A Framework for Urban Forest Naturalization

by

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the trees
so clustered
a bird could walk the branches
a thousand miles or more

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ABSTRACT

The way we design and manage cities directly impacts our quality of life. Urban ecosystems differ markedly from their rural surroundings due to the high density of built infrastructure, and are considered 'unnatural'. Naturalization is the process of altering the characteristics of urban greenspaces so that they resemble nearby reference ecosystems. Increasing naturalness has social, ecological, and economic benefits. The two objectives of this thesis were to create a conceptual framework for urban forest naturalization and test its use in guiding management decision-making. Thirty-eight framework dimensions were determined from a literature review on naturalness assessments and urban ecology, and using professional judgment. Sixteen urban sites in Halifax, Nova Scotia, and Winnipeg, Manitoba were visited that represented a broad range of urban settings, from an untreed roadside field to old-growth portions of parks. Each site received a score from 0 to 1 on each of sixteen naturalness dimensions. Applications of the framework to the sites demonstrated that management activities typically fell into one of three categories: stand initiation, site transformation, or monitoring. Within individual sites, the actions to achieve these goals differed. For instance, some 'transformation' sites required the introduction of native ground flora coupled with the removal of invasive species, while others required an increase in canopy differentiation and species composition. The framework thus helped pinpoint individual actions or goals that could be targeted to increase overall naturalness. Future research should consider how urban residents perceive different scores along the various framework dimensions, as psychological benefits may relate to perceived rather than ecological naturalness. Some dimensions may contribute to perceived but not ecological naturalness, or vice versa. It is important for managers to understand how their activities impact the experience of urban residents in greenspaces as well as non-human species.

LIST OF ABBREVIATIONS AND SYMBOLS USED

C:N = carbon:nitrogen

DBH = diameter at breast height

FEC = Forest Ecosystem Classification

IAS = invasive alien species

NSDNR = Nova Scotia Department of Natural Resources

US = United States

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CHAPTER 1 INTRODUCTION

The way that we design and manage our cities directly impacts our quality of life and the way cities interact with the surrounding environment (Jabareen 2006; Gaston et al. 2013). As the majority of the world's population lives in urban areas, and as this number is only expected to grow (World Health Organization 2014), urban planning and sustainability will only be of greater relevance. One way to increase the livability and sustainability of cities is to incorporate ecological naturalness into planning. Urban forests are often highly altered from non-urban forest conditions due to direct and indirect human impacts (e.g. McKinney 2002). Increasing the ecological naturalness of urban forests entails increasing their compositional, structural, and functional similarity to reference non-urban forests with minimal human influences. This occurs through urban restoration or afforestation projects, or, urban naturalization.

Naturalization is the creation or transition of an urban space to one with compositional, structural, and functional similarity to nearby non-urban forests considered 'natural' references. Natural urban forests are gaining momentum within urban planning. A review of twenty Canadian urban forest master plans found that many cite naturalness as a guiding principle or recommend actions related to increasing naturalness (L. Slapcoff, personal communication). The interest in naturalizing the urban forest relates to current international interest in conserving biological diversity. There is concern that biodiversity is being lost at an accelerated rate due to anthropogenic stressors and a desire to conserve biodiversity, as recognized for the first time in the 1992 Convention on Biological Diversity. Through naturalizing the urban forest, not only do we increase the habitat available for local species, but we also help foster connections between urban residents and the natural world. These connections hopefully translate into a conservation ethic, with a desire to protect biodiversity.

There is no comprehensive framework on the dimensions of urban forest naturalness or gradients along which the success of naturalization could be judged (Noss 2004). One of two objectives for this thesis was to create a comprehensive, though not exhaustive, framework for urban forest naturalness consisting of individual components or dimensions that could be assessed individually. These dimensions would act as gradients or state indicators along which naturalness and naturalization success could be judged. While dimensions are assessed individually, many can be managed collectively. The second objective was to demonstrate the utility of the framework in urban forest management through guiding naturalization activities using two Canadian cities. It may be that only some components of naturalness are of interest or feasible to manipulate at particular sites; some dimensions may be important conceptual components of naturalness, but inefficient or very difficult to specifically assess or manage. While many of the potential benefits arising from naturalization are social, our framework focuses primarily on ecological naturalness. Since naturalization is linked to a desire to conserve biodiversity and promote a conservation ethic, the biological qualities of a site are the primary targets of naturalization. However, some components of naturalness may be unsuitable for certain sites due to the socioecological setting in which urban forests occur.

Decisions regarding the management of the urban forest should consider potential benefits and disservices for humans and other species alike. While naturalization is an attempt to increase the similarities between urban and more-natural non-urban settings, it may be that some dimensions need to be tempered to sustain certain benefits to humans, such as feelings of safety. Chapter 2 is a brief overview of concepts, motivations, and practicalities underlying naturalness and naturalization. Chapter 3 explains the methods used in my research and their limitations. Chapter 4 consists of the naturalness framework, the rationale behind its structure and the dimensions chosen for inclusion, and potential applications of the framework. Chapter 5 demonstrates test applications of the framework to sixteen urban forest sites in Halifax, Nova Scotia, and Winnipeg, Manitoba. Test applications of the framework were to demonstrate which components practitioners should prioritize. Chapter 6 concludes the thesis, by summarizing the importance of naturalness and naturalization in liveable cities and the benefits urban forests could hopefully provide to city dwellers.

CHAPTER 2 FOUNDATIONS & CONCEPTS

A more-natural city, in which the urban forest ecologically resembles that of its natural non-urban surroundings, has many potential benefits for humans and non-humans alike. Greater feelings of well-being are associated with more-natural areas (Chiesura 2004; Peckham et al. 2013), and more-natural environments also provide habitat for many native species that are unable to establish under less-natural urban conditions. Determining where and how to naturalize cities requires incorporating ecological principles and knowledge to help manage spaces best suited socially and ecologically to their urban environment (Niemelä 1999). Ecological principles and knowledge may need to be tempered by human desires and values in urban planning. Not all spaces will be suitable for all goals, and some species may be unable to establish in urban areas regardless of naturalization (Soga et al. 2014).

Many urban residents prefer more-natural environments to built-up settings (Kaplan 1985). Urban residents identify stronger feelings of tranquility and contentment in more-natural areas, have reduced stress levels, and use natural areas for recreational and social purposes (Peckham et al. 2013; Sinclair et al. 2014). The more time people spend in these natural areas, the greater the physical and emotional benefits they receive (Kahn 2002; Jay and Schraml 2009; Hinds and Sparks 2011; Qin et al. 2013). Natural areas provide a relief or refuge from the urban environment (Chiesura 2004) – however, if more-natural areas were more abundant in urban settings, these experiences could be the norm rather than the exception.

Human interactions with a natural urban forest also have positive ramifications for other species, as it is through direct experiences that we develop extrinsic values, tied to lifestyle and the services the forest provides (Berenguer et al. 2005; Ward Thompson et al. 2007). For instance, a more-natural urban forest offers a unique recreational space, or provides edible plants such as local berries or medicines. We value the services provided to us by the forest and thus the forest itself. In the absence of direct interactions, if natural areas are not accessible and relevant to our lives, we can only identify abstract values around our concern for how the forest ought to be (Berenguer et al. 2005). These abstract values are not related to particular services or benefits, but are intrinsic states ascribed to forests. Since these are divorced from physical experiences with and awareness of the forest, it may be difficult to know if these values have changed or been lost, or advocate for their persistence. Since urban forests are the first and foremost and sometimes only interaction urban residents have with the natural world (Nowak et al. 2001), they could play a crucial role in developing these extrinsic values and attachments. More-natural urban forests could help reduce the human disconnect from nature and foster the development of citizens who are more connected to and concerned about non-human well-being, and aware of its ramifications for human well-being (Miller 2005).

Conversely, many regions of the urban forest could be said to contribute to an estrangement from nature if they only provide interactions with a homogeneous subset of regional flora and fauna and non-native species (Miller 2005). Ubiquity does not beget interest; landscape perceptions and preferences are positively linked to variety (Kaplan 1985). More-natural areas are more interesting to visit due to the variety in species and structural elements, or, the divergence from typical urban communities, and are associated with a feeling of discovery (Peckham et al. 2013). If we want urban residents to value the urban forest, which could act as a representative for natural non-urban settings and thus foster a connection to the broader non-human world, we need to rethink how we plan and manage our cities.

2.1 Urban Forests and the Urban Ecosystem

Simply put, the urban forest is all the trees in the city, from single trees planted in boulevards to remnant old-growth forests. Pauleit and Breuste (2011) identify four main groups of urban vegetation: (i) remnant forests, (ii) 'cultural' landscapes of agriculture, (iii) ornamental and designed spaces, and (iv) spontaneous urban vegetation (e.g. brown fields). Within any and all of these groups, the urban forest can be compositionally, structurally, and functionally different than its natural non-urban counterparts. Many areas in the urban forest are characterized as having a low richness of native flora and fauna, reduced structural diversity (e.g. the absence of a shrub layer), and lower canopy coverage and densities of trees (Beissinger and Osborne 1982; McKinney 2002; Turner et al. 2005). These and other characteristics are interrelated: the lower abundance of native insects stems from a lower availability of native plants (Tallamy 2004); some native bird species may be absent due to a lack of suitable habitat or attractants in the form of native vegetation, of which trees play an important part (Mills et al. 1989; Sandström et al. 2006).

While many characteristics of the urban forest are the direct product of human management and alteration, the physical urban environment also plays a significant role in shaping its characteristics. Urban ecosystems are a novel habitat type with globally homogenous characteristics. They have a large amount of impermeable surface, high degrees of fragmentation, higher temperatures than surrounding non-urban areas, and highly alkaline and dry soils with altered nutrient inputs (Thompson and McCarthy 2008; Williams et al. 2009; Alberti 2010; Lundholm and Richardson 2010; Werner 2011). Just as a mixed hardwood forest would be difficult to establish in a peat bog, achieving certain forest types in urban areas may be impossible, with or without human intervention, due to environmental constraints. Furthermore, a site's location within the urban environment can influence its potential for non-urban composition, structure, and function. Some environmental effects increase towards the urban core, making fringe areas more and core areas less similar to the surrounding non-urban environment (McKinney 2002). A site's position in the landscape should thus influence our expectations for its potential naturalness (McKinney 2002).

Urban areas are characterized by the urban heat island effect. Temperatures are higher than in non-urban surroundings, as solar radiation is absorbed by built infrastructure with high albedos and re-emitted at longer wavelengths, heating the surface air (Parlow 2011). Urban areas also have higher precipitation levels, due to the increased amounts of aerosols from vehicle emissions in the urban atmosphere acting as condensation nuclei. The high proportion of impermeable ground cover and built infrastructure means that water is lost as run-off or pools on the surface in many areas, rather than being absorbed and retained by soil and roots (Illgen 2011). What water is available may be contaminated by air pollution and ground-level contaminants.

Urban soils often have higher levels of heavy metals, nutrients, and pollutants than non-urban soils, and have a higher pH and hydrophobicity (McDonnell et al. 1997; Alberti 2005; Newbound et al. 2010; Burton and MacDonald 2011). Soils may be compacted and shallow, which impedes their ability to absorb and retain water. Compaction occurs from human activities, such as trampling or mowing, which may also lead to greater rates of soil erosion (Zipperer and Guntenspergen 2009; Edmondson et al. 2011). Many urban soils are 'sealed', meaning they have been covered by an impermeable layer, and no longer have gas exchange or water infiltration capabilities (Sauerwein 2011). The low retention of water by soils negatively affects root growth and thus the potential for overall vegetation growth (Pavao-Zuckerman 2008; Millward et al. 2011). Soils are fundamental to the establishment and growth of the urban forest – some species may not be able to grow in contaminated soils, or see their growth rates reduced while more-tolerant species succeed.

Air pollution is much higher in urban environments due to emissions from anthropogenic activities (Parlow 2011). Carbon dioxide and nitrogen oxide are emitted from vehicles, industrial operations, and general human activity. Pollution negatively impacts many species: pollution-sensitive lichens may be absent from urban areas or close to industrial activities (Conti and Cecchetti 2001; Cristofolini et al. 2008), but have been shown to return following the reduction of pollution, particularly sulphur dioxide, levels (Rose and Hawksworth 1981). This same suite of human activities contributes to the high levels of noise pollution in urban areas, particularly at low frequencies. Noise pollution affects the ability of animals such as birds to communicate (Rheindt 2003; Nemeth and Brumm 2010; Halfwerk et al. 2011), and detracts from perceived naturalness and human experiences in urban greenspaces (Swanwick 2009; Peckham et al. 2013).

All together, the urban environment and its residents have been aptly described as a biotic filter (Williams et al. 2009). On their own, urban areas may not tend towards natural conditions due to environmental barriers to arrival and establishment (Doody et al. 2010). Species are differentially favoured and hindered by urban conditions, and thus succeed at different points along the urban-rural gradient (e.g. Lizée et al. 2011). Species richness is often highest in suburban regions; however, this community is a mixture of native and non-native urban adaptors, or, species able to tolerate some anthropogenic impacts (McDonnell et al. 1997; Crooks et al. 2004; Turner et al. 2005; McKinney 2008). Overall, native biodiversity declines in response to urbanization due to the pressures of the urban ecosystem (McKinney 2002; Colding 2011).

2.2 Naturalness

The term natural typically refers to a state unaffected by humans. Naturalness is the degree to which something resembles the natural state; it is thus a continuum (Machado 2004). Naturalization is the creation or transition of an urban space to one with compositional, structural, and functional similarity to nearby non-urban forests considered ‘natural’ references. It is an inherently contradictory process, as it may involve human influence and interference in existing conditions to foster naturalness, which some say compromises naturalness (Hunter 1996; Angermeier 2000). One solution has been to separate the idea of wildness from that of naturalness. Wildness refers to the absence of any and all human intervention in an ecological system, whereas naturalness refers to a location’s similarity to a desirable reference condition, which itself may have low levels of human influence (Cole 2000). Thus some spaces may require human intervention to maintain certain species or ecological processes due to stronger, detrimental anthropogenic influences; these spaces might then have some degree of naturalness but are not wild (Cole 2000; Higgs 2003). Conversely, spaces without human influence may be wild but not particularly natural due to the unfettered presence of many non-native species, such as the spontaneous vegetation in abandoned city lots.

Naturalness is a continuum. The urban forest spans all trees in the urban environment, and these situations can all be less or more natural from a site-specific perspective. The ways in which a boulevard is natural are fewer than those for a park containing remnant forest. Naturalization, too, occurs along a continuum of effort. In some settings, we may only be able to increase the naturalness of the tree canopy composition; in others, we may be interested in increasing habitat suitability for particular species through altering stand structure. The specific actions of naturalization are determined by both ecological and social and cultural factors (Higgs 2003; Gobster 2012). Depending on the site’s location within an urban setting and size, some ecological goals may be unrealistic. Depending on how humans use and value the space, some ecological goals may be undesirable.

2.3 Benefits and Disservices of Natural Areas

2.3.1 Anthropocentric Benefits

The anthropocentric benefits of natural spaces are important to emphasize when advocating for naturalization. We are interested in and aware of how systems and their changes impact us. Biocentric reasoning, or ascribing an intrinsic value to certain states and qualities of non-human systems regardless of their potential utility, is less common and potentially less meaningful (Kahn 2002). Extrinsic values, derived from the services we receive from the urban forest, drive understanding and attachment and thus advocacy for these values (Berenguer et al. 2005).

Natural spaces in cities are valued for providing feelings of tranquility and escape from the built city and immersion in the non-human world, and fostering feelings of contentment and well-being (Chiesura 2004; Swanwick 2009; Peckham et al. 2013). The feeling of immersion is both structural and aesthetic. Structurally, densely wooded areas create a visual screen, and larger areas provide more opportunity for a separation between the built city and the urban resident to develop (Kaplan 1985). Aesthetically, the sounds created by birds, the wind in leaves, and the path underfoot help urban residents connect with a non-human space and disengage from themselves and the urban lifestyle (Swanwick 2009; Peckham et al. 2013).

The potential economic services of natural spaces are not at the forefront of the urban resident's mind (Peckham et al. 2013). Potential benefits would include the decreased maintenance costs over time, despite potentially higher initial establishment costs. As spaces become more natural, management activities switch from tree planting and invasive species removal to lower intensity monitoring and passive management, such as retaining deadwood or not mowing (Green Seattle Partnership 2006). The latter activities require both less time and less money in the form of materials or labour. Naturalizing the streets and parks around one's house could increase property values, as certain types of natural areas are desirable components of a neighbourhood (Donovan et al. 2013). More-natural areas may better provide services such as stormwater control due to their greater capacity to uptake and store water. Some hypothetical economic benefits require city dwellers to actively use natural spaces – there may be decreased health costs associated with spending time outdoors, for instance, due to reduced stress levels (Lee and Maheswaran 2011; Donovan et al. 2013).

The feelings of tranquility and contentment urban residents experience in natural areas may be linked to perceived rather than ecological naturalness. Our experiences fuel our perceptions and vice versa (Chiesura 2004); what one person perceives as natural or native may be unnatural or non-native to another. For example, urban residents express higher feelings of well-being in areas they consider more biodiverse, even if these areas do not have the most appropriate or greatest biodiversity (Dallimer et al. 2012). The touted benefits may result primarily from structural naturalness, such as canopy cover. The structure of a park determines the degree to which visitors feel immersed in the non-human world (Kaplan 1985). It may not matter which species are creating that structure. This could be problematic if goals to increase ecological naturalness negatively impact elements of perceived naturalness (Gaston et al. 2013).

2.3.2 Ecological Benefits

The ecological naturalness of urban areas is strongly linked to soil and tree species composition. Soil characteristics such as litter layer depth and decomposition rates affect nutrient cycling and thus resource availability, and the physical structure affects hydrodynamics (Perry 1994; McDonnell et al. 1997). The availability of water and nutrients affect which plants are able to establish and grow, which in turn affects soil chemical composition (Perry 1994). What plant species are present, particularly with regards to trees, affects many other dimensions of naturalness, such as the potential for native regeneration, or whether suitable habitat or attractants are present for other flora and fauna (Corbin and Holl 2012). For instance, native insect abundance and species richness decline with urbanization due to a loss of the native plant food base (Raupp et al. 2010). Most plant-eating insects are highly specialized and can feed only on plants with which they share a co-evolutionary history (Blair and Launer 1997; Tallamy 2004).

