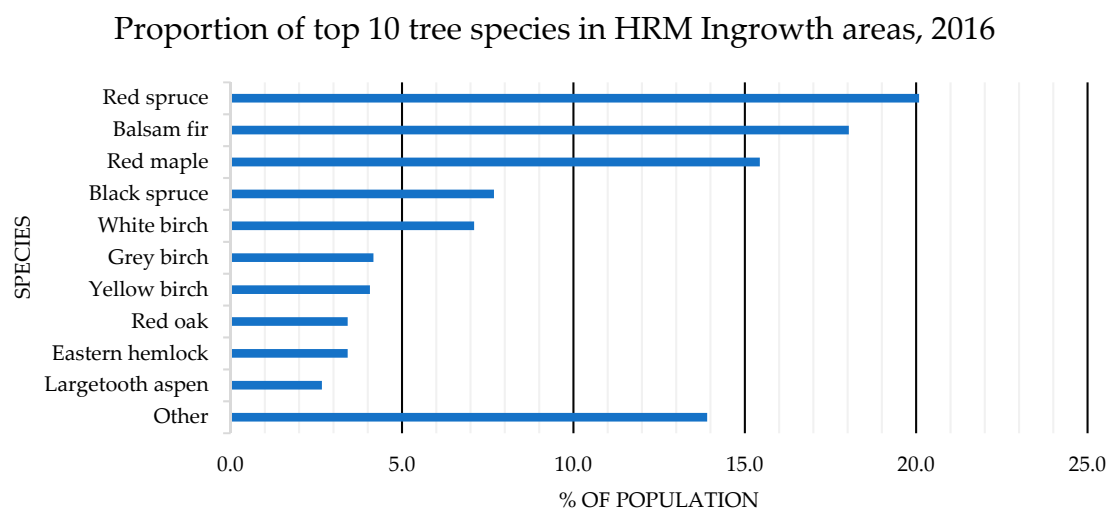

Figure 6. Ten most commonly occurring tree species found in HRM ingrowth areas (Data: 11). With a high proportion of early successional species such as Balsam fir, White birch, and Grey birch, a smaller proportion of mid-successional species such as Red spruce and Yellow birch, and some late-successional species such as Red oak and Eastern hemlock, this species distribution is demonstrative of a healthy Acadian forest.

As a point of interest, the dominant species of the Ashburn/Armdale ingrowth plots have been extracted from the overall "Ingrowth" sample and isolated in Figure 7. Unfortunately, the two plots sampled only rendered 99 data entries, and as such may be too small to be statistically significant. However, this small proportion of naturalized forest space within the Ashburn/Armdale neighborhood reinforces our claim that institutional greenspace is important to the maintenance of ecological integrity in urban forests. The naturalized population within this sample shows a significantly higher proportion of long-lived, foundational Acadian forest species, most notably, American beech and Eastern hemlock. While on the surface this is suggestive of more mature Acadian forest patches, these species are at risk in NS due to the proliferation of beech bark disease (*Cryptococcus fagisuga*) [70,71] coupled with the