Guilds of species that are unable to establish in less-natural urban conditions might find habitat in more-natural portions of the urban forest. More-natural portions of the urban forest support a greater diversity of native species (e.g. Melles et al. 2003; Raupp et al. 2010). Urban vegetation is often simplified, fragmented, and more open than many forest types; unsurprisingly, many forest interior and forest-dependent species are absent from urban areas (Chace and Walsh 2006; Soga et al. 2014). Urban areas encroach on natural habitat either directly, through expansion, or indirectly, through requiring the alteration or loss of natural areas for services or resources. Increasing the amount of potential habitat for native species, which may be declining outside of city limits due to human encroachment, is an important benefit of more-natural urban areas.

2.3.3 Disservices of Naturalness

Natural areas can appear unkempt or messy. While research on aesthetics indicates that more-natural areas are more desirable than built-up areas or those with greater levels of human influence, less-natural areas can be more visually attractive due to a greater degree of maintenance or the presence of particular ornamental species (Kaplan 1985; Jorgensen et al. 2002; Lindemann-Matthies and Marty 2013). The initial stages of naturalization in particular may be unattractive if they entail removing large numbers of invasive plants, leaving bare patches and/or stumps in previously vegetated areas.

Natural areas can create feelings of unease due to real or perceived danger (Talbot and Kaplan 1984). Urban residents identify heightened feelings of unease and fear of crime in some treed urban areas due to the perceived potential for concealed individuals or activities (Talbot and Kaplan 1985; Koskela and Pain 2000; Kuo and Sullivan 2001; Jorgensen and Anthopoulou 2007). Danger may also be present in non-human species that live in natural areas. Large predators such as coyotes or cougars can be supported in more-natural urban areas and pose threats to humans; smaller predators, such as coyotes or some birds of prey, might threaten pets and small children.

2.4 Naturalization

Vegetation recruitment differs in urban settings from non-urban reference forests. Plants and components of the seed bank may have arrived from direct or indirect human activity (e.g., from the large community of introduced species in the surrounding environment) (Sullivan et al. 2009). Urban areas may initially have slow natural regeneration due to difficult starting conditions (e.g., a highly altered soil seed bank and compacted soil), the surrounding built rather than natural environment, hindrances to vegetation development (e.g., mowing), and absence of some animal dispersers from urban areas or naturalizing areas at the outset (Robinson and Handel 1993; Millward et al. 2011).

Naturalization initiatives may plant clumps of trees and other woody species in the hopes that they will expand outwards and/or alter site conditions to attract new species (Corbin and Holl 2012). The plantings do not always act as regeneration sources but sometimes as habitat primers; natural regeneration often comes from the surrounding vegetation (Robinson and Handel 2000; Sullivan et al. 2009). Animal dispersers may be attracted to the site, and wind-borne propagules may be trapped by the vegetation clumps in otherwise open areas (Corbin and Holl 2012). However, the seedlings may be both native and non-native species (Ruiz-Jaén and Aide 2006). Since many of the non-native species that thrive in urban areas grow rapidly and have highly dispersive and plentiful reproductive strategies, such as wind-dispersed seeds, they may dominate the regenerating communities if these are not controlled (McKinney 2006).

When non-native species are present, regenerating urban areas are not the ecological equivalent of natural non-urban forests. Different community members play different roles and this may differentially magnify outwards (Miles 1979). However, the role of alien species in future successional stages is unknown since minimal long-term research exists. Little is known on how species plantings and management decisions affect urban forest composition and structure beyond the initial few years, as only a handful of studies have looked at periods of greater than five years (Oldfield et al. 2013). Evidence from short-term findings suggests that naturalization can beget further naturalization. Native tree species, once established, support more-diverse and -numerous bird and arthropod communities (Tallamy 2004; Chace and Walsh 2006). These animals may be involved in propagule dispersal, in altering competition among plant species (e.g., via herbivory), or in processes such as nutrient recycling or decomposition, further naturalizing the site. Through altering soil composition, attracting or trapping dispersers and propagules, native trees may create conditions more favourable to the establishment of other native flora (Corbin and Holl 2002).

The focus of naturalization is to alter the biophysical characteristics of a site. In doing so, there are desired social outcomes, such as a greater sense of place and connection with the natural world. Humans and non-humans alike benefit from increasing the ecological similarity between cities and their surroundings. The many benefits of naturalization should provide great impetus for its incorporation into urban forest management. Chapter 3 outlines the methods by which a framework for urban forest naturalness from an ecological perspective was created and tested. The framework hopefully provides managers with both a conceptual and practical tool to facilitate naturalization.

CHAPTER 3 METHODS

3.1 Framework Creation

The framework was generated through consulting two bodies of literature. The first, existing naturalness assessments in non-urban forests, was used to compile lists of dimensions of interest in non-urban settings. Naturalness assessments are common in non-urban forests, particularly in Europe (e.g. Winter et al. 2010; Liira et al. 2007), wherein a wide variety of forest components are compared between identified 'natural' reference sites and the sites of interest. One of their goals is often to identify the impacts (or lack thereof) of logging or management. The dimensions from non-urban naturalness assessments were compared with a second body of literature, studies on urban ecology. Papers included summaries of changes along urban-rural gradients and a conceptual framework on how urban biotic communities are shaped by different biotic filters (e.g., Pickett et al. 2001; Ruiz-Jaén and Aide 2006; Williams et al. 2009).

If dimensions previously used in non-urban naturalness assessments overlapped with features of the urban environment that changed along urban-rural gradients, these were included within the framework. Dimensions not clearly linked to urbanization were excluded. Some dimensions were modified to reflect our interest in nativeness. For instance, non-urban forest naturalness assessments often compare basal area to a most-natural reference point; we were interested in how much of that basal area was occupied by native species. Some dimensions were absent from non-urban forest assessments but are prominent components of urban ecosystems. These were determined both from existing urban ecology literature and through researcher reflection, and included in the framework.

Once the dimensions were assembled, the framework was organized into compositional, structural, and functional dimensions, as in Noss (1990) and McElhinny et al. (2005), and environmental dimensions:

- Compositional dimensions refer to species that are present and their abundances and assemblages;

- Structural dimensions refer to the physical arrangements, abundances, characteristics, and complexity of the vegetation;
- Functional dimensions refer to the occurrence and rate of ecological processes, such as the disturbance and gap dynamics of a forest stand;
- Environmental dimensions are external influences on a site resulting from its urban location.

Within each of the four categories, dimensions were organized into broad sub-categories (e.g., “3.3.1 Native tree species” contains three compositional dimensions). Each dimension was accompanied by support from the literature for its link to naturalness in an urban setting. The framework has no final or cumulative naturalness index score. Due to disinterest in producing an aggregate score, each dimension can have its own assessment and visualization; the impetus behind the framework is to dissect ecological complexity into its constituent parts to better understand and manipulate them, a task which would be undermined through combining the scores on each dimension into an abstract overall grade. The lack of a cumulative final score also related to the variety of dimensions. Some were conceived as proportions, some were absolute values, and some were binaries (e.g. yes/no). Similarly, no weighting was applied *a priori* to the dimensions; the weighting would be arbitrary, as the importance of each dimension is contextual.

3.2 Site Selection

Sixteen vegetated urban sites were selected in Halifax, Nova Scotia, and Winnipeg, Manitoba (Table 1). Halifax occurs in a humid maritime region with average precipitation levels of ~1400 mm/year (Statistics Canada 2007). The region is part of the Acadian forest, a diverse temperate forest with over thirty native tree species. Prior to the arrival of Europeans, it is estimated that over 50% of the province of Nova Scotia was covered by late-successional forests of red spruce, beech, sugar maple, eastern hemlock, yellow birch, and white pine (Mosseler et al. 2003). The region is varied in its topography, soil conditions, and climate, resulting in a mosaic of forest types and vegetation communities across the landscape. The Mi’kmaq historically occupied Halifax, and Europeans settled the region in the 1700s. This human presence lies behind many elements of the landscape – Point Pleasant Park, a 77-hectare park at the tip of the Halifax Peninsula, was occupied for millennia by the Mi’kmaq and logged by European settlers to make

Table 1 Urban forest sites visited in Halifax, Nova Scotia (NS), and Winnipeg, Manitoba (MB)

Site	Site code	X/Y coordinate	Area sampled (m^2)	Vegetation	Province
Ardmore Park	Ardmore	44°39'5.1"N 63°36'22"W	960	Sparse non-native trees (e.g. Norway maple)	NS
Assiniboine Park	AS	49°51'17"N 97°14'53"W	800	Aspen-oak forest	MB
Barrington & Cornwallis	BC	44°39'15"N 63°34'53"W	456	Sparse red maple	NS
Don Bayer Sports Park	Burnside	44°42'50"N 63°35'24"W	1200	Grass & low shrubs	NS
Churchill Drive	Churchill	49°51'21"N 97°7'51"W	1200	Sparse native trees (e.g. bur oak)	NS
Flinn Park (A)	FP A	44°38'30"N 63°36'44"W	766	Red/Norway maple-pin cherry	MB
Flinn Park (B)	FP B	44°38'30"N 63°36'44"W	800	Red maple-grey birch-English oak	NS
Fort Needham Park (east)	FN E	44°39'59"N 63°36'4"W	800	Norway spruce-mountain ash	NS
Fort Needham Park (west)	FN W	44°39'55"N 63°36'7"W	800	Norway spruce-mountain ash	NS
Hemlock Ravine	HR	44°41’28”N 63°40’14”W	1200	Old-growth hemlock	NS
McBeth Park (river)	MB R	49°57'17"N 97°4'51"W	800	Old-growth riparian forest (e.g. American elm; plains cottonwood)	NS
McBeth Park (inland)	MB F	49°57'19"N 97°4'58"W	800	Old-growth riparian forest (e.g. American elm; plains cottonwood)	MB
Point Pleasant Park	PPP	44°37’31”N 63°34’26”W	1200	Old-growth pine/maple	MB
Shubie Park	Shubie	44°42’09”N 63°33’24”W	1200	Old-growth hemlock	NS
St. Mary's Boat Club (forest)	SMBC F	44°38'13"N 63°36'9"W	1628	White birch-Norway/red maple with invasive understory	NS
St. Mary's Boat Club (waterfront)	SMBC W	44°38'16"N 63°36'16"W	1131	White birch-red maple with grassy sections	NS

way for a military base. Local natural benchmarks, in the sense of being without anthropogenic influence, are thus few and far between on the landscape.

Winnipeg occurs in a warm and relatively humid region of the prairies, where average annual precipitation is ~510 mm/year (Statistics Canada 2007). The region is a transition between the boreal forest to the east and aspen parkland to the west, and many portions were historically tall grass prairie. Native tree species are fewer and forest types more discrete within the city: trembling aspen and bur oak form much of the natural forest canopy interspersed with rough-fescue grasslands. Thus, many portions of the city would naturally be untreed. Trees are more abundant along the rivers, where wooded riparian areas with elm and cottonwood snake their way. The geographic distribution of parkland and prairie was influenced by routine burning by Aboriginal groups such as the Cree who occupied and traded in the area for millennia (Karst 1995). Europeans began settling in the area in the mid-1700s, interrupting and suppressing Aboriginal land management practices, allowing treed patches to expand.

The two cities embody very different foundations for urban forestry. In Halifax, the natural ecosystem in the absence of urbanization is a multitude of Acadian forest types. In Winnipeg, the natural ecosystem may be grassland, bur-oak aspen stands, or riparian forests. The forests differ in their diversity and species composition, structure, and disturbance regimes. Applying the framework to locations with very different natural ecosystems helps determine whether the framework is tied to place-specific conceptualizations of naturalness, or whether it could be widely applied. A further reason for selecting the two cities is that research on urban forest values has been carried out in four Canadian cities in recent years, including Halifax and Winnipeg (Peckham et al. 2013; Sinclair et al. 2014). The values-elicitation work demonstrates that both cities have citizens interested in and attracted to natural areas for similar reasons. The values research also highlights the social importance of some spaces, and how ecological goals may need to be tempered by these uses.

Lastly, I happen to call both cities home, and had easy access to accommodations and logistical support in both locations.

All sites were visited one to three times between June and October of 2014. Twelve sites in Halifax were chosen based on consultations with municipal and provincial employees working with urban and near-urban parks, and were visited in May prior to data collection to determine that the sites represented a broad range of forest conditions. The four old-growth sites were our candidates for the most-natural urban conditions in the Halifax region, whereas a park with the oldest trees in Winnipeg represented that city's most-natural urban end-point. The least-natural condition was an untreed field alongside a major roadway. Sites in Winnipeg were chosen using the City of Winnipeg (2014) parks database. Greenspaces in Winnipeg are assigned an overall 'grade' related to a vegetation inventory; four sites were chosen to represent a mixture of vegetation types and conditions. Tree canopy and deadwood data for four Halifax old-growth sites were obtained in the summer of 2014 from a concurrent research project, and these sites were visited in the fall of 2014 to collect data on ground flora and the soundscape.

3.3 Data Collection

One to three 20 m x 20 m plots were positioned within each urban site. The number of plots was determined by the size of the site: some could only contain one or two plots. If the site was larger than the area of the three plots, I intentionally positioned plots to capture relatively homogeneous portions of each site, if the site was larger than three plots. I wanted similar conditions between plots since naturalization activities would not be randomly distributed across a site, but tailored to particular locations and conditions. Thus random sampling to characterize the whole was not of interest. Within each plot, all trees greater than 1.3 m in height were identified and their diameter at breast height (DBH) was measured. Identification to species followed Scoggan (1957), Roland and Smith (1969), and Farrar (1995). All snags encountered within the plot were identified to species or genus (if possible), had DBH and approximate height measured, and were assigned to a decay class.

Ten 1 m x 1 m quadrats were systematically placed within each 20 m plot (Figure 1) (Epstein 2005). All vascular ground flora within the plots were identified to species or genus and their percent cover was visually estimated. Due to the difficulty in identifying grasses throughout the field season, grasses were treated as a single group and thus were not included in assessments. Identification followed Scoggan (1957) and Roland and Smith (1969).

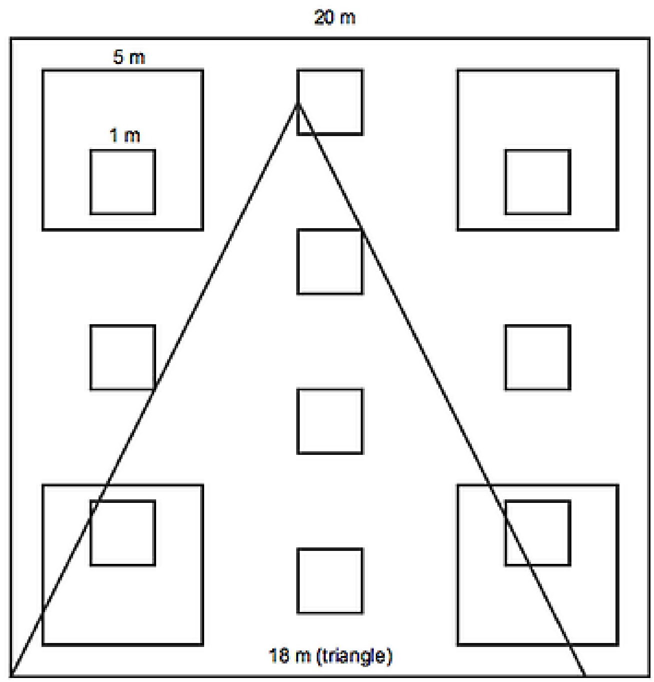

Figure 1 Diagram showing the lay-out of 5 m x 5 m plots, 1 m x 1 m plots, and 18 m-sided triangle within each 20 m x 20 m plot

Four 5 m x 5 m plots were positioned at the corners of each 20 m x 20 m plot (Figure 1). All tree regeneration (any tree less than 1.3 m in height) was identified to species or genus and counted. Seedling density was determined as an average among all 5 m x 5 m plots across all 20 m x 20 m plots; averages were thus calculated among 4 to 12 plots. If the site was smaller than the area of all four 5 m x 5 m plots, all seedlings within the entire site were counted (e.g. Barrington & Cornwallis), and the total area of the site was used to calculate a single value for density. Three 18-m transects were positioned within each 20 m x 20 m plot to create a 54-m triangle (Figure 1) (NSDNR 2003; Thompson 2004). Any downed deadwood with a diameter greater than or equal to 7.0 cm that intersected a transect was included and its diameter recorded.

3.4 Data Manipulation & Naturalness Scoring

Native proportion of tree canopy individuals: Each tree within the plots or site was identified to species or genus. Naturalness scores ranged from 0, if all trees were non-native species, to 1, if all trees were native species.

Native proportion of basal area: Basal area was calculated from diameter at breast height, measured on every tree at least 1.3 m tall using the formula:

$$(d^2 * \pi)/40\ 000$$

Where d = diameter at breast height (cm). Basal area was combined for all native and non-native species. Naturalness scores represent the proportion of basal area represented by native species and ranged from 0, if all basal area represented non-native species, to 1, if all basal area represented native species.

Native proportion of regenerating trees: Each seedling within the 5 m x 5 m plots was identified to species or genus. Individuals within ambiguous genera were excluded from the nativeness calculation (i.e., *Acer* sp. could be native or non-native), while individuals in non-native genera were included in the calculation. Naturalness scores ranged from 0, if all seedlings were non-native species or genera, to 1, if all seedlings were native species.

Proportion of ground cover occupied by native species: All non-grass vascular flora within the 1 m x 1 m plots were identified to species or genus. Individuals only identified to genera were excluded from the nativeness calculation if the genus included non-native species, while those in genera without any native members were included in the calculation. Naturalness scores ranged from 0, if the coverage consisted of all non-native species, to 1, if the coverage consisted of only native species.

Proportion of tree individuals associated with invasive alien species (IAS): IAS designations were determined through consulting the Canadian Wildlife Federation's *Invasive Species Encyclopedia*, a comprehensive online resource on invasive species in Canada (Canadian Wildlife Federation 2015). All trees within the 20 m x 20 m plots were identified to species. Naturalness scores ranged from 0, if all trees were invasive alien species, to 1, if no trees were invasive alien species.

Proportion of ground flora cover associated with IAS: IAS designations were determined through consulting the *Invasive Species Encyclopedia* (Canadian Wildlife Federation 2015). Ground cover was assessed in the 1 m x 1 m plots positioned within the larger 20 m x 20 m plots. Naturalness scores ranged from 0, if all ground flora cover consisted of invasive alien species, to 1, if no ground flora cover consisted of invasive alien species.

Tree canopy FEC concordance: Each urban site was keyed out to the lowest level possible within its provincial Forest Ecosystem Classification (FEC) (Zoladeski et al. 1995; Neily et al. 2013). Several forest stands could only be classified to forest type due to the absence of native ground flora; one site could not be keyed out at all due to the absence of a tree canopy. Concordance was calculated through comparing all tree species listed within the associated vegetation type and all tree species identified within the urban forest site. Any tree species present within the urban forest site but not listed in the FEC were considered 'discordant', and detracted from the overall concordance measure. Naturalness scores ranged from 0, if no species occurred on the FEC species list, to 1, if all species within the plots occurred on the FEC species list.

Density of canopy trees adjusted to time since stand-replacing disturbance: Density was extrapolated to the trees per hectare (ha) basis site-by-site. A minimum stand density of 1000 trees was considered natural, based on densities reported by Gower et al. (1997) and Stewart et al. (2003). All stands with a density of at least 1000 trees received a naturalness score of "1", and the others were assigned a score representing their proportion of the minimum density (e.g. 140 trees/ha was assigned a naturalness score of 0.14).

Proportion of ground covered by tree canopy: Canopy coverage was calculated using the online program iTree Canopy (v6. 1) (http://www.itreetools.org/canopy/index.php). Fifty random points were selected by the program within the boundaries of each site and were classified by myself as either "tree" or "non-tree". Naturalness scores ranged from 0, if there was no tree canopy coverage, to 1, if the entire plot was covered by tree canopy. Lower canopy cover could be considered fully natural in ecosystems not considered in this study.

Stage-adjusted number of distinct canopy layers: Trees were sorted into three categories by diameter: an overstory canopy (>20 cm DBH), a mid-story canopy (>10-20 cm DBH), and an understory/shrub layer (0-10 cm DBH). Naturalness scores were one of three categories: (1) *unnatural* sites received a score of 0, if one canopy category was absent from the plot; (2) *semi-natural* sites received a score of 0.5, if all three canopy categories were present but in unexpected ratios (e.g., where the understory/shrub layer was not the most abundant); and (3) *natural* sites received a score of 1, where all three canopy categories were present in expected ratios (understory > mid-story > overstory). I determined the category classes based on the forests in which the framework was tested; other forest types would have required different, regionally appropriate distinctions.

Shrub layer FEC concordance: Each urban forest site was keyed out to the lowest level possible within its provincial Forest Ecosystem Classification (FEC) (Zoladeski 1995; Neily et al. 2013 2011). Stands that could be keyed out to a particular vegetation type within a forest type were assessed for the concordance of its shrub layer with the FEC. Shrub species were (a) those listed within the FEC and (b) those with perennial woody growth considered as shrubs elsewhere. All shrub species listed with a particular vegetation type were compared with all shrub layer species identified with the urban forest site. Any species present in the urban forest site but not listed in the FEC were considered 'discordant', and detracted from the overall concordance measure. Naturalness scores ranged from 0, where no shrub species present at the site were on the FEC species list, to 1, where all species present at the site were on the FEC species list.

Stage-adjusted presence of trees considered large/old for the species: Size (represented by DBH) and age at maturity for each tree species encountered in our plots were determined through consulting the US Forest Service hardwood and softwood silvics database (Burns and Honkala 1990). Any tree with a DBH over what was listed as "mature" within the silvics database was considered large/old. Professional judgment was extended to include those trees that are regionally large or old when the information within the US Forest Service website did not necessarily support such a designation. If large/old trees were present, the site received a score of 1, whereas their absence resulted in a score of 0.

Standing and downed deadwood volume: All snags encountered within the 20 m x 20 m plots were measured and counted. All pieces of downed woody debris that intersected the 54-m transect with a minimum diameter of 7 cm were measured. The volume of all downed woody debris and snags was calculated using the formula (van Wagner 1968):

$$V = \pi^2 * \Sigma (d^2/8L)$$

Where, V = volume of wood per unit area (m^3/ha);

d = diameter of downed CWD at intersection (cm), and;

L = length of sample line (m)

The recorded deadwood volumes were compared to the average volumes across a variety of Nova Scotia forest types published in Townsend (2004) and the Nova Scotia Department of Natural Resources (NSDNR) website. Townsend (2004) recorded an average volume of coarse woody debris of 20.43 m^3/ha, and the NSDNR website recorded an average of 21.5 m^3/ha. Stewart et al. (2003) found that deadwood and snag volumes varied from 45 to 91 m^3/ha for deadwood and 17-57 m^3/ha for snags, but the researchers were only assessing old-growth forests. An assessment of old, ecologically healthy riparian forest in Manitoba found an average volume of 20.89 m^3/ha (Martinson et al. 2008). A minimum volume of 20 m^3/ha was used for the present study. Any volume equal to or greater than the minimum received a naturalness score of "1"; all volumes below were assigned a score representing their proportion of the minimum.

Potential to support interior conditions: A site had the potential to support interior habitat if any point within the overall site was 50 m or more from an anthropogenic edge (i.e. sidewalk; development). Moffatt et al. (2004) found that interior conditions existed at 50 m from a forest edge, and Vallett et al. (2010) used 45 m. Unpaved footpaths were not considered to break up interiorness, though edge effects have been recorded along trails (Godefried and Koedam 2004); site features under a canopy, such as a footpath, were not visible in remotely sensed data and thus could not be considered.

Frequency and duration of anthropogenic sounds: The soundscape was listened to for three intervals throughout the site visit (at the start, middle, and end). The identity (e.g. mowing; dog barking), relative persistence (i.e. how much of the two-minute period did it occupy), and predominance (i.e. how loud relative to other sounds heard at that moment) of the sounds heard at the site were noted to create an overall impression of the soundscape. Based on the pre-dominance of different sounds, sites were categorized as having "human" or "non-human" soundscapes. All sites were visited during weekday mornings or afternoons, to minimize the impact of the rush and lunch hours (i.e. when there is an assumed increase in traffic) of the nine-to-five workweek. Only two broad categories were established to account for the low sampling rate of this subjective and dynamic phenomenon.

3.5 Data Analysis

After determining how to score each site on each dimension on a scale from 0 to 1, the correlation between scores on every pair of dimensions was calculated. Correlations greater than 0.8 were of interest, as I was interested in relationships that were highly correlated to a point of redundancy, and not simply significance.

3.6 Limitations

Sites were not selected randomly but intentionally to represent a range of urban settings. However, I could have compiled many sites for each condition of interest, and then selected several sites of each kind using stratified random sampling. It is possible that biases in site selection pre-dispose our results to sort into particular categories. However, some site types are unique within each of the cities. Assiniboine Forest is the largest urban and largest most-natural forest in the City of Winnipeg; it has no parallel. Similarly, there are few old-growth urban sites in Halifax, and we sampled a subset of them. Based on a concurrent study (Toni et al. [in prep]), the old-growth sites not considered in this study were as natural as the ones we did assess in terms of canopy composition and deadwood volume. Thus we do not feel the subset we selected influenced our conclusions on the end-point for urban naturalness.

It is possible that selecting other sites would have resulted in more than three broad management groups (see section 5.5). For instance, we might have attempted to represent a broad range of vegetation types, rather than focusing on forest conditions (e.g. presence/absence of native canopy and understory) However, as we did not go into the study with a pre-conceived number of groups or a notion that sites would form groups, our results were not consciously shaped by our study design.

We were limited by our own knowledge in assessing dimensions. For example, as neither of us in an entomologist, a detailed inventory of the invertebrate communities would have been a difficult task. Our selection of dimensions, while motivated by the belief that vegetation forms a template for the remainder of forest components, was also shaped by our strengths as researchers. Other dimensions we did not assess might have revealed different patterns of naturalness, or suggested other promising avenues for naturalization.

CHAPTER 4 NATURALIZATION FRAMEWORK

Toni S and Duinker PN. 2015. A framework for urban-woodland naturalization in Canada. Environ. Rev. **23**: 1-16. dx.doi.org/10.1139/er-2015-0003

4.1 Introduction

Urban forest naturalization has gained momentum within a variety of institutions and organizations (Evergreen 2001). Urban forests are historically the realm of the non-native and the manicured; as of late, a broad range of organizations and academics has become interested in establishing the species assemblages and ecological processes seen in more-natural forests. Community groups are spearheading participatory restoration projects (Evergreen 2001); homeowners are embracing 'ecological' gardening by planting native species and eliminating their mowed lawns (Lindemann-Matthies and Marty 2013); and urban forest master plans in Canada are directing an increase in the naturalness of the urban forest through planting forest-like stands and native species (Ordóñez and Duinker 2013). As the momentum for naturalization increases, so does the impetus for exploring its conceptual and practical dimensions. What does it mean for an urban forest to be natural? How do we assess and manage naturalness? And how does the naturalness of the urban forest affect city residents?

The term natural typically refers to a state unaffected by humans. Naturalness is the degree to which something resembles the natural state; it is thus a continuum (Machado 2004). Naturalization is the creation or transition of an urban space to more-natural characteristics. It is an inherently contradictory process, as it may involve human influence and interference in existing conditions to foster naturalness. Cole (2000) divided the notion of naturalness from wildness. Wildness to Cole (2000) is the absence of human influence, whereas naturalness is the similarity of a site to a state unaffected by humans. Human influence may thus be used to create a natural but not a wild state.

Urban residents identify stronger feelings of contentment and tranquility in spaces they consider more natural and biodiverse (Dallimer et al. 2012; Peckham et al. 2013) – even if these spaces do not, in fact, have the highest levels of or most appropriate biodiversity. Our experiences fuel our perceptions and vice versa (Chiesura 2004); what one person perceives as natural may be unnatural to another. Human perceptions of urban forests are related to their ability to identify common species, and thus are a product of their education on and exposure to the outdoor world (Dallimer et al. 2012). This creates an important role for urban forests. As the first and foremost (and sometimes only) interaction urban residents have with the natural world outdoors, both the perceived and actual ecological characteristics of urban green spaces can help shape human preferences and ideas of nature (Sinclair et al. 2014).

Urban residents may be responding more to an ecosystem dominated by trees in particular patterns than to the species composition of that ecosystem. Many urban residents are unable to identify native species to the same degree as professionals (Fischer et al. 2014). Naturalization could help directly or indirectly foster these connections with regional biota. A small, self-selected group of urban residents interested in the urban forest is drawn towards restoration projects, tree plants, and local stewardship groups (Evergreen 2001). These projects provide opportunities for education on regionally appropriate species and ecosystems. For the remainder of urban residents, environmental education could occur more obtusely. In making native species and forests the norm, urban residents unconsciously develop a sense of place tied to their regional ecosystems. The species, patterns, and processes seen in their urban forest become what they expect and identify with. In the absence of naturalization, these relationships can lie with alien and/or invasive plant and animal species, and with manicured and managed forest structures (Gobster 2012).

While the general public may differ in their perception of naturalness from professionals, they are not uninterested in conservation and environmental management. The public and professionals share a common line of reasoning between how they feel about a topic and what they believe should be done; it is only the content of their beliefs (i.e. whether they consider a species native or natural) that differs, which influences their choice of action (Fischer et al. 2014).

It is true that urban areas may only offer a semblance and subset of the regional forest due to the well-documented environmental impacts of urbanization, such as the urban heat island effect, increased precipitation, and altered soil chemistry (Pickett et al. 2001). Different forests grow in different environments. However, a primary aim of naturalization is to reintegrate the urban forest to a greater extent into its surroundings, increasing the habitat available for native species. These environmental differences must be accounted for and mitigated or integrated. Simultaneously, naturalization can aim to create forested areas accessible to the public, albeit altered from their most-natural incarnations, where interest in and attachment to the natural world could develop.

The objectives of this paper are threefold: (1) to outline the concepts underlying naturalization and how they influence naturalization decisions and goals; (2) to develop a framework of urban forest naturalness; and (3) to explore potential applications of naturalness assessments in urban forest naturalization. In urban areas, social motivations and consequences are as prominent a component of ecosystems as the biophysical. There are usually many competing interests associated with any green space, of which reinstalling a high degree of ecological naturalness may be but one. People may want access for running, dog walking, sports, picnics, or use as a thoroughfare. People may want a forest in an area that does not naturally support forests. Naturalization is a process of working towards the best possible outcome as determined by both ecological and cultural factors (Higgs 1997). Our framework will focus on ecological naturalness. It is up to local decision-makers to determine which dimensions are of interest in their setting and can be feasibly improved.

4.2 Concepts Underlying Naturalization

4.2.1 Naturalization

Naturalization is a form of ecological restoration with prominent social dimensions (Gobster 2012). For purely biological goals, a desirable outcome for naturalization would be a forest ecologically indistinguishable from undeveloped non-urban woodland. However, this ideal is altered by safety concerns (natural disturbances may threaten humans and infrastructure), human access demands and impacts (paths can fragment woodlands and facilitate the arrival of alien species), and the feelings humans hold towards different species and landscapes (Pickett et al. 2001; Jorgensen et al. 2002; Gobster et al. 2007). Eradication may be desirable for some native species, such as mammalian predators, or undesirable for some aliens, such as attractive birds or plants. Humans act as a biological filter in urban environments, favouring certain species and types of species (Williams et al. 2009).

Clewell and Aronson (2013) detail three perspectives on ecological restoration, titled the legacy, utility, and recovery models. The environmental legacy model is an attempt to return sites to their "preimpairment" condition in terms of biodiversity patterns and ecosystem processes. This approach is rarely considered within ecological restoration as of late, due to its assumptions about ecological stability. The environmental utility model is an attempt to repair specific ecosystem goods and services that are valued by people. These services may be habitat provision, water retention, or enhanced recreation. Ultimately, Clewell and Aronson (2013) advocate for the third perspective, the ecological recovery model. This perspective advocates for the creation of self-sustaining, resilient ecosystems that both resemble historic ecosystems to the greatest degree possible, as determined by modern-day and site-specific constraints, and provide a suite of services, many of which may be valued by humans. Our initial motivations in naturalization may initially fall under the utility model – we want more-natural spaces for human enjoyment and experiences, and/or for habitat provisioning for particular species. Our goals may be best achieved through visualizing naturalization through the recovery model.

Cultural influences determine not only how natural a space is allowed to become, but also how natural a space is perceived as being. For a typical urban resident, the perceived level of management and anthropogenic influence are the critical determinants of how natural a space is deemed (Peckham et al. 2013; Sinclair et al. 2014). Naturalness is related to a sense of immersion within a space and a simultaneous disconnect from the built city. Perceived naturalness is derived from prominent structural components such as canopy cover and the sensory environment (i.e. light and noise pollution) (Dallimer et al. 2012).

Perceived naturalness differs from ecological naturalness. Much debate occurs over whether humans can create or manage natural areas, or whether doing so inherently compromises naturalness (Hunter 1996; Angermeier 2000). One solution has been to separate the idea of wildness from that of naturalness. Wildness refers to the absence of any and all human intervention in an ecological system, whereas naturalness refers to a location's similarity to a desirable reference condition (Cole 2000). Some spaces may require human intervention to maintain certain species or ecological processes due to stronger, detrimental anthropogenic influences; these spaces might then be natural but not wild (Cole 2000; Higgs 2003). Conversely, spaces without human intervention may be wild but lack naturalness due to the unfettered presence of many non-native species, such as abandoned city lots.

In naturalization, human influence is inherent and unavoidable. Our effort can be put towards relinquishing control, and potentially fostering wildness through increasing naturalness. Naturalness, however, is the goal. We cannot devalue spaces simply because of a human hand; urban areas are only expected to grow in size, and our thinking must similarly grow to view the role of cities in the broader ecological landscape (Alberti 2010). The aim of naturalization is that urban forests can more closely replicate or foster communities and processes as seen in the surrounding undeveloped woodlands, thus offering a greater chance of human interaction and integration with regional biota.

4.2.2 The Ecology of Naturalization

Many studies are on urban restoration or afforestation, rather than 'naturalization' (Oldfield et al. 2013). Regardless of the terms, these studies and results are highly applicable to naturalization, as they deal with the mechanics of site preparation, the consequences of different planting schemes in terms of pattern or species composition, and the landscape influence.

Vegetation recruitment differs in urban settings from non-urban forests. Plants and components of the seed bank may have arrived from direct or indirect human activity (e.g., from the large community of introduced species in the surrounding environment) (Sullivan et al. 2009). Urban areas may initially have slow natural regeneration due to difficult starting conditions (e.g., a highly altered soil seed bank and compacted soil), the surrounding built rather than natural environment, hindrances to vegetation development (e.g., mowing), and absence of some animal dispersers from urban areas or naturalizing areas at the outset (Millward et al. 2011; Robinson and Handel 1993). Some studies have found that urban forests recovering independently from a hurricane are biologically indistinguishable from regenerating non-urban and undisturbed forests (Burley et al. 2008), or that the number of naturally regenerating seedlings in an urban area is on par with that of non-urban regeneration (Lehvävirta and Rita 2002). However, since the former study took place in a large, old, previously densely forested park, and the latter in the "most natural" subset of forests at least 84 years of age located within the city, their demonstrated levels of natural regeneration may not be relevant to recently treed, naturalizing stands.

Urban restoration initiatives may plant clumps of trees and other woody species in the hopes that they will expand outwards and/or alter site conditions to attract new species (Corbin and Holl 2012). The results of restoration studies are important to consider in naturalization decision-making, as they indicate what might independently occur after initial human interventions. The plantings do not always act as regeneration sources but sometimes as habitat primers; natural regeneration often comes from the surrounding vegetation (Robinson and Handel 2000; Sullivan et al. 2009). Animal dispersers may be attracted to the site, and wind-borne propagules may be trapped by the vegetation clumps in otherwise open areas (Corbin and Holl 2012). However, due to the potpourri that is the urban species pool, the seedlings may be both native and non-native species (Ruiz-Jaén and Aide 2006). Since many of the non-native species that thrive in urban areas grow rapidly and have highly dispersive and plentiful reproductive strategies, such as wind-dispersed seeds, they may dominate the regenerating communities if these are not controlled (McKinney 2006).