invasion of beech leaf-mining weevil (Orchestes fagi) [72] and the increasingly prevalent Hemlock woolly adelgid (*Adelges tsugae*) [73] as illustrated in Figure 8. With an absence of interference from buildings or infrastructure, diseases and pests are likely the cause of nearly 90% of the American beech trees sampled in these plots being labelled as snags. Without intervention by forest managers, the ecological integrity of these forest patches is likely to further degrade as the Hemlock woolly adelgid becomes more common within the HRM. Though the early-successional species distribution within these plots is similar to that found on the OAGC, with only mid-successional species like Yellow birch and Red maple to replace longer-lived species, it could be hypothesized that the naturalized plots are in worse condition than forest patches on the OAGC.
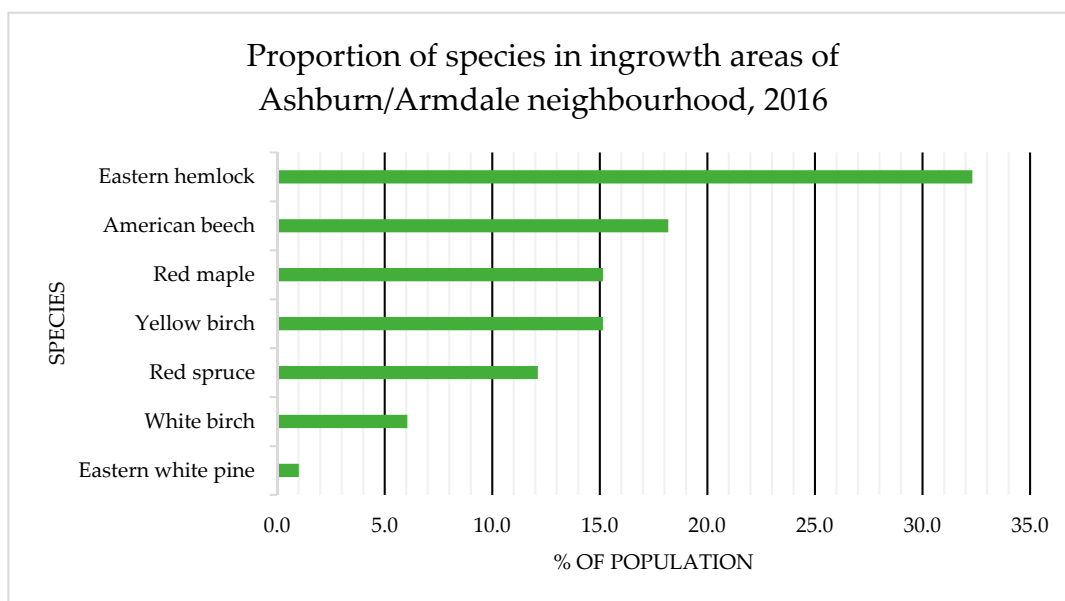
Figure 7. Species distribution in the Ashburn/Armdale neighborhood (adapted from source data: 11). Large proportions of the longest-lived, foundational species of the Acadian forest, such as Eastern hemlock, American beech, and Red spruce, suggest that ecological integrity of naturalized forest patches may be higher than those patches found on golf courses.
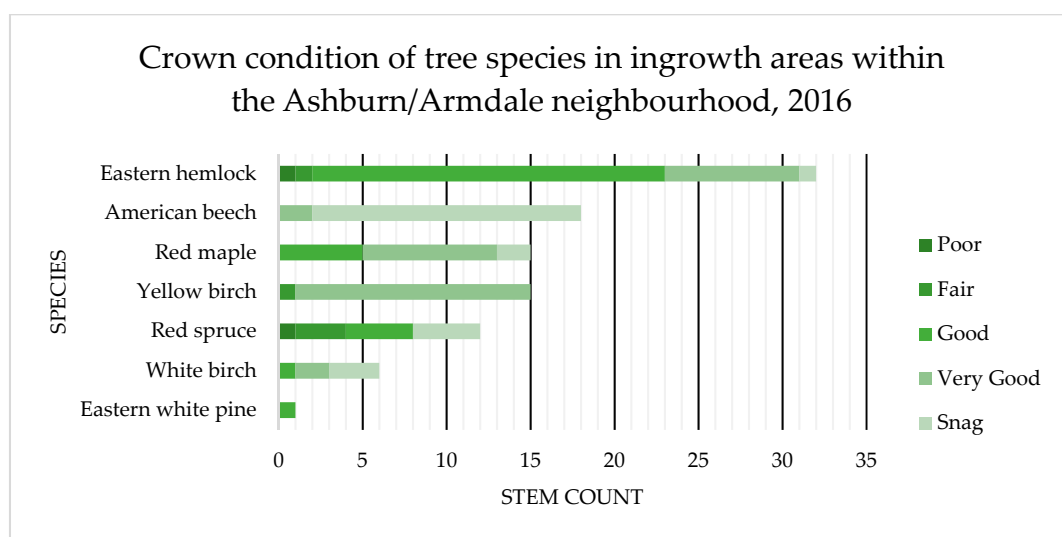

Figure 8. Crown condition of trees sampled within ingrowth areas of the Ashburn/Armdale neighborhood (adapted from source data: 11). The high proportion of snags among the American beech population suggest that disease and pest infestations have decimated the population of this foundational Acadian species.