When non-native species are present, regenerating urban areas are not the ecological equivalent of natural non-urban forests. A site's species composition is influenced by environmental conditions and the positive or negative influence of other species: the species that are able to arrive and establish in turn influence future establishments by altering site conditions and interacting with potential neighbouring species (Miles 1979; Mascaro et al. 2013). Different community members play different roles and this may differentially magnify outwards. However, the role of alien species in future successional stages is unknown since minimal long-term research exists. Little is known on how species plantings and management decisions affect urban forest composition and structure beyond the initial few years, as only a handful of studies have looked at periods of greater than five years (Oldfield et al. 2013). Since not all species arrive within the first few years or successfully establish upon arrival, and since natural systems are continually dynamic, short-term studies only begin to capture vegetation dynamics in naturalization. Succession is a process occurring over decades or centuries.

Evidence from short-term findings suggests that naturalization can beget further naturalization. Native tree species, once established, support more-diverse and -numerous bird and arthropod communities (Tallamy 2004; Chace and Walsh 2006). These animals may be involved in propagule dispersal, in altering competition among plant species (e.g., via herbivory), or in processes such as nutrient recycling or decomposition, further naturalizing the site. Herbivores play a dual role in the assessment of success – their presence can itself be a component of ecological integrity if they are native species, and they can interact with plants as dispersers or consumers, shaping vegetation patterns. Through altering soil composition, attracting or trapping dispersers and propagules, native trees may create conditions more favourable to the establishment of other native flora (Corbin and Holl 2002). Many, though certainly not all, urban alien plant species deemed 'urban exploiters' are characterized as shade-intolerant, fast-growing generalists that would do poorly under a closing canopy (McKinney 2002). Naturalization is a dynamic process of reintegration. Over time, intolerant alien species may simply be pushed out of naturalizing systems as the clustered planting of woody species creates relatively damp and shaded microsite conditions (Ruiz-Jaén and Aide 2006). Conversely, if alien and invasive species persist, they may create conditions suitable for other alien species and further reduce naturalness (McKinney 2006).

The urban landscape also influences the potential naturalness of a site. Urban areas are a unique habitat type with a global distribution. They have a large amount of impermeable surface, communities of alien and native species, high degrees of fragmentation, higher temperatures than surrounding non-urban areas, low shade, and highly alkaline and dry soils with altered nutrient inputs (Thompson and McCarthy 2008; Williams et al. 2009; Alberti 2010; Lundholm and Richardson 2010; Werner 2011). Just as a mixed hardwood forest would be difficult to establish in a peat bog, achieving ecological integrity to non-urban standards in urban areas may be impossible beyond a certain level, with or without human intervention, due to environmental constraints. Furthermore, a site's location within the urban environment can influence its potential for non-urban levels of ecological integrity. Some effects increase towards the urban core, making fringe areas more similar to the surrounding non-urban environment (McKinney 2002). A site's position in the landscape should thus influence our expectations for its potential naturalness (McKinney 2002).

4.2.3 Why Naturalize?

Urban forests are socio-ecological: human desires and values, among which may be a desire for greater ecological integrity, are balanced with ecological processes (Gobster 2012). Residents often seek out 'natural' spaces within cities for restorative experiences (van den Berg et al. 2007). Studies have found that participants' blood pressure drops and their ability to complete attention-requiring tasks is heightened after time in or even after looking at photos of non-human environments (van den Berg et al. 2007). People identify feelings of calmness and restoration in more-natural areas (Peckham et al. 2013). Research on aesthetics indicates that more-natural areas are more desirable than built-up areas or those with greater levels of human influence, though not unconditionally (Jorgensen et al. 2002; Lindemann-Matthies and Marty 2013): less-natural areas can be more visually attractive due to a greater degree of maintenance or the presence of particular ornamental species; less-natural areas may be considered safer, with better sightlines and lower potential for interacting with wild animals; less-natural areas, such as mowed fields, provide spaces for gathering and recreation.

Urban residents identify habitat provision and conservation of local flora and fauna as values for naturalizing urban spaces (Peckham et al. 2013). However, since most urban residents cannot identify common species, it cannot be said that they differentiate among and appreciate particular species for their nativeness; perceived naturalness may be driving their experience (Dallimer et al. 2012). Naturalizing the urban forest could promote an ecological aesthetic. If ecologically healthy forests were accessible to urban residents, the forests could function as teaching tools if properly managed and provide less-ecologically impoverished experiences. This would feed back into the already-present valuation of conservation and habitat provision and strengthen its foundation.

The urban forest comprises a broad spectrum of types of tree situations. All trees within the urban area are part of the urban forest, whether they occur in forest-like parks or as lone individuals on a median. There are manicured parks and botanical gardens, abandoned properties with natural regeneration, remnant old-growth forests from before urbanization, treed streets, and everything between. All could contribute their part to the naturalness of the urban forest as a whole. A native tree species, even a lone tree in a built-up world, might still provide habitat to fauna that an alien species cannot.

The goal of naturalization is to move some or all dimensions towards more-natural states, based on circumstance and resources. Through dissecting naturalness into individual components, more-directed actions could be taken in areas of greatest concern or interest (Brūmelis et al. 2011). This results in more cost-effective management or restoration attempts, and a greater chance of achieving desired results.

4.3 The Framework

4.3.1 Framework Development

Naturalness assessments are not a novel concept for non-urban forests (Parkes et al. 2003; McRoberts et al. 2012). Nor is compiling comprehensive lists of features by which to assess forest structural complexity (McElhinny et al. 2005) or biodiversity (Noss 1990). However, a framework for urban forest naturalness assessment is lacking. Noss (2004) proposed establishing gradients along which urban restoration success could be evaluated, and to our knowledge no one has taken up the charge.

Our framework represents the application of accumulated knowledge on the ecological effects of urbanization to urban forest management (Table 2). The framework was generated through consulting two bodies of literature. The first is existing naturalness assessments in non-urban forests, from which we compiled lists of dimensions of interest in non-urban settings. These were compared with the second body of literature, studies on urban ecology. The papers include summaries of changes along urban-rural gradients and a conceptual framework on how urban biotic communities are shaped by different biotic filters (e.g., Pickett et al. 2001; Ruiz-Jaén and Aide 2006; Williams et al. 2009). If dimensions previously used in non-urban naturalness assessments were demonstrated to change with urbanization, these were included within the framework. Dimensions not clearly linked to urbanization were excluded. Some dimensions are absent from non-urban forest assessments but are prominent components of urban ecosystems.

Table 2 Dimensions of urban forest naturalness

Category	Sub-category	Dimension
Composition	Native tree species	Native proportion of tree canopy individuals
		Native proportion of basal area
		Native proportion of regenerating trees
	Native non-tree flora species	Native proportion of ground cover
		Stage-adjusted composition of epiphytic lichen functional guilds
	Native fauna	Proportion of expected native non-avian vertebrate species absent
		Proportion of expected native avian species absent
		Proportion of expected native invertebrate species absent
		Native proportion of non-avian vertebrate species
		Native proportion of avian species
		Native proportion of invertebrate species
	Invasive alien species (IAS)	Proportion of tree cover associated with IAS
		Proportion of ground flora cover associated with IAS
		Presence of invertebrates associated with IAS
	Concordance of tree canopy composition with regional FEC classification	Proportion of the tree canopy that is listed as possible canopy components associated with the soil/site type based on the prevailing regional forest ecosystem classification (FEC)
Structure	Overstory density	Density of canopy trees adjusted to time since stand-replacing disturbance
	Canopy cover	Proportion of ground surface covered by tree canopy
	Canopy layers	Stage-adjusted number of distinct canopy layers
	Tree size	Shape of the diameter distribution
		Stage-adjusted presence of trees considered large/old for the species
		Inhibition of tree form by cultural practices
	Tree location	Degree of randomness of tree location
	Coarse deadwood	Stage-adjusted volume of standing and downed deadwood
	Soil composition	Presence of intact A and B soil horizons and humus forms

Category	Sub-category	Dimension
	Soil composition	Soil seed bank composition
		Soil chemical composition
Function	Anthropogenic intrusion	Perceived naturalness
		Proportion of stand occupied by built infrastructure and paths
		Proportion of stand occupied by manicured grass
	Adjacency & connectedness	Isolation within the urban matrix
		Distance from non-urban sources of dispersers
	Natural regeneration	Stage-adjusted density of seedlings
		Evidence of natural disturbance patterns
	Interiorness	Proportion of site considered interior
Environment	Freedom from human-made toxins	Presence of indicator lichen groups
	Light pollution	Nighttime light intensity
	Noise pollution	Frequency and duration of anthropogenic sounds

These were determined both from existing urban ecology literature and through researcher reflection, and included in the framework. For example, urban trees are more affected by human practices such as pruning or pollarding, but this difference is not captured in the result of any study.

4.3.2 Rationale for Framework Structure

The incorporation of ecological principles and knowledge into urban planning can help create and manage spaces best suited socially and ecologically to their urban environment (Niemelä 1999). While naturalness is idealized in particular states of nature, such as ecologically healthy, old non-urban forests, this is only one end of a continuum. Stands can be more or less similar to these ideals, and there are spatially and temporally natural patterns of vegetation for any given site. In identifying individual dimensions that can be visualized and manipulated, urban foresters or any group interested in naturalization can determine the current and potential characteristics of a site, for both specific components and overall naturalness. Furthermore, only some components of naturalness may be of interest or feasible to manipulate.

Studies on urban-rural gradients are classically designed along gradients of built area (Fontana et al. 2011, after McDonnell and Pickett (1990)). The most artificial end of the gradient is a completely built or 'sealed' environment. Naturalness lacks finite, most-natural end points, but more-natural states are often embodied in nearby non-urban forests. Forests near urban areas indicate what is currently occurring in the region though not necessarily what is ultimately possible, since these communities are also often constrained by human factors. Many non-urban forests have a history of human modification, such as understory burning by indigenous populations (e.g., Bjorkman and Vellend 2010), or the conversion of land into plantations of non-native species (Brūmelis et al. 2011). Forests near urban areas are likely to have redeveloped following logging, or from abandoned farmland.

In considering the range of more-natural references for urban forests, both non-urban and urban remnant forests should be considered. Non-urban forests are relatively well researched, often with forest ecosystem classification schemes with characteristic features that distinguish forest types, and extensive lists of flora and fauna (e.g., Neily et al. 2013). Referencing these or their underlying data can indicate the idealized potential for a site. Comparisons with urban forests without a history of intensive human development are also informative. These can indicate how forests have responded to urbanization and thus the overall potential for urban naturalness: which species have been lost or gained, or alterations in age classes or overall physical forest structure.

Our framework was created based on our experiences with Canadian forests. Thus there are no urban settings older than several hundred years, and reference ecosystems are relatively abundant and well known. Some of these reference ecosystems are non-urban, but necessarily non-human landscapes: Aboriginal populations have been modifying the landscape of Canada for millennia (e.g. burning land for increased berry production or to clear trails (Davidson-Hunt 2003)), the appreciation of which is still developing in settler populations today.

Many dimensions in our framework exist along a continuum of artificial to hypothetically most-natural. Ruiz-Jaén and Aide (2006) assessed and visualized the restoration success of a reforested riparian area along eleven variables of composition, structure, and function. Our framework visualizes naturalization similarly, with each dimension as a slider along which sites can be placed and then moved. Some dimensions can be directly manipulated, such as the native proportion of different flora communities, whereas others are responsive, such as epiphytic lichen diversity (Liira et al. 2007). This distinction is not uncommon in restoration. Studies often adopt a "build it and they will come" approach to faunal communities (Palmer et al. 1997). This approach views vegetation as a template, and assumes that native faunal communities will be attracted to and supported by areas in which native vegetation (both composition- and structure-wise) has been re-established.

The framework is organized into compositional, structural, and functional dimensions, as in Noss (1990) and McElhinny et al. (2005), and environmental dimensions:

- Compositional dimensions refer to what species are present and their abundances and assemblages;
- Structural dimensions refer to the physical arrangements, abundances, characteristics, and complexity of the vegetation;
- Functional dimensions refer to the occurrence and rate of ecological processes, such as the disturbance and gap dynamics of a forest stand;
- Environmental dimensions are external influences on a site resulting from its urban location.

The first three types of dimensions are interrelated, as structure can beget function, and certain species are only present when certain processes are occurring (Noss 1990; McElhinny et al. 2005; Brūmelis et al. 2011). Some traits of forests fall within multiple categories: regeneration is structural, in terms of stand density and the distribution of age classes and diameters; compositional, in terms of the species array; and a process. Environmental dimensions have an over-arching though nonreciprocal relationship with other dimensions, as they affect composition, structure, and function.

Within each of the four categories, dimensions were organized into broad sub-categories (e.g., "3.3.1 Native tree species" contains three compositional dimensions). Each dimension was accompanied by support from the literature for its link to naturalness in an urban setting. The framework had no final or cumulative naturalness index score. Due to disinterest in producing an aggregate score, each dimension can have its own assessment and visualization; the impetus behind the framework is to dissect ecological complexity into its constituent parts to better understand and manipulate them, a task which would be undermined through combining the scores on each dimension into an abstract overall grade. The lack of a cumulative final score also related to the variety of dimensions. Some were conceived as proportions, some were absolute values, and some were binaries (e.g. yes/no). Similarly, no weighting was applied *a priori* to the dimensions; the weighting would be arbitrary, as the importance of each dimension is contextual.

Finally, the framework is intended to be comprehensive without being exhaustive. We view it as a conceptual framework to guide management thinking; not all dimensions are of interest or potentially even feasible for assessment or manipulation. Determining which invertebrate species are absent would be impractical for managers, but still a component of naturalness. At the end of our framework, we outline a test application of the framework to urban forest management, and how different subsets of the dimensions may be of interest to different naturalization projects.

4.3.3 Compositional Dimensions

4.3.3.1 Native tree species

i. Native proportion of tree canopy individuals

A fundamental component of naturalness in the urban forest is the composition of the tree canopy. Stands in the urban forest often consist of only a tree canopy, due to the maintenance of lawn in lieu of an understory. What tree species are present affects all other dimensions of naturalness: the potential for native regeneration, whether suitable habitat or attractants are present for other flora and fauna, and the chemical composition of the litter layer and soil.

A greater overall richness of native species is not necessarily indicative of greater naturalness. Studies contrasting natural, semi-natural and residential areas in urban areas do not find the greatest number of native species in natural areas but instead in areas with what are deemed middling levels of urbanization (Turner et al. 2005). This relates to the intermediate disturbance hypothesis, or, the notion of an intermediate level of ‘disturbance’ from urbanization resulting in a peak in plant species richness, attributed to high spatial and temporal heterogeneity (McDonnell et al. 1997; McKinney 2008). Instead, the proportion of native species in the community is indicative of naturalness, as it captures both the presence of natives and the absence of aliens (Gibbons et al. 2008).

There are fewer and less dominant native tree species in urban forests than in more-natural woodlands, where alien species are often entirely absent (McDonnell et al. 1997; Moffatt et al. 2004; Turner et al. 2005; Gibbons et al. 2008; Le Roux et al. 2014). While it has been argued that the absence of aliens is not itself a sign of naturalness (LaPaix et al. 2009), this creates a paradox wherein the presence of aliens is a sign of reduced naturalness but their absence not particularly informative. This framework views aliens as detractors to naturalness, and a greater native proportion of tree individuals as an indicator of greater naturalness.

One of the difficulties in studying the urban tree canopy is deciding whether to distinguish cultivars. The naturalness of a site is decreased through the novel genes and associated attributes of bred plants. However, if one is not concerned with naturalness at the genetic level, native cultivars bred for urban environments may even be desirable due to their increased likelihood of survival and health (e.g., increased ability to compartmentalize wounds), or their sex (e.g., absence of odorous or prolific propagules) (MacFarlane and Meyer 2005). Furthermore, obtaining regionally appropriate genotypes for native trees may be difficult or impossible from supplier nurseries, and the identification of cultivars can be prohibitively time-consuming or impossible (e.g., Hope et al. 2003).

ii. Native proportion of basal area

The dominant species in a forest stand are not necessarily captured in studying numerical abundance. A numerically dominant species could be present as many small, regenerating trees in the canopy of fewer but larger and more physiognomically dominant trees of a different species. Urban forests typically have a high representation, both in individuals and in basal area, of non-native species (Turner et al. 2005). Therefore, the greater the native proportion of tree basal area, the greater the naturalness of the site.

iii. Native proportion of regenerating trees

Seedlings represent the future potential of the canopy: if only non-native species are present as seedlings, intervention, both in non-native removal and native establishment, would be necessary to achieve a more-natural tree canopy in the future. Since urban natural regeneration does not always come from canopy individuals (Robinson and Handel 2000; Sullivan et al. 2009; Oldfield et al. 2013), not even a native canopy can promise a native understory or future canopy.

Natural regeneration in urban woodlands is highly variable: it can contain non-native species, unknown outside of urban forests (McDonnell et al. 1997; Le Roux et al. 2014), or be biologically indistinguishable from non-urban forest regeneration (Burley et al. 2008). In one study, the nativeness of spontaneous urban regeneration varied from 20 to 67% (Lundholm and Richardson 2010). As the nativeness of the regeneration increases, so does the present and potential site naturalness.

4.3.3.2 Native non-tree flora species

i. Proportion of ground cover occupied by native species

Researchers have advocated for the use of ground flora as an indicator of ecological integrity since ground flora are easily measured and responsive to a wide variety of conditions (LaPaix et al. 2009). The composition and structure of the forest understory is one of the areas most impacted by human management and urbanization (McDonnell et al. 1997; Kowarik 2005). Urban forests have highly disturbed and altered understories, from high rates of human and domestic pet visitation and from the direct alteration of vegetation; this promotes the establishment of early successional and often non-native species under mature tree canopies (Kowarik 2005). Native plant dominance and species richness are significantly lower in urban ground cover, both from the loss of native species and the gain of alien species in disturbed, open understories (Drayton and Primack 1996; Moffatt et al. 2004; Le Roux et al. 2014). Many alien species are adapted to anthropogenic disturbances and the associated soil and light conditions (McKinney 2002).