On a larger scale, both the forests in the 54 ingrowth plots, and those patches sampled on the OAGC appear to be immature overall as characterized by significant regeneration. This is evidenced by the distribution of stem size across both populations regardless of species (Figures 9 and 10). Since early-successional species were heavily represented in both populations, it is unsurprising that size distribution demonstrates a disproportionate abundance of smaller trees comparative to larger, and presumably more mature, trees. It is important to note here that tree size is not usually a good indicator of tree age, since growth rates vary across species and are highly influenced by nutrient and light availability. For example, the shade-tolerant Eastern hemlock has a potential life expectancy of approximately 600 years [13,55] and can live suppressed in the understory for decades until an opening in the canopy provides enough light for it to grow. Thus, it cannot always be assumed that a small tree is a young tree, nor that a large tree is truly mature.
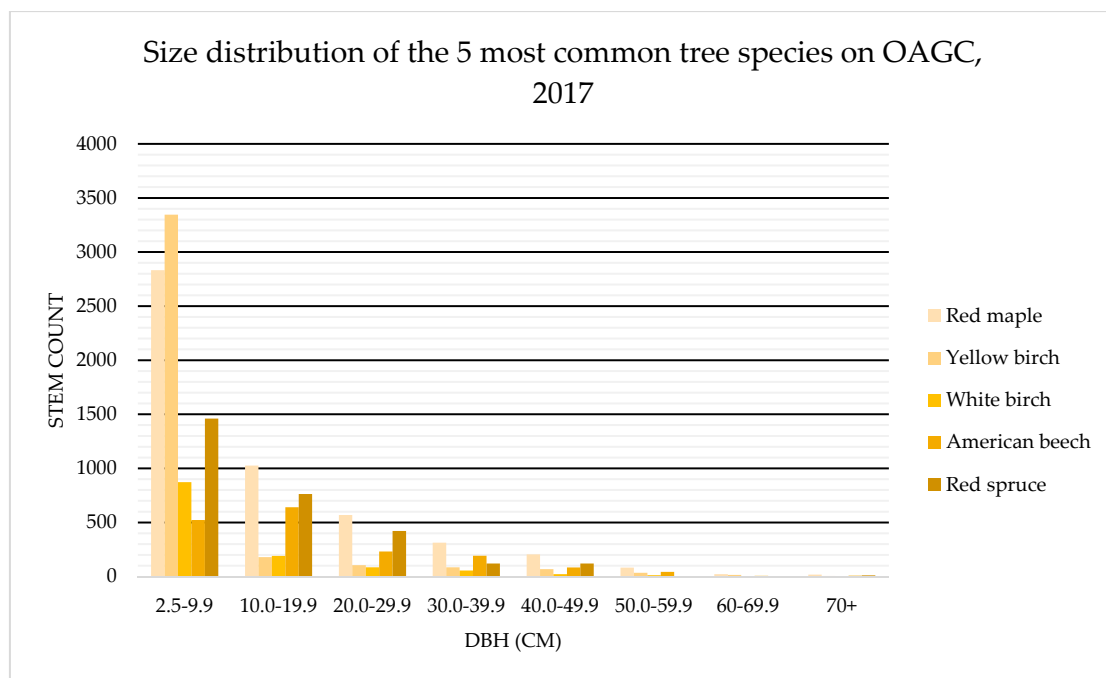

Figure 9. Size distribution of the five most commonly occurring species (based on plot projections) on OAGC in 2017. The abundance of small-stemmed Yellow birch and Red maple (as opposed to Pin cherry, White birch, or Trembling aspen) suggest that the forest is moving into a more mature phase.

Nonetheless, size distribution can provide insight into the compositional structure of a forest, since many species are characteristically present in young forests, such as trembling aspen (*Populus tremuloides*), striped maple (*Acer pensylvanicum*), or in the case of OAGC, White birch. Furthermore, by identifying which species comprise the understory and which contribute to the overstory, one may identify the current forest phase or predict the likely future forest structure, assuming that severe weather or any other disturbance event does not complicate natural succession. According to the size distribution on OAGC compared to that seen in ingrowth plots, it can be inferred that the course population is more mature, due to the presence of trees with a DBH over 50 cm. However, the low proportion of big trees is surprising, considering that the course is nearly 100 years old [39]. Due to the abundance of later successional species such as Yellow birch within the smallest size class, it is likely that the OAGC forest is a second growth mid-successional forest. Its capacity to move beyond an early successional stage and sustain a healthy size distribution, with representation from species of various age classes, is suggestive of ecological integrity [2].

Conversely, ingrowth patches showed a significant lack of big trees, which may be the result of edge effects often experienced by ecosystems bordering development [74]. While, the decreasing

representation of Balsam fir, Red maple, and White birch in the larger size categories is consistent with the normal lifecycle of these species, the low representation of longer-lived species in this population threatens the longevity of the forest. It seems that the size and species distribution among OAGC trees is more representative of a healthy forest dynamic than what is seen within the naturalized plots.
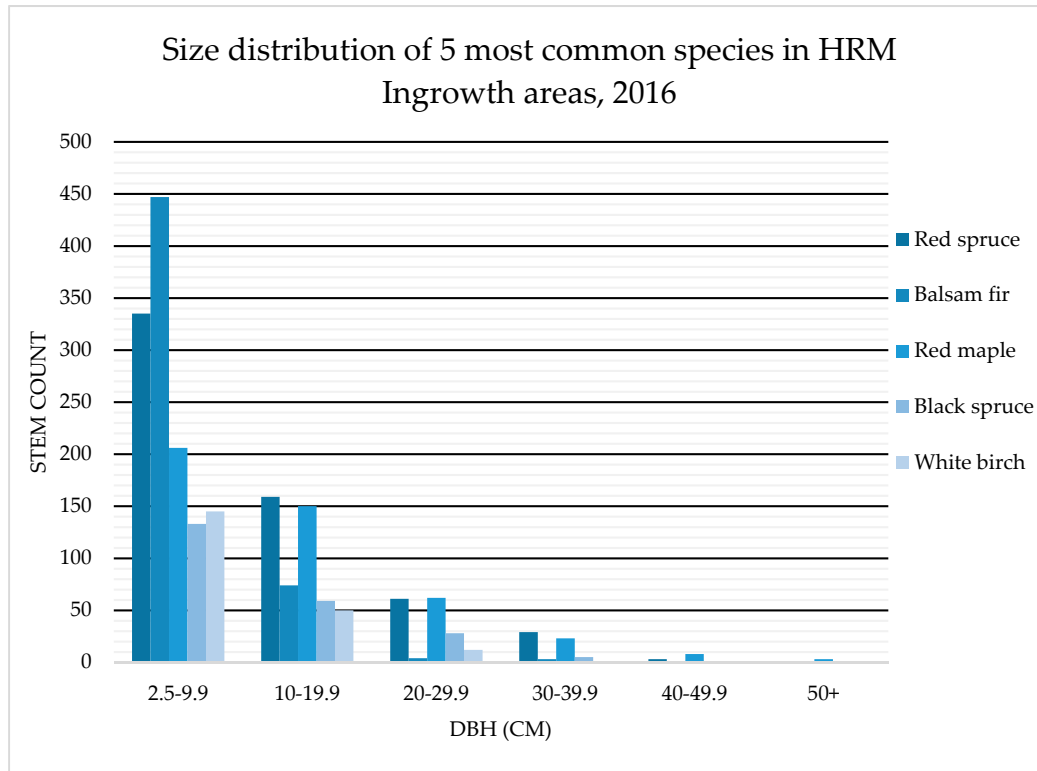

Figure 10. Size distribution of the five most commonly occurring species in HRM ingrowth areas (adapted from source data: 11), demonstrating a lack of mature or big trees.