The dominance of non-native species in the understory has important implications for other individuals, since the success of a plant is influenced by the identity of its neighbours. Plants are affected by interspecific competition, either through the direct monopolization of space, or indirectly through altered microclimates, which would differ among different sets of species (Miles 1979). If non-native species are present, communities differ both compositionally and functionally.

ii. Stage-adjusted composition of epiphytic lichen functional guilds

Lichens have been used as indicators of non-urban and urban forest naturalness and urban environmental quality (Liira and Sepp 2007; Winter et al. 2010; Llop et al. 2012). Lichen richness and abundance are lower in human-influenced forests and in urban areas affected by air pollution (McCune 2000; Ranta 2001; LaPaix and Freedman 2010). The lower species richness in urban areas consists of pollution-tolerant species such as *Physica millegrana* (McCune 2000). Since different species and communities colonize and grow on different substrata, small changes in environmental conditions manifest in divergent lichen communities (Seaward 2008) (see 3.6.1.i). Cristofolini et al. (2008) found that lichen species composition shifted away from pollution-tolerant species to more sensitive species within a few hundred metres of a cement factory.

Determinations of lichen community naturalness would need to take into account the age of the stand. More-natural old stands have more lichens and a greater diversity of stress-intolerant lichens, as there has been time for colonization and a variety of microhabitats to establish. More-natural young forests may lack lichens or particular functional groups due to the slow growth and microhabitat specificity of lichens, but are not lower in naturalness.

4.3.3.3 Native fauna

i. Proportion of expected native non-avian vertebrate species absent

Different faunal communities are associated with different forest types. Those species present in non-urban forests but absent from urban forests of a similar type should be considered indicators of reduced naturalness. Native fauna are often absent from urban areas, potentially from habitat loss or alteration (McKinney 2008), competition from non-native species (Shochat et al. 2010), or novel sources of predation (e.g., domestic cats) (Mitchell and Beck 1992). Urbanization is associated with a decreased non-avian vertebrate richness (McKinney 2008), and reptile richness is lower in unforested urban sites than in reforested and forested urban reference sites (Ruiz-Jaén and Aide 2006). Since urbanization, Philadelphia has lost over one third of its recorded amphibian species and almost half its recorded reptile species (Grant et al. 2011).

Small-body mammals are also affected by certain components of urbanization: species richness declines with proximity to residential areas and with increasing coverage by paths and barren ground (Dickman 1987). Small-body mammal species may therefore be absent from more-developed areas. Many large-body mammal species are ubiquitously absent from developed areas (Dickman 1987). The size of the site in question should influence our expectations for the faunal community. Overall, since urbanization poses a hindrance or deterrent to many native faunal species, a more-natural forest would have a more-complete faunal community.

ii. Proportion of expected native avian species absent

Native avian richness also declines along a gradient of urbanization (Chace and Walsh 2006). Though the abundance of birds may increase, overall avian diversity decreases with urbanization as fewer, often alien species become dominant (Melles et al. 2003; Crooks et al. 2004; Lizée et al. 2011). As the community members shift, so do the community characteristics: larger, aggressive birds are more abundant and higher in species richness in suburban areas, whereas small birds are associated with natural forests (Parsons et al. 2003; Catterall et al. 2010); seed-eaters, omnivores, and ground gleaners are common in urban areas, while insectivores are more prevalent in non-urban forests (Beissinger and Osborne 1982; Parsons et al. 2003; Chace and Walsh 2006; Catterall et al. 2010; Lizée et al. 2011).

The decline in insectivorous birds accompanies a decline in their food base: native insect abundance and species richness decline with urbanization due to a loss of the native plant food base (Raupp et al. 2010). Urban vegetation is simplified, fragmented, and more open than many forest types; unsurprisingly, many forest interior and forest-dependent species are absent from urban areas (Chace and Walsh 2006). These altered environmental conditions favour communities of birds termed urban exploiters or adapters, able to thrive in human-dominated landscapes (Crooks et al. 2004).

iii. Proportion of expected native invertebrate species absent

Invertebrate communities are a fundamental component of naturalness. Just as in vertebrate communities, invertebrate species richness decreases with urbanization (McKinney 2008). The shift in urban vegetation from native to alien species results in depauperate and less abundant insect communities (Tallamy 2004; Raupp et al. 2010). Phytophagous arthropods decline in abundance and diversity with urbanization (Raupp et al. 2010), as do those feeding on or using lichens as habitat, as lichens are sensitive to environmental and atmospheric pollution (McIntyre 2000). Most plant-eating insects are highly specialized and can feed only on plants with which they share a co-evolutionary history (Blair and Launer 1997; Tallamy 2004). In the absence of this food base, only insects able to switch hosts or with low host specificity would be successful in urban areas; generalist invertebrate species are common in urban forests (Ruiz-Jaén and Aide 2006). The shifts in invertebrate communities affect higher trophic levels by reducing or altering their food base, and can affect plant communities where ant dispersal is a common short-distance dispersal strategy (Drayton and Primack 1996). When native invertebrates are absent, the naturalness of many interrelated dimensions is reduced.

iv. Native proportion of non-avian vertebrate species

In general, urban vertebrate populations in different regions of the world are more similar than they are different (Adams and Lindsey 2011). Native species are extirpated by habitat loss and alteration, competition, and/or predation, and are replaced by globally distributed synanthropic non-native species (e.g., *Mus domesticus*) (McKinney 2006). Alien plant species may be unpalatable or may structurally differ and thus not serve as visual attractants, deterring the presence or success of native herbivores (Beauchamp et al. 2013). Less-natural stands, due to alien plant species or the impacts of urbanization, would support a less-native faunal community.

v. Native proportion of avian species

Urbanization has a homogenizing effect on avian communities, resulting in a subset of introduced species occupying cities across ecoregions and the globe (McKinney 2006; Fontana et al. 2011). Native avian species richness and diversity decline as urbanization increases, and native species are generally replaced with fewer, alien species that numerically dominate avian communities (Beissinger and Osborne 1982; Melles et al. 2003; Crooks et al. 2004; Chace and Walsh 2006; Fontana et al. 2011). In instances when avian community richness remains similar between forested and suburban sites, non-native species enter or are more prevalent in the suburban mix (Parsons et al. 2003; Catterall et al. 2010).

Native birds have co-evolved with native vegetation and the biotic communities these support, and native avian richness is strongly related to the volume and structure of native vegetation (Mills et al. 1989; Sandström et al. 2006). Native plants may be attractants, may support more species-rich or abundant insect communities, or may provide a specific type of habitat. When non-native plants dominate the vegetation, these services are absent or reduced, and native birds must adapt or leave. While the success of native birds may be being hindered, opportunities may be present for birds, often alien species, able to exploit or adapt to urban conditions.

vi. Native proportion of invertebrate species

Arthropod responses to urbanization vary among taxonomic groups – some increase in abundance and diversity whereas others decline or are unaffected (Hochuli et al. 2009). In a four-year naturalization experiment, an "increasingly wide variety" of insects was captured within the naturalizing plots over time, suggesting that an increase in the diversity and cover of native plants led to more-diverse insect communities (Sullivan et al. 2009).

While it is difficult to generalize arthropod responses to urbanization, some urban arthropod communities may be more diverse due to the added presence of synanthropic alien species (McIntyre 2000). Many native insect species decline in abundance or are extirpated with urbanization, due to habitat loss or alterations, changes in resource availability, or predation (e.g., Blair and Launer 1997). Concurrently, alien insects are introduced with alien plant hosts or arrive independently and replace the declining native populations (Faeth et al. 2011). A globally distributed and homogenous community of urban-adapted species arises, with members such as the common fruit fly (*Drosophila melanogaster* Meigen) and roaches (McIntyre and Rango 2009). The presence of alien invertebrate species reduces naturalness.

4.3.3.4 Invasive alien species (IAS)

i. Proportion of tree cover associated with IAS

Invasive species are one of the top threats to native biodiversity (Wilcove et al. 1998; Mooney and Cleland 2001; Venter et al. 2006). In 2004, at least 27% of vascular plant species in Canada were alien invasive species (Environment Canada 2004). Invasive species form a distinct sub-category of non-native species due to their potential to outcompete and prevent native species from establishing, and their potential to quickly spread beyond their current distributions (thus reducing the naturalness of surrounding areas) (Mooney and Cleland 2001; Environment Canada 2004).

Invasive species share many characteristics with species termed “urban adapters”: high reproductive rates, tolerance of a wide range of environmental conditions, and potentially few predators or competitors (Adams and Lindsey 2011). Some invasive tree species, such as Norway maple (*Acer platanoides* L.), have been intentionally planted in urban areas due to the very traits that make them invasive, such as their rapid growth and tolerance of a broad range of environmental conditions (Nowak and Rowntree 1990). It should be noted that invasiveness can only be assigned to those species that have already exhibited invasive habits – with time, other non-native species may prove to be as dominant a restructuring force.

ii. Proportion of ground flora cover associated with IAS

Invasive plants can modify environmental or soil conditions such that the establishment or growth of native species is inhibited, and the establishment of other invasives is facilitated (McKinney 2006; Jordan et al. 2008). Some, like scotch broom (*Cytisus scoparius* (L.) Link) and purple loosestrife (*Lythrum salicaria* L.), may have been planted as ornamentals. Other species, such as dandelion (*Taraxacum officinale* F.H. Wigg), are considered weeds by many urban residents. When ground flora are used as a measure of urban ecological integrity, invasive alien plant species function as indicators of high levels of urbanization (Moffatt et al. 2004).

iii. Presence of invertebrates associated with IAS

Invasive alien invertebrate species can have impacts that extend beyond outcompeting other invertebrates: the defoliation of eastern deciduous forests by gypsy moths (*Lymantria dispar* L.), and the high mortality rates of ash (*Fraxinus* spp.) trees from the emerald ash borer (*Agrilus planipennis* Fairmaire), demonstrate the havoc wrought by invasive insects on other trophic levels (Sharov et al. 2002; Poland and McCullough 2006). The establishment of European species in North America is facilitated by the close taxonomic relationships between North American and European species, and the abundant and fairly even distributions of host species (Niemelä and Mattson 1996). European alien species generally colonize the same host genus they use in Europe, however, North American hosts lack defenses due to the absence of a shared evolutionary history (Niemelä and Mattson 1996; Brockerhoff et al. 2006). The lack of defense contributes to the success of the insects from the outset. As control and eradication have low success rates for invasive species, due to the very traits that can make a species invasive (e.g., particularly high propagule pressure resulting from a high reproductive rate), any invasive insects are signs of both current and potential depressions in naturalness.

4.3.3.5 Concordance of tree canopy composition with regional FEC classification

i. Proportion of the tree canopy that is listed as possible canopy components associated with the soil/site type based on the prevailing regional forest ecosystem classification (FEC)

While the presence of native species makes a site more natural, these species may not be in natural assemblages. A sign of urbanization is native species in unnatural habitats or groupings, such as early successional species under a mature forest canopy (Moffatt et al. 2004). In unmanaged forests, species may be present only in certain combinations or at certain points in succession, due to common responses to environmental conditions and/or direct interactions between co-occurring species (Botkin 1990). Classifying urban forest stands according to their soil/site type would indicate whether the canopy composition and associated understory could ever arise without human intervention.

The utility of this dimension is constrained by the degree to which a natural soil profile even exists at a site (see sections 3.2.2.7 and 3.2.2.9 for soil horizons and chemistry). A further complication arises through considering whether communities are static and classifiable. The current distribution of a tree species may not reflect its potential distribution, as species may be limited by competition and not environmental factors. In the absence of that competitor, in managed or unmanaged forests, a species may be successful in a novel environment. If that competitor is absent due to urban stressors, it is philosophically difficult to say whether the presence of a formerly inhibited species is a naturally occurring component of that ecosystem.

4.3.4 Structural Dimensions

4.3.4.1 Overstory density

i. Density of canopy trees adjusted to time since stand-replacing disturbance

Stand density is altered in managed stands. The woody vegetation of urban stands is often less dense than in unmanaged forests (Turner et al. 2005). Urban trees may be spaced tens of metres apart. In any forest, stand density is determined relative to the time since stand-replacing disturbance. Assessing stand density in the urban forest requires a comparison with the natural mean density for the forest type, derived from regional forest classifications and research. Difficulties may arise if the mean has a large standard deviation, as determining what is ‘near’ or ‘far’ from the natural mean would be highly subjective.

4.3.4.2 Canopy cover

i. Proportion of ground surface covered by tree canopy

In many temperate natural forests, apart from immediately after a stand-replacing or gap-forming disturbance, an almost continuous canopy should be in place over all suitable growing area; very little direct light reaches the forest floor (Spurr and Barnes 1980). Managed urban forests are more horizontally heterogeneous: urban greenspaces often have fragmented tree canopies with intervening areas of grass and built infrastructure (Beissinger and Osborne 1982). Impervious land cover reduces both canopy cover and potential growing space, and exists only in areas affected and frequented by humans. Therefore, the more natural an urban area, the more forest cover it has, excluding natural gaps that may occur over unsuitable growing space, such as large expanses of barren rock or wide watercourses. There are exceptions: the oak savannahs of eastern Ontario do not have a continuous canopy, for instance. The amount of forest cover determines, unsurprisingly, the state of many other components of forest communities: the greater the tree coverage of an area, the greater the avian richness and diversity (Mills et al. 1989; Alberti 2005; Fontana et al. 2011).

4.3.4.3 Canopy layers

i. Stage-adjusted number of distinct canopy layers

The number of canopy layers is determined by the forest type and time since stand-replacing disturbance. Managed stands in the urban forest typically lack a tall shrub mid-story and have simplified stand structures and reduced canopy layer diversity (Ruiz-Jaén and Aide 2006; Sandström et al. 2006; McKinney 2008; Roberge et al. 2008; Le Roux et al. 2014). Those urban sites with a shrub layer have significantly fewer native shrub species and fewer dominant native shrub species (Moffatt et al. 2004). This is often from safety or aesthetic and not ecological reasons. While measuring canopy layers is common in forest stand assessments, the distinctions between canopy layers can be nebulous and/or arbitrary and there is little agreement on measurement (McElhinny et al. 2005).

4.3.4.4 Tree size

i. Shape of the diameter distribution

Diameter distributions can be assessed at the species level, by when a species typically appears in succession. Early successional species without conspecific replacement have unimodal distributions; late successional species have reverse-J distributions (increasingly fewer individuals with age); species that establish after stochastic disturbances may have random distributions (Luken 1990). If trees that typically exhibit distributions with high levels of regeneration have different distributions (e.g., unimodal), this suggests that the natural patterns of these species are constrained (i.e. they are not regenerating on site; propagules from elsewhere are not arriving and establishing).

At the site level, diameter distribution can be determined by time since stand-replacing disturbance. Reverse-J age-class distributions are common in old, unmanaged stands (Uotila et al. 2002; Gibbons et al. 2008; McRoberts et al. 2012), and have also been documented in urban forests (Nowak 1993). We might expect a unimodal diameter distribution in the urban forest resulting from a singular planting event of even-aged seedlings or saplings, and subsequent hindrances to regeneration (e.g., mowing). We might also expect reverse-J distributions in even-aged mixed forests, composed of species of a variety of growth rates and diameter sizes, if the smaller diameter species are numerically dominant.

ii. Stage-adjusted presence of trees considered large/old for the species

In any forest, old trees are unique for their irreplaceable array of services: a variety of microhabitats, such as cavities for hole-nesting birds, and their influence on microclimate and water availability (McElhinny et al. 2005; Roberge et al. 2008; Winter and Möller 2008; Lindenmayer et al. 2012). The number of large trees has frequently been used in characterizing forests (McElhinny et al. 2005). In the absence of old trees, their ecological functions are reduced or absent.

Whether or not old trees are expected depends on the time since stand-replacing disturbance, and whether the forest typically undergoes stand-replacing disturbances. Areas with smaller, gap-forming disturbances have a mosaic of tree sizes and ages; forests with stand-replacing disturbances should occasionally have few or no old/large trees. Large trees, however, may be absent from urban areas. In fragmented, open habitats, such as urban parks, large trees are at a higher risk of injury or mortality from exposure to environmental factors such as wind – their removal may be citizen-driven, due to the higher risk of damage, or their absence a product this higher vulnerability. In other regions, deforestation for development may have occurred recently and the canopy has not had time to mature.

<u>iii. Inhibition of tree form by cultural practices</u>

Trees in cities are currently pruned and shaped for aesthetic and practical reasons. If tree branches begin to interfere with aboveground power lines, telephone wires, or built infrastructure, their branches are pruned. There is also a long history of human modification of the shape and size of trees; in Europe, tree forms were altered through pollarding. Pollarding originally provided fodder for livestock or wood, and carries on today due to its historic importance (Petit and Watkins 2003).

4.3.4.5 Tree location

<u>i. Degree of randomness of tree location</u>

Urban trees planted by humans generally occur in orderly patterns for reasons both practical and aesthetic. Studies of natural and anthropogenically-influenced landscapes using fractal geometry reveal that managed landscapes have much simpler patterns (Turner 1989). Unmanaged stands have randomly distributed individuals, though it is possible to find orderly placements of species based on their establishment or dispersal mechanism (e.g., seedlings growing in a row on a nurse log).

4.3.4.6 Coarse deadwood

<u>i. Stage-adjusted volume of standing and downed deadwood</u>

Deadwood, standing and downed, is typically absent from urban forests for aesthetic and safety reasons (Sandström et al. 2006; McKinney 2008; Le Roux et al. 2014). It is also removed from some managed non-urban stands; natural and near-natural stands can have 10-20 times the volume of deadwood of a managed stand (Winter et al. 2005). In theory, an increased amount of tree damage and thus deadwood might be expected due to higher exposure to wind in urban forests, where there is a greater amount of exposed edge habitat (Fraver 1994). Yet the number of dead trees, mature trees with hollows, and logs are all significantly lower in urban areas than nearby nature reserves (Le Roux et al. 2014).

Deadwood plays as important and irreplaceable a set of roles in forests as trees (McElhinny et al. 2005). It provides many unique microhabitats based on size and decay stage and species; its absence thus reduces the potential for naturally occurring decomposer communities and habitat for cavity nesters and woodpeckers (Winter et al. 2005; Sandström et al. 2006; Roberge et al. 2008; Newbound et al. 2010). The removal of deadwood alters nutrient cycling and water relations within the site (e.g., the slow release of nutrients back into the soil; downed deadwood can retain water during dry periods).

4.3.4.7 Soil composition

<u>i. Presence of intact A and B soil horizons and humus forms</u>

Urban soils often form their own distinct taxonomic class of soils (Pavao-Zuckerman 2008). They may consist of a mixture of soil and rocks brought in from other locations, and surface soil may be missing its original substrate and drainage properties (Pickett et al. 2001). The altered urban soils may overlay the natural soil profile, or may overlay unnatural materials such as rocks and debris (Sullivan et al. 2009). A and B horizons may thus be buried, mixed, or absent. As a whole, urban soils are more compacted and have shallower horizons. This affects water infiltration rates and the potential for root growth (and thus for trees to reach maturity) (Pavao-Zuckerman 2008; Millward et al. 2011). The absence of and altered soil horizons may constrict the potential naturalness of stands.

<u>ii. Soil seed bank composition</u>

The soil seed bank indicates a component of the future naturalness of a site. Species present as vegetation may not have seeds in the soil seed bank (Moffatt and McLachlan 2003; Beauchamp et al. 2013). Native seeds may have depleted their reserves from outwaiting the arrival of favourable conditions, or may not have been present in the seed bank at all, while non-native seeds can arrive from the surrounding large community of urban non-native species (Luken 1990; Beauchamp et al. 2013). Alien annual species are associated with urbanization and dominate the seed bank (e.g., at a 2.3x greater density); urban seed banks have both a significantly lower species richness and greater alien diversity than non-urban forests (Moffatt and McLachlan 2003). Regionally, seeds of some alien species are found only in urban seed banks, while seeds of many common native species are found only in non-urban seed banks (Moffatt and McLachlan 2003).

In addition to the altered inputs from vegetation to the soil seed bank, urban soils may contain a set of seeds from elsewhere. Urban soils may have been deposited on site and have brought along their own set of propagules and dormant seeds (Rebele 1992). The soil seed bank is of particular importance in fragmented areas to which certain means of dispersal (e.g., ant; explosive dehiscence) may be limited. In these areas, understanding the soil seed bank is of key importance to management decisions since it signifies whether native regeneration could be expected to emerge from the seed bank.

iii. Soil chemical composition

Urban soils are considered to be of lower quality than rural soils. They have a higher C:N ratio, and larger pools of recalcitrant carbon (i.e. carbon that decomposes slowly and is unavailable to microorganisms) (Pickett et al. 2001). Urban soils have enriched nutrient levels, high concentrations of heavy metals, and a higher pH (McDonnell et al. 1997; Alberti 2005; Newbound et al. 2010; Burton and MacDonald 2011). In a survey of British flora along urban-rural gradients, species known to associate with alkaline soils were positively linked with urbanization (Thompson and McCarthy 2008). Soil chemistry differs due to external nutrient inputs, altered litter production and decomposition rates, pollutant deposition and absorption, and the impact of non-native species on nutrient cycling (Pickett et al. 2001). The differing nutrient levels have been used as indicators of soil naturalness: low available phosphorous is an indicator of low anthropogenic influence in urban areas (Sullivan et al. 2009).

Soil chemistry and composition negatively affect soil microarthropod and microbial activity and communities. Their activity may be reduced and organisms may be absent entirely (McDonnell et al. 1997; Pickett et al. 2001). Larger organisms dependent on the soil for resources are also affected. High concentrations of some elements can enable the presence of certain functional groups (e.g., heavy metal-tolerant species) or the absence of others (e.g., fungal mutualisms are reduced in soils with high nutrient concentrations) (Lundholm and Richardson 2010; Newbound et al. 2010).

4.3.5 Functional Dimensions

4.3.5.1 Anthropogenic intrusion

i. Perceived naturalness

Since naturalization is a process occurring in a socio-ecological system, perceived naturalness is also a dimension of urban forest stands. Objects intentionally placed, such as flag tape, signs, or railings, can detract from the perceived naturalness of the site. Objects unintentionally left behind (i.e. garbage) also detract from perceived naturalness and can have negative ecological impacts if sufficiently abundant or potent. Paradoxically, the presence of litter reduces perceived naturalness while also signifying the absence of human intervention or management in that space (Peckham et al. 2013). These indicators of human access are also indicators of the ecological characteristics of a space: alien plant species are positively associated with more-frequented and -disturbed areas, whereas native plants are negatively associated (Moffatt and McLachlan 2004).

An index of perceived naturalness would have arbitrary categories and delimitations. Thus, the index and scoring is up to framework users to delimit in a way they find informative. Indices could consider the frequency at which garbage or signage is encountered during a walk-about, for example, or the permanence of each object.

ii. Proportion of site occupied by built infrastructure and paths

Built infrastructure is an indicator of the level of human access to and investment in a space. Built infrastructure and paths occupy space that would otherwise be occupied by trees in natural forests, fragments stands, and reduces native flora and fauna diversity (Alberti 2005). Paths create an edge effect that would otherwise be absent, as different plant communities and soil characteristics occur near paths and up to several metres away (Godefried and Koedam 2004). Pioneer and non-native species are common along the edges of paths under mature forest canopies that are naturally associated with later successional understory species (Godefried and Koedam 2004; Kowarik 2005). The numbers of naturally regenerating saplings and of mammalian species are negatively related to the amount of path area in a forest (Dickman 1987; Lehvävirta and Rita 2002).

Conversely, urban residents see the absence of well-maintained paths as a sign of a lack of care and responsibility for an area and can devalue these areas (Peckham et al. 2013). Without built infrastructure, it may be too inconvenient or difficult to access some sites, reducing the ability of many urban residents to enjoy the forest. Similarly, infrastructure could be designed to harmonize with site conditions, and contribute to more-positive experiences. Depending on the balance of ecological and social goals for the site, maintaining built infrastructure may be an important component of a naturalization project.

iii. Proportion of site occupied by manicured grass

While manicured grass (or lawn) is a component of the vegetation cover of an area and more-natural than built infrastructure, it is still maintained at the expense of native ground flora. Lawns differ significantly from forests: they have lower surface cover and soil organic carbon than forests, and have higher surface temperatures and light levels (Lundholm and Richardson 2010). The continual mowing required for a lawn eliminates natural regeneration. Natural regeneration may be further hindered since the lawn microclimate may inhibit seed germination, or the area may have a depleted seed bank (Hallett 2007). Maintaining lawn at the expense of native ground flora also negatively affects faunal communities. Host plants for larvae are absent, ground-nesting species lack shelter, and the food resources and microenvironments provided by ground flora are absent (Blair and Launer 1997). Without question, where grass is abundant, one of the first steps in naturalization is to cease mowing.

4.3.5.2 Adjacency & connectedness

i. Isolation within the urban matrix

Urban forests are made up of fragments isolated in a matrix of built, relatively inhospitable land cover, and are often most closely connected to degraded habitat patches. The flipside of isolation is connectivity. Connectivity is often assessed as a physical value (i.e. the presence or amount of vegetation between forest patches) though this does not represent functional connectivity per se (i.e. whether the presence of vegetative corridors is enabling the movement of some animals between otherwise isolated patches) (Noss 2004; Beier and Noss 2008). Connectivity is also taxon-specific. For example, the total amount of urban forest cover has a greater influence on bird diversity than the spatial arrangement of the forest patches (Alberti 2005). For plant species, isolation is a measure of the hostility of the surroundings for disperser movement or for propagule establishment in steps. More-isolated patches may have lower plant richness because it is more difficult for colonizers to reach the site (Crowe 1979; Moffatt et al. 2004).

When surrounded by land uses other than forest, urban forest patches have a higher likelihood of being disturbed by humans. Urban residents favour and frequent green spaces close to home (Kowarik 2005; Tyrväinen et al. 2007; Diduck 2012); urban forest patches closer to residential developments may have increased visitation by domestic cats. Visitation rates to urban parks, by humans and cats, are negatively correlated with several components of avian reproductive success (Mitchell and Beck 1992; Chace and Walsh 2006). Thus, the isolation of a site affects its potential ecological integrity.

One of the difficulties in studying connectivity is the taxon-specific scale at which isolation or connectivity is relevant. Isolation for an ant-dispersed species is different from isolation for a migratory bird. It may be that connectivity matters for the lowest common denominator: when patches are not connected for an organism with a short dispersal range, this isolation ramifies through food webs and ecological processes. It should also be noted that increasing connectivity might also increase the potential for non-desirable species, such as IAS, to move between urban areas, too. While it is certainly unnatural for a forest stand to be surrounded entirely by a built urban environment, the assessment of connectivity is nebulous.

<u>ii. Distance from non-urban sources of dispersers</u>

Connectivity has been assessed as a patch's distance from a non-urban and presumably more-natural forest (Robinson and Handel 2000), and its distance from older and larger forest patches within the urban matrix (Crowe 1979). Both are assumed to function as sources of dispersers, and patches closer to these potential sources do have a greater amount of natural regeneration (Robinson and Handel 1993; Sullivan et al. 2009). Distance, like isolation, is a species- or taxon-specific barrier. Species that disperse via ants, explosive dehiscence, or that have large, gravity-dispersed seeds are distance-limited and would have greater difficulty travelling between highly disparate fragments (Williams et al. 2009). Plants with light, wind-carried seeds would be less affected by distance, and bird community composition is unrelated to either the distance to a remnant forest or the size of an adjacent remnant forest patch (Parsons et al. 2003). However, though not explicitly assessing connectedness, Thompson and McCarthy (2008) found few and/or weak associations between a species' response to urbanization and its dispersal mechanism in a survey of British flora. While planting native species and mimicking natural habitat structure changes the composition of a site, the reestablishment of dispersal to and from surrounding communities is also a dimension of naturalness and a function of the intervening environment (Burton and MacDonald 2011).

4.3.5.3 Natural regeneration

i. Stage-adjusted density of seedlings

Natural regeneration may initially be slow in urban areas due to difficult starting conditions (e.g., a highly altered soil seed bank and compacted soil), a surrounding built rather than natural environment, hindrances to vegetation development (e.g., continual mowing), and the absence of some animal dispersers from urban areas (Robinson and Handel 1993; Millward et al. 2011). Natural forests have many more seedlings than will survive to maturity; most will die in a process of self-thinning (Schnitzler and Borlea 1998). In managed urban forests, a greater amount of resources is invested in each seedling and most or all are expected to live to maturity. Furthermore, young trees are often only present at the initiation of an urban stand; little regeneration is present after the initial plantings mature due to continual mowing. Some urban areas can have higher levels of natural regeneration, such as the spontaneous vegetation that develops in abandoned or underused areas (Lundholm and Richardson 2010). In these naturally regenerating urban stands, the density of seedlings can be highly similar to that of non-urban stands (Burley et al. 2008).

ii. Evidence of natural disturbance patterns

Natural disturbances may be undesirable in urban areas due to perceived and real danger. Fire and windthrow are common disturbance agents, but may threaten nearby lives and infrastructure. Controlled burning is used in High Park in Toronto, Ontario, to support the development of native oak savannah woodland, but involves extensive precautions to control the spread of fire (City of Toronto 2002). The dangers inherent to some natural disturbances may impede their use by humans. However, in the absence of periodic natural disturbances, early successional, disturbance-adapted species may be unable to establish in urban settings. Only shade-tolerant species would be able to establish under a canopy. Even if natural disturbances are created or allowed, they may be insufficient on their own to recreate non-urban forests, and may need to be coupled with planting native species and/or removing alien species (City of Toronto 2002).

4.3.5.4 Interiorness

i. Proportion of site considered interior

Forest edges have greater exposure to wind and light and a more variable climate than regions farther from the edge. Interior regions have more consistent conditions, with lower temperature and light and higher moisture levels. Interiorness is a function of both patch size and shape; smaller and/or elongated patches have more-pervasive edge effects as a higher proportion of the stand is near an edge. Urban forests typically have small, fragmented stands with low area-perimeter ratios (Kowarik 2005); urbanization thus reduces the amount of interior habitat (Moffatt et al. 2004). Smaller forest patches have lower bird and tree species richness overall, in part due to a reduced support for species that favour interior habitats (Alberti 2005; Chace and Walsh 2006). Similarly, the growth of some lichen species is highly reduced by edge effects (Esseen and Renhorn 1998).

Urban forest boundaries are abrupt and have higher rates of disturbance from increased human accessibility and wind velocity, and altered temperature and light regimes. Different plant communities are found at edges, with more and more-dominant alien species close to boundaries (LaPaix et al. 2012). Similarly, open-habitat and generalist insect species are found at edges (Kotze et al. 2012). Edge effects are not just limited to forest boundaries: internal fragmentation can rise from fragmentation by paths and roads. Alien species and early successional species can be found along paths and up to a few metres away in an otherwise mature forest site (Godefried and Koedam 2004; LaPaix et al. 2012).

The distance that edge effects penetrate into a forest stand is a function of the edge composition. Denser edges, with greater side-canopy, reduce the penetration of wind and radiation to a greater degree than open edges (Hamberg et al. 2009). Since the opportunities to reduce edge effects through increasing forest patch size or shape may be few in an urban context, restructuring patch edges may help foster more-interior conditions in a portion of the stand. Reducing the fragmentation by paths and roads would also create more opportunity for interior conditions.

4.3.6 Environmental Dimensions

4.3.6.1 Freedom from human-made toxins

i. Presence of indicator lichen groups

Elevated levels of toxins and pollutants are present in the atmosphere and soils of urban areas. Pollutants have a variety of origins, such as acid rain, combustion, or leachate from anthropogenic materials or fertilizers. Despite acid rain, urban soils are often more alkaline due to the "overriding influence" of high pH pollutants (Newbound et al. 2010). Heavy metals are also more concentrated in urban environments than in more-natural forests. Lichens are used in both urban and rural environmental monitoring due to their affinity for particular microclimates and sensitivity to pollutants (Seaward 2008).

Lichens are used to monitor air quality as they have many characteristics that contribute to the concentration of toxins and pollutants within their thalli. They are long-lived, indiscriminately absorb nutrients from the atmosphere, and desiccate when environmental moisture is low, thus concentrating internal substances (Nash 2008). The overall absence of lichens due to stress from toxins would be a component of both compositional and environmental naturalness, since pollutants are signs of unnatural conditions. Lichens have returned to previously depauperate environments once pollutant reduction programs were implemented (Nash 2008).

4.3.6.2 Light pollution

<u>i. Nighttime light intensity</u>

Light intensity at night is high in urban forests, and has documented impacts on birds, mammals, and insects. Light pollution affects the communication, reproduction, foraging, and other temporal activities in animals. Birds in urban areas with higher nighttime light intensities begin their morning activity sooner and extend their evening activity longer than birds in rural forests, and female birds lay their eggs earlier in the season (Miller 2006; Kempenaers et al. 2010; Dominoni et al. 2014). The speculative reasons behind the increased singing period are numerous: birds in well-lit areas may have greater access to resources, such as the invertebrates drawn to streetlamps, and male birds of higher quality are known to sing earlier in the morning (Kempenaers et al. 2010; Dominoni et al. 2014); males may incorrectly perceive the photoperiod and time their circadian activities around the altered light levels. If the latter, these changes in behaviour may have negative ramifications if singing earlier makes less-fit males 'seem' more attractive and leads to maladaptive mate choices by female birds (Kempenaers et al. 2010).

Scotobiological studies find that higher nighttime light levels also affect the activity of nocturnal mammals. Bat eye construction and physiology are better suited to dim than bright light (Fure 2006). Bats delay their ‘commuting’ when light levels are higher (Stone et al. 2009). Light may negatively affect bat navigation through changing their perception of the landscape, and bats may change their commute to avoid lit areas. Altering bat commuting timing and behaviour can negatively affect their energetics, through increasing the time spent flying and by decreasing the amount of time available to hunt. Conversely, since invertebrate prey may be drawn towards street lamps (Eisenbeis 2006), feeding efficiency may be higher.
The darkest and most-natural nighttime conditions possible for a forest would be found beyond the range of urban light pollution. Increases in light levels above this would decrease naturalness. Urban forests are unable to achieve the most-natural conditions for light levels due to their surroundings; the most-natural outcome is a minimization of light pollution, thus creating relative pockets of darkness.

4.3.6.3 Noise pollution

i. Frequency and duration of anthropogenic sounds

Noise pollution can disrupt perceived naturalness and affect the behaviour of fauna. Louder or more frequent noise affects the ability of birds to project and hear songs for mating, territory defense, warning, and communication between parents and offspring. There is evidence for both short- and long-term behavioural modifications in birds in response to anthropogenic noise (Bermúdez-Cuamatzin et al. 2011; Luther and Derryberry 2012). Birds may sing louder or at higher frequencies, or alter the timing of their singing to occur at quieter periods. In areas where daytime noise is loudest, birds may sing nocturnally, whereas singing occurs diurnally in areas with lower levels of noise (Fuller et al. 2007). Birds with lower-pitched song frequencies are more negatively affected by the ‘acoustic masking’ from low-frequency vehicular traffic and road noise (Rheindt 2003; Nemeth and Brumm 2010; Halfwerk et al. 2011).

The amount of anthropogenic noise is linked to lower clutch sizes and fledgling success rates, particularly in low-frequency singers (Francis et al. 2009; Halfwerk et al. 2011). If birds do alter their singing, these alterations could isolate urban birds from other populations in quieter areas by decoupling the timing of activity, or require an increased expenditure of energy to sing more or louder, thus negatively affecting fitness.

Urban residents identify the absence of anthropogenic noise as a feature of more-natural parks (Tyrvaïnen et al. 2007). Frequent or loud anthropogenic noise masks the sounds of the forest, and thus detracts from the perceived naturalness. Residents value the urban forest for its tranquility, birdsong, and sense of refuge from the noisy urban environment, and seek out well-forested areas for respite (Diduck 2012; Peckham et al. 2013). However, since more-formal environments, such as manicured public gardens, also can evoke similar feelings of quiet and renewal, noise pollution is a function of the size of the park and its location within the urban environment (Özgüner and Kendle 2006).