4.2. Tree Health: OAGC Versus Ingrowth

As mentioned above, a notable similarity between both the ingrowth and OAGC populations is the poor health of American beech trees, as almost every individual encountered in both populations was found dead or in a very poor condition, as can be seen in Figures 11 and 12. Where these populations differ is that while the American beech was among the top five most common species found on the OAGC, it only represented 1.6% of the Ingrowth population. The predominant presence of this foundational, old-growth species on the OAGC is suggestive of a more mature forest. However, the population of American beech on the OAGC has been decimated almost definitely due to disease and pest outbreaks [70–73], as demonstrated by the bark cankers evident on every snag. Although this deadwood could play a significant role in natural ecosystem dynamics, such as providing nutrients for soil and habitat for wildlife [68], much of it has been removed from forest patches by the course maintenance crew and standing dead trees are often cut as they can present a safety hazard. Conversely, the predominantly healthy crowns found among Red maple, Yellow birch, and Red spruce populations on the OAGC are demonstrative that the forest still maintains a robust mid-successional forest. However, innovative management measures may need to be taken to replace longer-lived species within the OAGC forest.
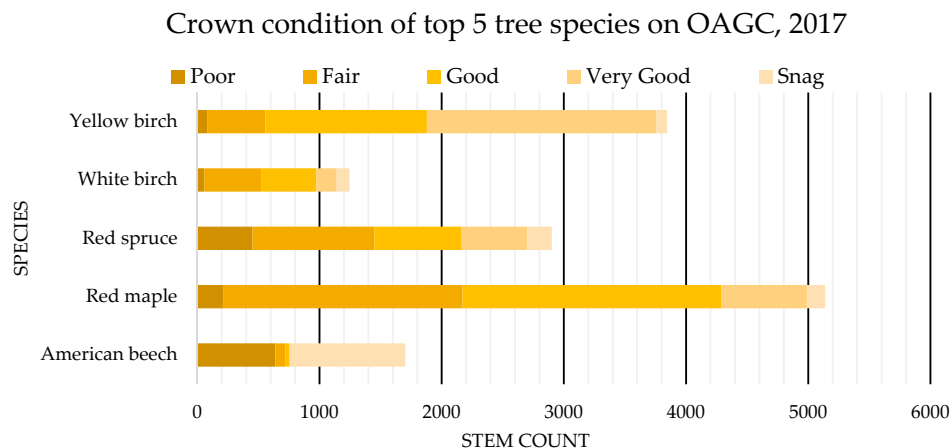
Crown condition of top 5 tree species on OAGC, 2017

Figure 11. Canopy condition of the top five measured species found on OAGC in 2017, showing the significant impact of pests and disease on American beech.
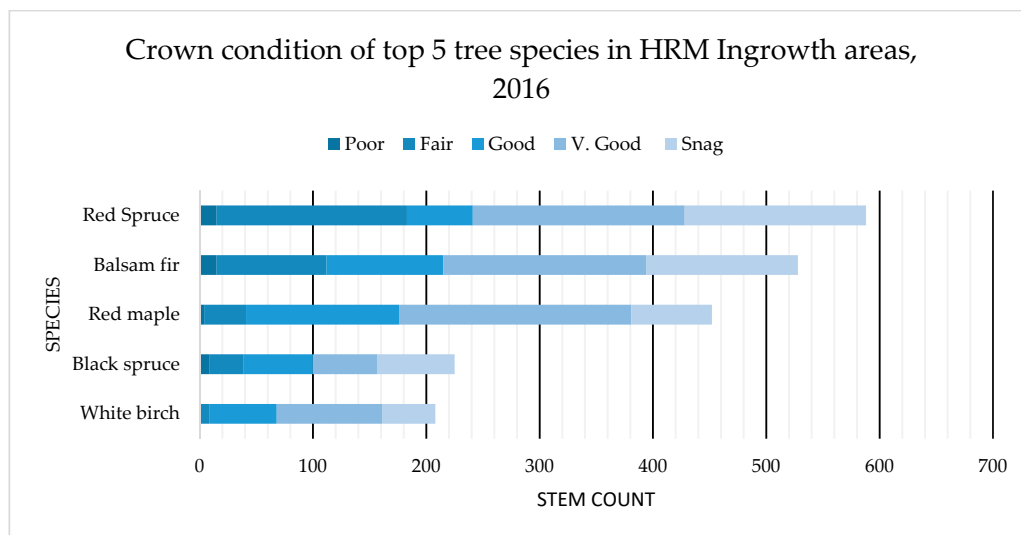

Figure 12. Canopy condition of top five species in HRM ingrowth areas, 2016 [adapted from source data: 11] demonstrating significant dieback across all species.

While each of the five most commonly occurring species in the HRM Ingrowth plots has a strong population, with crown condition of the majority of stems ranked between "Fair" and "Very Good", the population overall had a much higher proportion of snags across species comparative to the OAGC, as is shown in Figure 12. Although significant dieback is to be expected in early-successional species, especially Balsam fir, which has the tendency to grow in dense and highly competitive thickets, it could be a sign of decline in longer-lived species, such as Red spruce. When compared, the tree canopy on the OAGC is in better condition overall than in HRM Ingrowth plots. However, the dieback of American beech and the anticipated dieback of Eastern hemlock present challenges to maintaining ecological integrity in both populations. The possibility of human intervention on the OAGC increases the likelihood that these populations will survive these challenges, while the fate of the Ingrowth population is far less certain.

5. Conclusions

Study findings demonstrate that sustainable forest management in well-treed areas of golf courses can increase landscape/habitat connectivity and improve the health of multi-successional, native forests. These findings are meant to highlight the opportunity to increase the ecological integrity of the institutional golfscape by applying knowledge of the ecosystem dynamics of functional native forests.

Serving as a case study, this comparative analysis used the Acadian forest as a model, to determine whether golf courses, which are generally not perceived as ecologically functional spaces, might, in fact, have ecological merit that is sometimes even higher than that of naturalized forests. Further assessment of ecosystem services, or the adoption of a more holistic ecosystem-based study approach (e.g., including an inventory of local groundcover or local bird and amphibian species) would be a good addition to future studies using this comparative methodology. For this study, trees were made the primary focus because healthy trees: improve air quality [75]; aid in carbon sequestration and therefore contribute to climate change mitigation [45]; reduce erosion [63]; enhance biodiversity [31]; and improve water quality through filtration [63], among other ecosystem services [75,76]. Furthermore, OAGC trees will provide direct benefits to course visitors. These include physical benefits, such as protection from prolonged sun exposure through shade provision [77], and mental health benefits, such as reduced stress and increased feelings of wellbeing from spending time under and near the canopy [78].

Undoubtedly, golf courses take up a lot of space; truly, entire neighborhoods can fit inside their borders, as has been shown through increasingly common redevelopment of these spaces into residential areas [30]. Moreover, historically unsustainable maintenance practices have influenced skepticism of these spaces, despite the ecological value to be found in their forested areas. This suggests that many golf courses may no longer make ecological sense in their current incarnation and must be environmentally revitalized to maintain relevance in modern society. To do so, golf course managers must highlight the ecological value of these institutional greenspaces and adopt improved management practices to support and enhance this value. This is especially true as the impacts of climate change intensify. Golf course forests could significantly contribute to mitigating and adapting to climate change under the appropriate management conditions. For example, old growth species could be prioritized when planting trees as longer-lived trees have higher potential for carbon sequestration. In the context of the OAGC, this prioritization would enhance the health and structure of the forest considering the small amounts of longer-lived species such as Eastern hemlock and Red oak.

Furthermore, community buy-in through the provision of increased public access to these traditionally elitist spaces could improve public perceptions of golf courses and ultimately contribute to their longevity as ecologically functional spaces. OAGC managers have taken laudable first steps in this regard, largely by supporting this investigation and allowing public access to the course. As a result of the latter, folks living in the surrounding neighborhood often use OAGC pathways as walking/running trails and the lawn area as recreational space through the winter (e.g., snow shoeing and cross-country skiing). However, significant progress is now required on the ground to maintain and enhance the course's ecological integrity. In the Acadian forest context, this could mean considering innovative approaches to deal with pests and diseases, such as following treatment regimens to reduce impacts or replacing susceptible species with better-adapted counterparts. For example, while the American beech is sensitive to the mining weevil and Beech bark disease, its European cousin, *Fagus sylvatica*, is not. This replacement could present an opportunity to maintain the native species structure of the Acadian forest, and would serve as an interesting case study for assisted migration theory—a climate change adaptation strategy that is becoming increasingly common in Canadian forest management [79]. Alternatively, at risk foundational species like American beech and Eastern hemlock could be replaced by other long-lived native, but less common, species such as Red oak and Eastern white pine.

Ultimately, achieving sustainability on golf courses—and institutional greenspaces in general—means striking a balance between social responsibility and ecological integrity. This is largely dependent on a shift of cultural norms away from traditional perceptions of golf, moving away from inapt adages ("turf trumps trees") and toward new innovation ("turf and trees"). Blending modern, sustainably-minded management practices with the traditional aspects of golf course management could be a hole-in-one for urban forestry, providing a reasonable par for the development and management of ecologically viable, but still socially functional spaces—transitioning golf course management away from its environmentally detrimental reputation toward a more conservational future.

Appendix A

Table A1. Complete List of Acadian Tree Species Binomials.