4.4 Application Opportunities & Challenges

The primary intent of our framework is, for a range of sites of naturalization opportunity, to identify unique components of naturalness so that they can be addressed individually, and thus help focus management actions. If a site is weak in particular dimensions, actions can be directed to increase the naturalness of these components. Similarly, some dimensions may be of more interest than others, such as increasing habitat suitability for a particular species or inhibiting the success of another. The fine-scale visualization provided by the framework would enable municipal and community organizations to use their time and money more effectively and efficiently in naturalization.

Determining management plans and actions will be first a matter of understanding site history and constraints. What kind of site preparation may be needed, such as bringing in soil or removing undesirable non-native species from the vicinity (Sullivan et al. 2009)? What kind of legacy in soil conditions might exist? What are the patterns and level of human use, and how will these affect expectations and potential naturalness?

Second, management plans become a matter of determining the 'appropriate' forest for the site. Through visualizing the desired and more-natural forest type, initial plantings can be determined by mimicking non-urban early successional stages. In sites that are already wooded, naturalization is a process of stand modification. Undesirable species may be present, and while their physical removal may be part of the process, naturalization is also about creating a habitat that inhibits non-native establishment. Since each site will be able to move along the continuum of a dimension only so far and only initially in steps, both long-term and initial plans for the space can be visualized. The landscape context of each site is important to understand: if the surrounding landscape does not support small mammal movements, for example, we can hardly expect them to independently arrive on site (Clewell and Aronson 2013).

Initial actions can relate to specific dimensions of naturalness, such as altering the planting pattern or determining the appropriate species composition. Later actions might relate to overall stand management, such as snag retention or woodland connectedness. Not all species will require introduction: ruderal species would be able to arrive independently. When determining the species composition, planting a few members of many species is a superior strategy to planting many members of a few species. Since site conditions in a naturalizing area will be heterogeneous, a variety of species with varying tolerances and traits may be more able to sustain processes over time (Clewell and Aronson 2013).

In a companion study to this paper, the framework was applied to sixteen urban sites (Toni and Duinker [in prep]). Sixteen dimensions were assessed on sites ranging from an untreed roadside field to an old-growth hemlock stand. We assessed dimensions pertaining to the structure and composition of the vegetation, assuming the "build it and they will come" approach to understanding naturalization potential (Palmer et al. 1997). We were also limited by our own abilities as researchers to assess certain dimensions. One site was on the western-facing slope of Fort Needham Park in the north end of the Halifax peninsula. Fort Needham is a medium-sized park, consisting of a glacial drumlin and its slopes and surrounded by residential development.

Assessments of the tree canopy within the site found that 65% of the trees were native species, but these only comprised 29% of the total basal area. The remaining 35% of trees were all invasive alien species; though fewer in number, they were the more-dominant components of the site. The canopy species had a medium level of concordance with the Nova Scotia FEC; the shrub layer, conversely, had zero concordance. This was odd as 96% of the ground flora cover consisted of native species. Thus, while these species were native, they were not the understory naturally associated with the native overstory species.

The stand scored low on several naturalness dimensions. Only 5% of the trees had a DBH over 40 cm, and none of these was considered large or old for their species. As the entire park's tree population was blown down during the Halifax Explosion in 1917, very old trees could not be expected. There were extremely low levels of downed woody debris (3.3 m^3/ha) and no standing deadwood.

Potential actions for the park include: (a) planting native species as per the FEC in the understory of the existing canopy; (b) planting more seedlings than could be expected to survive to increase the amount of standing and downed deadwood; and (c) planting ground flora species as per the FEC among the existing native species to shift the understory composition.
Through understanding how to alter planting patterns, community organizations and municipal workers can be directed towards naturalization activities with a greater chance of success. Success can be both ecological and social, as organizations may feel their actions were dedicated to a meaningful ecological goal instead of primarily for publicity. Success can be achieved through watching what was intended take hold, providing more habitat for regional flora and fauna, and increasing human interactions with more-natural spaces, resulting in greater psychological benefits and connections with nature. Clarifying our understanding, expectations, and intent for naturalization can lead to more-directed plans.

It is not expected that in applying the framework, every dimension would need to be assessed. The dimensions that are of interest for assessment and management will vary among sites and among framework users. We reiterate that the dimensions of naturalness we identified relate only to urban areas in naturally forested areas in North America, though it is philosophically and practically interesting to create a natural forest in a naturally unforested area such as prairie grasslands. Some dimensions are important conceptual components of naturalness, but unnecessary or inefficient to assess and manage. We believe that core measurements should consider (a) tree species and sizes, (b) the native proportion of regeneration, and (c) whether invasive species are present in the canopy or understory. These dimensions capture the present and potential future naturalness, as well as threats to site naturalness. All are possible to assess on-site and easily taught to non-professionals. The ease of assessment makes these dimensions of interest to managers, as citizens can be engaged in planting, monitoring, and removal programs.

Climate change may challenge many of the conditions around which our framework is built. The individual response of each species to a changing climate may decouple members in existing forest types and communities (e.g. Zhu et al. 2012). Assessing compositional naturalness through consulting regional forest ecosystem classifications, for example, would rely too heavily on historic conditions, rather than the contemporarily relevant conditions. A re-evaluation of reference ecosystems and our conceptualization of naturalness may be necessary.

4.5 Conclusion

The benefits of a naturalized urban forest are many. Since urban forests provide the first and foremost experiences many urban residents have with the non-human world, they have an important role as a teaching tool and in fostering a sense of place tied to their local environment. Humans in urban areas can be disconnected from the non-human world; naturalization can help forge this relationship, through residents participating in naturalization projects or simply walking through a landscape uncontrolled by humans. Naturalized urban forests also create more habitat for native species in a world where human developments and their associated communities are ever expanding. Our framework helps urban forest decision-makers assess the current naturalness of a site, and envision what could occur. Not all dimensions will be of interest for every stand. Furthermore, our framework is not exhaustive. There are innumerable dimensions that could be considered, and some may be yet unknown to science. We cannot presume to know entirely how to recreate a natural forest, but we can work towards facilitating its reoccurrence.

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CHAPTER 5 FRAMEWORK APPLICATION

Toni S and Duinker PN. 2015 [submitted]. Applying a naturalness framework to urban forest management: experiences in Halifax and Winnipeg. Urban Ecosys.

5.1 Introduction

The urban forest provides the first and foremost interaction many city dwellers have with nature (Nowak et al. 2001). The characteristics of the urban forest are therefore of great significance as they strongly contribute to our perceptions of and feelings towards non-human species and spaces. Urban forests typically offer a highly modified subset of regional flora and fauna (McKinney 2006). These are combined with, and often replaced by, urban-adapted non-native species (Melles et al. 2003; Chace and Walsh 2006; McKinney 2006). When living in cities, we can become disconnected from our surroundings.

Connections to the natural world are fostered in more-natural parts of the city (Peckham et al. 2013). Naturalness is the degree to which a site or ecosystem resembles a reference ecosystem minimally affected by humans (Cole 2000). There are many kinds of natural areas in cities: remnant forests relatively undisturbed by development; forests that have redeveloped post-logging; spontaneous vegetation in abandoned portions of the city; or the result of urban forest naturalization. Some spaces may require human intervention to maintain certain species or ecological processes due to stronger, detrimental anthropogenic influences; these spaces can, though, still be considered natural (Cole 2000; Higgs 2003).

Urban areas are socioecological systems, and have both ecological and perceived naturalness. Ecological naturalness is the degree to which the species, structure, and processes of an ecosystem resemble those of a natural reference ecosystem (Winter 2012). Perceived naturalness is how natural we judge a space to be, and depends on the viewer. Our perceptions are fuelled by our experiences and vice versa (Chiesura 2004), and what appears natural to one person may appear less or more natural to another (Fischer et al. 2014). Urban residents seek out spaces that provide a sense of tranquility and disconnect from the built urban environment, feelings often associated with what are deemed more-natural areas in the city (Peckham et al. 2013; Sinclair et al. 2014). The spontaneous vegetation may not offer the same experience as the remnant forest.

Naturalization is the creation of more-natural areas through altering the structure, composition, and/or processes of the urban forest so that they better resemble those of unmanaged non-urban forests. Naturalization does not set out to minimize human control over spaces; rather, it sets out to reduce the negative impacts of humans and urbanization on biodiversity. Thus, reducing some human impacts may require overriding human influence of another kind.

Urban forest management is increasingly incorporating naturalization. Urban forest master plans across Canada list naturalness as a guiding principle and suggest increasing the planting of native species and/or forest-like stands (Ordóñez and Duinker 2013). Municipalities are forming partnerships with community stewardship organizations spearheading naturalization projects (Evergreen 2001). Urban residents are individually embracing naturalization through planting native species and turning to 'ecological gardening' (Lindemann-Matthies and Marty 2013). While the focus of naturalization is to alter the biophysical characteristics of a site, many of the desired outcomes are social, such as a greater sense of place and connection with the natural world.

There are many social and ecological benefits to naturalization. Native flora and fauna may be unable to establish in less-natural urban forests due to the absence of visual attractants, suitable microclimates, mutualisms, resources, or shelter (Robinson and Handel 1993; Corbin and Holl 2002; Millward et al. 2011). An important determinant of faunal success is whether native tree species are present; native tree species support more-diverse and –numerous bird and arthropod communities (Tallamy 2004; Chace and Walsh 2006). More-natural areas are often considered more attractive, and foster feelings of tranquility, disconnection from the city through immersion in the natural world, contentment, and peace (Peckham et al. 2013; Sinclair et al. 2014). Naturalized areas also provide more opportunities for humans to interact with a variety of local flora and fauna, fostering a stronger connection with the non-human world.

There are difficulties and challenges to naturalization. Naturalized areas can appear unkempt or ugly (Ozgüner and Kendle 2006). Urban residents, particularly women, can feel uneasy around treed and shrubby areas, due to the higher perceived risk in areas with obscured sightlines (Talbot and Kaplan 1984; Koskela and Pain 2000; Kuo and Sullivan 2001; Jorgensen and Anthopoulou 2007). Naturalized areas can also contain hazards to human infrastructure and safety. There may be snags or large dead branches that can fall during extreme weather events (Lopes et al. 2009). Lastly, naturalized areas can provide habitat for undesirable species, such as predators or herbivores that graze gardens (Clucas and Marzluff 2011). It is important to balance these potential disservices when contemplating where and how to naturalize the urban forest, as some outcomes or characteristics may be undesirable.

The need for balance arises from undertaking an inherently social process in a human-dominated environment. Naturalness is a normative construct: our goals are culturally determined, and areas will only be naturalized to whatever point is feasible, desirable, or considered 'natural'. Thus, to naturalize, we need to understand what we believe makes a forest less or more natural. This involves determining 'state descriptors', or dimensions, that can situate sites and ecosystems along continuums of naturalness (Machado 2004). The proposed dimensions of naturalness in the non-urban forest (e.g. Parkes et al. 2003; McElhinny et al. 2005; McRoberts et al. 2012) and the urban forest (e.g. Williams et al. 2009) are many. Our paper describes test applications of a conceptual framework for urban forest naturalness to urban forest management (Toni and Duinker 2015).

Assuming that naturalization is a valid broad goal in an urban forest management program, we want to identify promising management directions across a range of sites that are characterized according to a suite of dimensions included in the framework. We assess sixteen dimensions from Toni and Duinker (2015) that readily assessable and contribute to the vegetation 'template' on which the other, non-flora dimensions might act. We identify an example of a suite of actions to increase the naturalness of the dimensions separately, but also discuss how the framework can be used to guide decision-making.

Urban forest naturalization can encompass planting native species and/or removing undesired species, or creating faunal and fungal habitat through retaining woody debris and snags. Due to the wide variety of potential applications of the framework, all greenspaces in the city are candidates for naturalization. Treeless lawns can be naturalized by discontinuing mowing and planting native species. Already wooded areas can be naturalized through increasing the proportion of native species and altering other site conditions. The incorporation of ecological principles and knowledge into urban planning can help create and manage spaces best suited to their urban environment (Niemelä 1999). While the assessment of naturalness is biophysically oriented, many of the motivations and hoped-for benefits are social. The end goal of naturalization will be determined by both ecological and social factors (Higgs 1997).

5.2 The Framework

Toni and Duinker (2015) created a framework to assess urban forest naturalness through consulting two bodies of literature (Table 3). The first consisted of existing naturalness assessments in non-urban forests, from which lists of dimensions of interest in non-urban settings were compiled (e.g., Parkes et al. 2003; McElhinny et al. 2005; Winter et al. 2010; McRoberts et al. 2012). These were compared with the second body of literature, studies on urban ecology. Such papers include summaries of changes along urban-rural gradients and a conceptual framework on how urban biotic communities are shaped by different biotic filters (e.g., Pickett et al. 2001; Ruiz-Jaén and Aide 2006; Williams et al. 2009). If dimensions previously used in non-urban naturalness assessments had a demonstrable link with urbanization, these were included within the framework. Some of our framework dimensions are not a part of non-urban forest assessments, but would be considered prominent components of urban ecosystems. These were determined both from existing urban ecology literature and our own reflections and deliberations, and included in the framework.

Overall, forests are assessed by their compositional, structural, functional, and environmental characteristics (Noss 1990; McElhinny et al. 2005):

- Compositional dimensions refer to what species are present and their abundances and assemblages;
- Structural dimensions refer to the physical arrangements, abundances, characteristics, and complexity of the vegetation;
- Functional dimensions refer to the occurrence and rate of ecological processes, such as the disturbance and gap dynamics of a forest stand;
- Environmental dimensions are external influences on a site resulting from its urban location.

The first three types of dimensions are interrelated, as structure can beget function, and certain species are only present when certain processes are occurring (Noss 1990; McElhinny et al. 2005; Brūmelis et al. 2011). Some traits of forests fall within multiple categories: regeneration is structural, in terms of stand density and the distribution of age classes and diameters; compositional, in terms of the species array; and a process. In the framework, environmental dimensions have an over-arching influence on other dimensions, as they affect forest composition, structure, and function, but they may not be particularly affected by the other dimensions.

Table 3 Dimensions of urban forest naturalness in Toni and Duinker (2015). Highlighted dimensions are those assessed within the present study.

Category	**Variable**	**Sub-variable**
Compositional	Native tree species	Native proportion of tree canopy individuals
		Native proportion of basal area
		Native proportion of regenerating trees
	Native non-tree flora species	Proportion of ground cover occupied by native species
		Stage-adjusted composition of epiphytic lichen functional guilds
		Proportion of the shrub layer species that are associated with the soil/site type based on the prevailing regional forest ecosystem classification (FEC)
	Native fauna	Proportion of expected native non-avian vertebrate species absent
		Proportion of expected native avian species absent
		Proportion of expected native invertebrate species absent
		Native proportion of non-avian vertebrate species
		Native proportion of avian species
		Native proportion of invertebrate species
	Invasive alien species (IAS)	Proportion of tree cover associated with IAS
		Proportion of ground flora associated with IAS
		Presence of invertebrates associated with IAS
	Concordance of tree canopy composition with regional FEC classification	Proportion of the tree canopy species that is listed as possible canopy components associated with the soil/site type based on the prevailing regional forest ecosystem classification (FEC)
Structural	Overstory density	Density of canopy trees adjusted to time since stand-replacing disturbance

	Canopy cover	Proportion of ground surface covered by tree canopy
	Canopy layers	Stage-adjusted number of distinct canopy layers
	Tree size	Shape of the diameter distribution
		Stage-adjusted presence of trees considered large/old for the species
		Inhibition of tree form by cultural practices
	Tree location	Degree of randomness of tree location
	Coarse deadwood	Stage-adjusted volume of downed deadwood
	Soil composition	Presence of intact A and B soil horizons and humus forms
		Soil seed bank composition
		Soil chemical composition
Functional	Artificiality	Perceived naturalness
		Proportion of stand occupied by built infrastructure and paths
		Proportion of stand occupied by manicured grass
	Adjacency & connectedness	Isolation within the urban matrix
		Distance from non-urban sources of dispersers
	Natural regeneration	Stage-adjusted density of seedlings
		Evidence of natural disturbance patterns
	Interiorness	Potential to support interior conditions
Environmental	Freedom from human-made toxins	Presence of indicator lichen groups
	Light pollution	Nighttime light intensity
	Noise pollution	Frequency and duration of anthropogenic sounds

Many dimensions in the framework exist along a continuum of artificial to hypothetically most-natural (Machado 2004). The framework visualizes naturalization similarly to Ruiz-Jaén and Aide (2006), where a dimension is a slider along which each site can be positioned and then moved through the process of naturalization. Other dimensions are visualized more as binary yes/no switches: whether large, old trees are present, or whether the site has the potential to support interior forest. All are framed such that the artificial or unnatural state receives a score of '0', and the most-natural state receives a score of '1'. An overview of the research supporting each dimension and the overall framework structure is presented in Toni and Duinker (2015).

5.3 Methods

5.3.1 Site Selection & Data Collection

Sixteen vegetated urban sites were selected in Halifax, Nova Scotia, and Winnipeg, Manitoba (Table 2). All sites were visited one to three times between June and October of 2014. Twelve sites in Halifax were chosen based on consultations with municipal and provincial employees working with urban and near-urban parks, and were visited in May prior to data collection to determine that the sites represented a broad range of forest conditions. The four old-growth sites were our candidates for the most-natural urban conditions in the Halifax region, whereas the least-natural condition was an untreed field alongside a major roadway. Sites in Winnipeg were chosen using the City of Winnipeg (2014) parks database. Greenspaces in Winnipeg are assigned an overall 'grade' related to a vegetation inventory; four sites were chosen to represent a mixture of vegetation types and conditions. Tree canopy and deadwood data for four Halifax old-growth sites were obtained in the summer of 2014 from a concurrent research project, and these sites were visited in the fall of 2014 to collect data on ground flora and the soundscape.