Broadleaf Species	
Latin Name	**English Name**
Acer pensylvanicum	Striped maple
Acer rubrum	Red maple
Acer saccharinum	Silver maple
Acer saccharum	Sugar maple
Acer spicatum	Mountain maple
Amelanchier spp.	Serviceberry
Alnus incana	Speckled alder
Betula alleghaniensis	Yellow birch
Betula papyrifera	White birch
Betula populifolia	Grey birch
Fagus grandifolia	American beech
Fraxinus americana	White ash
Fraxinus nigra	Black ash
Populus grandidentata	Largetooth aspen
Populus tremuloides	Trembling aspen
Prunus pensylvanica	Pin cherry
Prunus serotine	Black cherry
Prunus virginiana	Choke cherry
Quercus alba	White oak
Quercus rubra	Red oak
Sorbus americana	American mountain ash
Tilia americana	Common linden (Basswood)
Ulmus americana	White (American) elm
Conifer Species	
Latin Name	**English Name**
Abies balsamea	Balsam fir
Larix laricina	Eastern larch
Picea glauca	White spruce
Picea mariana	Black spruce
Picea rubens	Red spruce
Pinus resinosa	Red pine
Pinus banksiana	Jack pine
Pinus strobus	Eastern white pine
Thuja occidentalis	Eastern white cedar
Tsuga canadensis	Eastern hemlock

Appendix B Tree Species Distribution on OAGC and HRM Ingrowth Areas

Figure A1. Total tree species distribution on OAGC, 2017 (higher frequency) based on plot projections made using the area of forested patches.

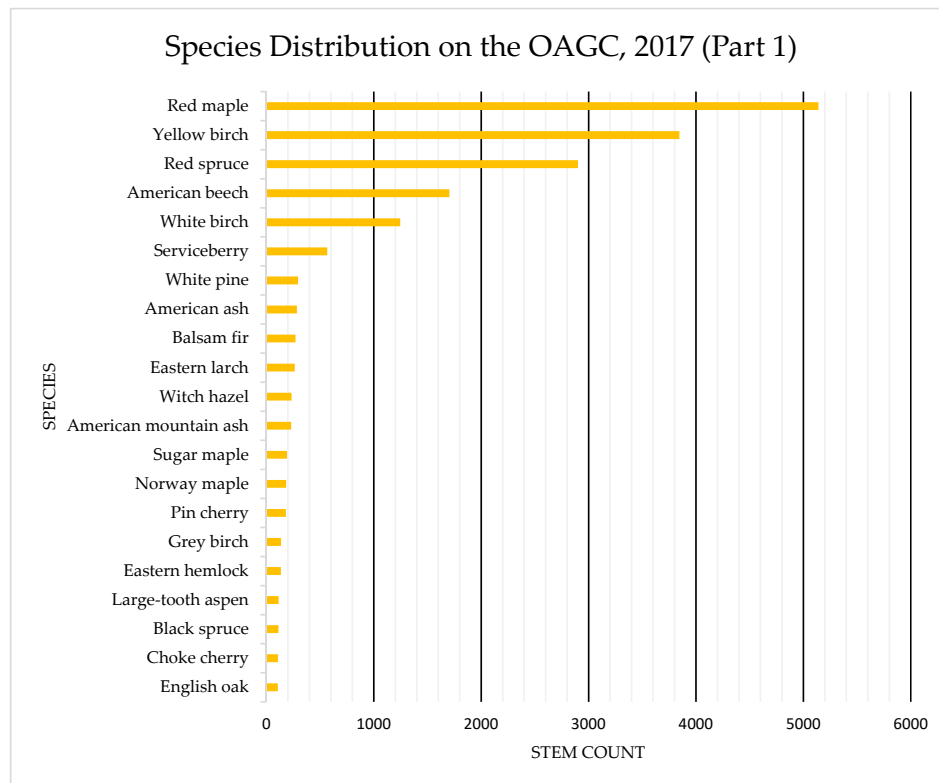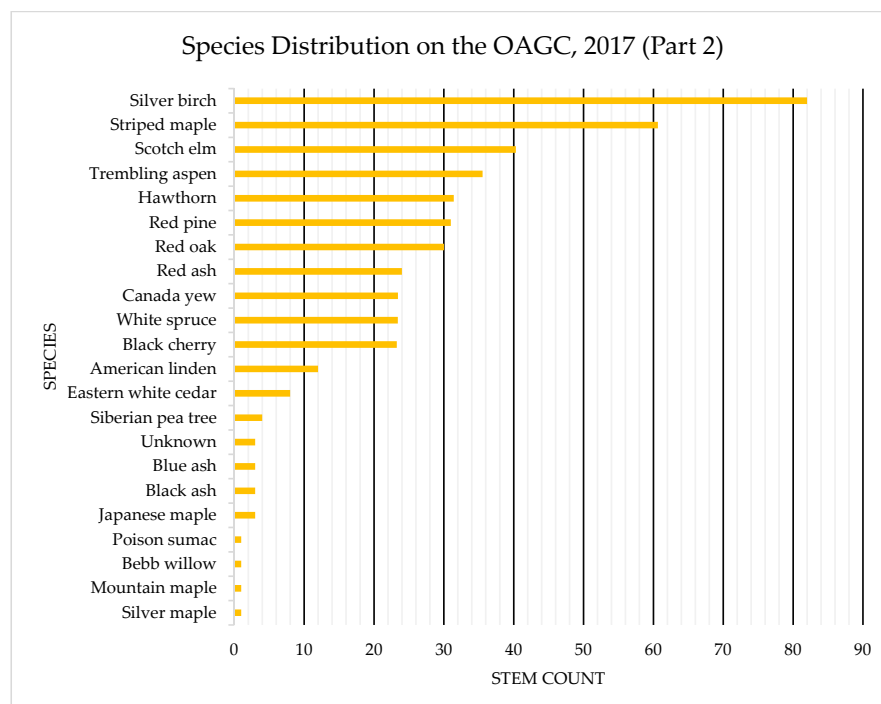**Figure A2.** Total tree species distribution on OAGC, 2017 (lesser frequency) based on plot projections made using the area of forested patches.
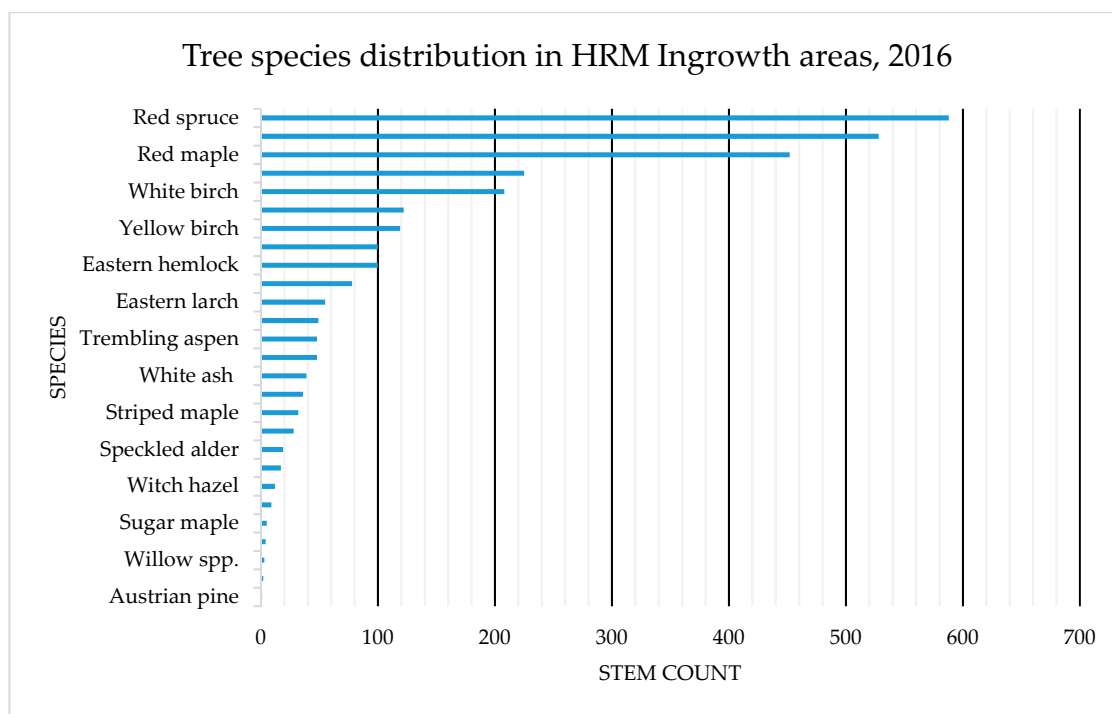
Figure A3. Total tree species distribution in halifax ingrowth areas, 2016 (adapted from source data: [11]).

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