One to three 20 m x 20 m plots were positioned within each urban site. The number of plots was determined by the size of the site: some could only contain one or two plots. If the site was larger than the area of the three plots, we intentionally positioned plots to capture relatively homogeneous portions of each site, if the site was larger than three plots. We wanted similar conditions between plots as naturalization activities would not be randomly distributed across a site, but tailored to particular locations and conditions. Thus random sampling to characterize the whole was not of interest. Within each plot, all trees greater than 1.3 m in height were identified and their diameter at breast height (DBH) was measured. Identification to species followed Scoggan (1957), Roland and Smith (1969), and Farrar (1995). All snags encountered within the plot were identified to species or genus (if possible), had DBH and approximate height measured, and were assigned to a decay class.

Ten 1 m x 1 m quadrats were systematically placed within each 20 m plot (Figure 2) (Epstein 2005). All vascular ground flora within the plots were identified to species or genus and their percent cover was visually estimated. Due to the difficulty in identifying grasses throughout the field season, grasses were treated as a single group and thus were not included in assessments. Identification followed Scoggan (1957) and Roland and Smith (1969).

Four 5 m x 5 m plots were positioned at the corners of each 20 m x 20 m plot (Figure 2). All tree regeneration (any tree less than 1.3 m in height) was identified to species or genus and counted. Seedling density was determined as an average among all 5 m x 5 m plots across all 20 m x 20 m plots; averages were thus calculated among 4 to 12 plots. If the site was smaller than the area of all four 5 m x 5 m plots, all seedlings within the entire site were counted (e.g. Barrington & Cornwallis), and the total area of the site was used to calculate a single value for density. Three 18-m transects were positioned within each 20 m x 20 m plot to create a 54-m triangle (Thompson 2004). Any downed deadwood with a diameter greater than or equal to 7.0 cm that intersected a transect was included, and its diameter was measured.

5.3.2 Data Manipulation & Naturalness Scoring

Native proportion of tree canopy individuals: Each tree with a minimum diameter at breast height of 1.0 cm within the plots or site was identified to species or genus. Naturalness scores ranged from 0, if all trees were non-native species, to 1, if all trees were native species.

Native proportion of basal area: Basal area was calculated from DBH, measured on every tree at least 1.3 m tall using the formula:

$$(d^2 * \pi)/40\ 000$$

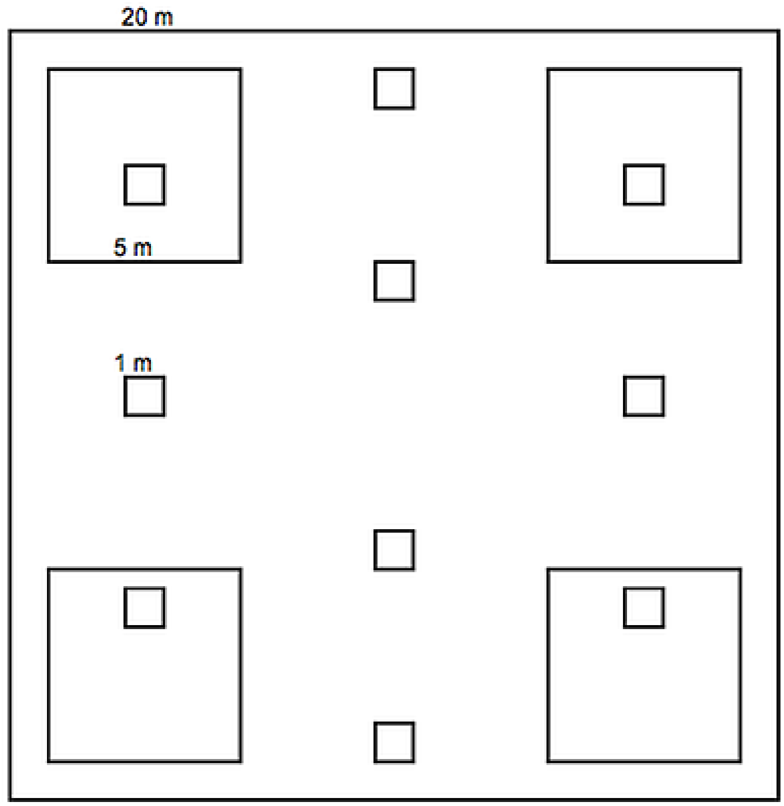

Figure 2 Lay-out of 5 m x 5 m and 1 m x by 1 m plots within the 20 m x 20 m plot

Where d = diameter at breast height (cm). Basal area was combined for all native and non-native species. Naturalness scores represent the proportion of basal area represented by native species and ranged from 0, if all basal area represented non-native species, to 1, if all basal area represented native species.

Native proportion of regenerating trees: Each seedling within the 5 m x 5 m plots was identified to species or genus. Individuals within ambiguous genera were excluded from the nativeness calculation (i.e., *Acer* sp. could be native or non-native), while individuals in non-native genera were included in the calculation. Naturalness scores ranged from 0, if all seedlings were non-native species or genera, to 1, if all seedlings were native species.

Proportion of ground cover occupied by native species: All non-grass vascular flora within the 1 m x 1 m plots were identified to species or genus. Individuals only identified to genera were excluded from the nativeness calculation if the genus included non-native species, while those in genera without any native members were included in the calculation. Naturalness scores ranged from 0, if the coverage consisted of all non-native species, to 1, if the coverage consisted of only native species.

Proportion of tree individuals associated with invasive alien species (IAS): IAS designations were determined through consulting the *Invasive Species Encyclopedia*, a comprehensive online resource on invasive species in Canada (Canadian Wildlife Federation 2015). All trees within the 20 m x 20 m plots were identified to species. Naturalness scores ranged from 0, if all trees were invasive alien species, to 1, if no trees were invasive alien species.

Proportion of ground flora associated with IAS: IAS designations were determined through consulting the *Invasive Species Encyclopedia* (Canadian Wildlife Federation 2015). Ground cover was assessed in the 1 m x 1 m plots positioned within the larger 20 m x 20 m plots. Naturalness scores ranged from 0, if all ground flora cover consisted of invasive alien species, to 1, if no ground flora cover consisted of invasive alien species.

Tree canopy FEC concordance: Each urban site was keyed out to the lowest level possible within its provincial Forest Ecosystem Classification (FEC) (Zoladeski et al. 1995; Neily et al. 2013). Several forest stands could only be classified to forest type due to the absence of native ground flora; one site could not be keyed out at all due to the absence of a tree canopy. Concordance was calculated through comparing all tree species with a diameter at breast height of at least 1.0 cm listed within the associated vegetation type and all tree species identified within the urban forest site. Any tree species present within the urban forest site but not listed in the FEC were considered ‘discordant’, and detracted from the overall concordance measure. Naturalness scores ranged from 0, if no species occurred on the FEC species list, to 1, if all species within the plots occurred on the FEC species list.

Density of canopy trees adjusted to time since stand-replacing disturbance: Density was extrapolated to the trees per hectare (ha) basis site-by-site. A minimum stand density of 1000 trees was considered natural, based on densities reported by Stewart et al. (2003) and Gower et al. (1997). All stands with a density of at least 1000 trees received a naturalness score of “1”, and the others were assigned a score representing their proportion of the minimum density (e.g. 140 trees/ha was assigned a naturalness score of 0.14).

Proportion of ground covered by tree canopy: Canopy coverage was calculated using the online tool iTree Canopy (v6. 1) (http://www.itreetools.org/canopy/index.php). Fifty random points were sampled within the boundaries of each site and were classified as either “tree” or “non-tree”. Naturalness scores ranged from 0, if there was no tree canopy coverage, to 1, if the entire plot was covered by tree canopy. Lower canopy cover could be considered fully natural in ecosystems not considered in this study.

Stage-adjusted number of distinct canopy layers: Trees were sorted into three categories by diameter: an overstory canopy (>20 cm DBH), a mid-story canopy (>10-20 cm DBH), and an understory/shrub layer (1-10 cm DBH). Naturalness scores were one of three categories: (1) *unnatural* sites, where one canopy category was absent from the plot, received a score of 0; (2) *semi-natural* sites, where all three canopy categories were present but in unexpected ratios (i.e., the understory/shrub layer is always expected to be the most abundant), received a score of 0.5; and (3) *natural* sites, where all three canopy categories were present in expected ratios, received a score of 1. We determined the category classes based on the forests in which the framework was tested; other forest types would have required different, regionally appropriate distinctions.

Shrub layer FEC concordance: Each urban forest site was keyed out to the lowest level possible within its provincial FEC (Zoladeski 1995; Neily et al. 2013). Stands that could be keyed out to a particular vegetation type within a forest type were assessed for the concordance of its shrub layer with the FEC. Shrub species were (a) those listed within the FEC and (b) those with perennial woody growth considered as shrubs elsewhere. All shrub species listed with a particular vegetation type were compared with all shrub layer species identified with the urban forest site. Any species present in the urban forest site but not listed in the FEC were considered ‘discordant’, and detracted from the overall concordance measure. Naturalness scores ranged from 0, where no shrub species present at the site were on the FEC species list, to 1, where all species present at the site were on the FEC species list.

Stage-adjusted presence of trees considered large/old for the species: Size (represented by DBH) and age at maturity for each tree species encountered in our plots were determined through consulting the US Forest Service hardwood and softwood silvics database (Burns and Honkala 1990). Any tree with a DBH over what was listed as “mature” within the silvics database was considered large/old. Professional judgment was extended to include those trees that are regionally large or old when the information from the silvics database did not necessarily support such a designation. If large/old trees were present, the site received a score of 1, whereas their absence resulted in a score of 0.

Standing and downed deadwood volume: All snags encountered within the 20 m x 20 m plots were measured and counted. All pieces of downed woody debris that intersected the 54-m transect with a minimum diameter of 7 cm were measured. The volume of all downed woody debris and snags was calculated using the formula (van Wagner 1968):

$$V = \pi^2 * \Sigma (d^2/8L)$$

Where, V = volume of wood per unit area (m^3/ha);

d = diameter of downed CWD at intersection (cm), and;

L = length of sample line (m)

The recorded deadwood volumes were compared to the average volumes as published in Townsend (2004) and the Nova Scotia Department of Natural Resources (NSDNR) website. Townsend (2004) recorded an average volume of coarse woody debris of 20.43 m^3/ha, and the NSDNR website recorded an average of 21.5 m^3/ha. Stewart et al. (2003) found that deadwood and snag volumes varied from 45 to 91 m^3/ha for deadwood and 17-57 m^3/ha for snags, but the researchers were only assessing old-growth forests. An assessment of old, ecologically healthy riparian forest in Manitoba found an average volume of 20.89 m^3/ha (Martinson et al. 2008). A minimum volume of 20 m^3/ha was used for the present study. Any volume equal to or greater than the minimum received a naturalness score of "1"; all volumes below were assigned a score representing their proportion of the minimum.

Potential to support interior conditions: A site had the potential to support interior habitat if any point within the overall site was 50 m or more from an anthropogenic edge (e.g. sidewalk; development). Moffatt et al. (2004) found that interior conditions existed at 50 m from a forest edge, and Vallett et al. (2010) used 45 m. Unpaved footpaths were not considered to break up interiorness, though edge effects have been recorded along trails (Godefried and Koedam 2004); site features under a canopy, such as a footpath, were not visible in remotely sensed data and thus could not be considered.

Frequency and duration of anthropogenic sounds: The soundscape was listened to for three intervals throughout the site visit (at the start, middle, and end). The identity (e.g. mowing; dog barking), relative persistence (i.e. how much of the two-minute period did it occupy), and predominance (i.e. how loud relative to other sounds heard at that moment) of the sounds heard at the site were noted to create an overall impression of the soundscape. Based on the pre-dominance of different sounds, sites were categorized as having "human" or "non-human" soundscapes. All sites were visited during weekday mornings or afternoons, to minimize the impact of the rush and lunch hours (i.e. when there is an assumed increase in traffic) of the nine-to-five workweek. Only the two broad categories were established to account for the low sampling rate of this subjective and dynamic phenomenon.

5.3.3 Data Analysis

Each site received a score from 0 to 1 on each of the sixteen dimensions. To determine whether scores on multiple dimensions converged on common management activities, we calculated the correlation between the scores on each pair of dimensions. Correlations greater than 0.8 were of interest to us due to their suggestion of redundancy.

5.4 Results & Discussion

5.4.1 Framework Dimensions

The ranking of the sites was fairly consistent for the majority of the dimensions that were measured (Figure 3). Higher scores were captured in old-growth and Winnipeg sites, whereas the lowest scores were almost always found in at least one of three sites (Ardmore Park, Barrington & Cornwallis, and Burnside). There were several exceptions. For instance, the proportion of ground flora consisting of invasive alien species varied from this pattern. Invasive alien species were found across sites types, from the untreed field to the old-growth sites alike. This should not be surprising, due to the pervasiveness of invasive species in urban settings and their highly dispersive capabilities. It is thus always important to determine whether invasive species are present, no matter how natural the other dimensions of naturalness.

Several dimensions received a score of 0 for naturalness at many of the sites. Only half the sites visited had any degree of shrub FEC concordance; the average FEC concordance score was 0.27. Many sites were lacking a shrub layer entirely, such as Ardmore Park; other sites had a shrub layer made up of species entirely inappropriate for the site type. Past research suggests that a shrub layer is often absent from urban forests (Moffatt et al. 2004; Sandström et al. 2006). Urban sites with a shrub layer often have many non-native species (Moffatt et al. 2004; Le Roux et al. 2014). The scores (or absence thereof) on this dimension suggest that it is of particular interest for naturalization, as a natural shrub layer may require human intervention for establishment in many locations.

One dimension stood out as inefficient to assess in as fulsome a manner as we did. The volume of dead woody debris was relatively more intensive to measure: our assessment required transect-based data collection. In the end, sites either had minimal to no deadwood, or a more-than-sufficient volume. Thus our method of measuring this dimension was perhaps unnecessarily complicated, and managers may be better off assessing the dimension as a binary presence/absence score.

Only four sites, three of which had high scores on almost every other dimension, had a soundscape dominated by non-human noises and thus received a score of 1; the remainder received a score of 0, for anthropogenic sound dominance, or 0.5, if sounds were recorded equally. While a more-dense forest might attenuate urban noise to a greater degree and produce its own repertoire of leaf rustling and

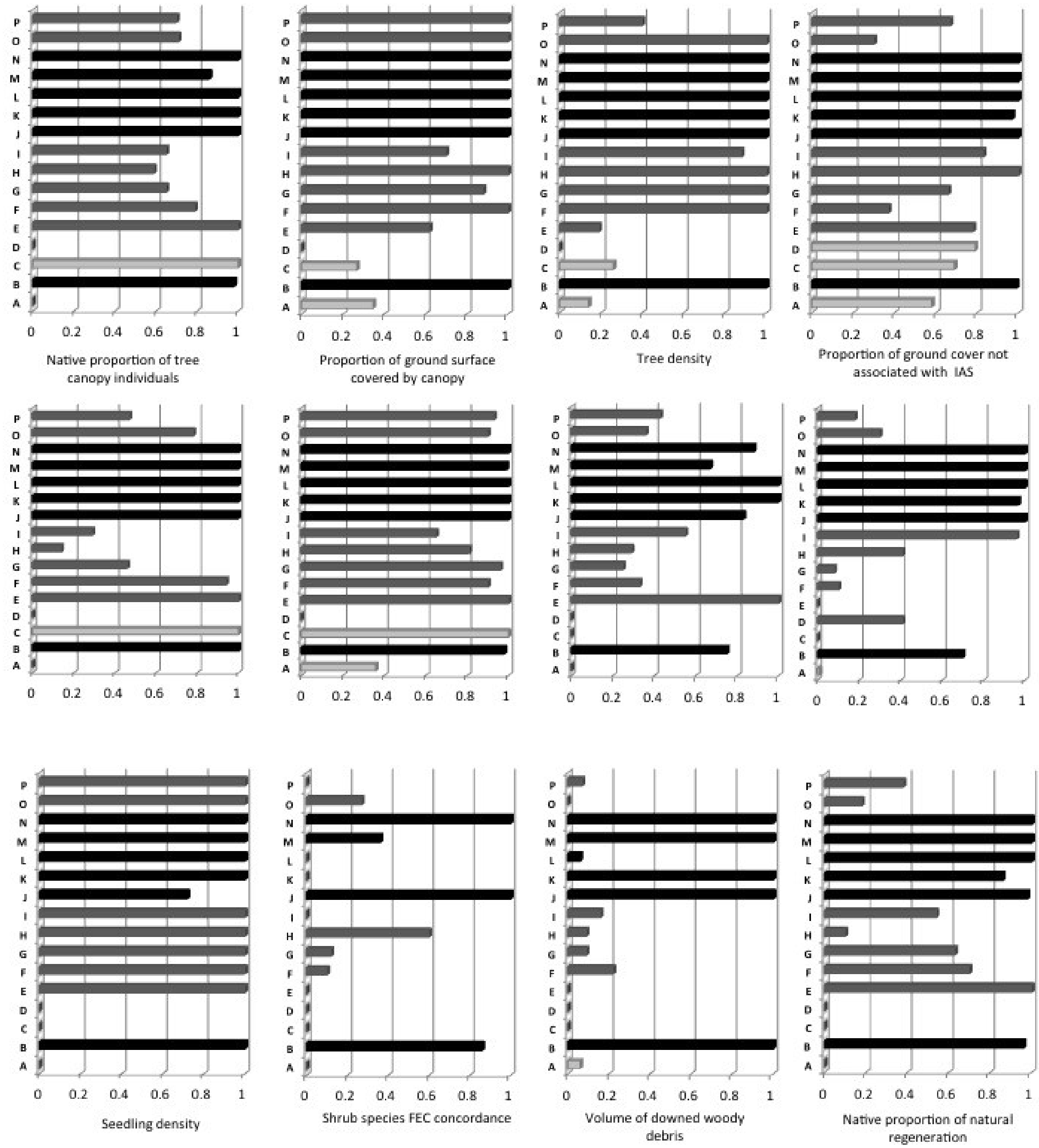

Figure 3 Naturalness scores of sixteen urban forest sites in Halifax, Nova Scotia, and Winnipeg, Manitoba: Ardmore (A); Assiniboine Forest (B); Barrington & Cornwallis (C); Burnside (D); Churchill (E); Flinn Park A (F); Flinn Park B (G); Fort Needham east (H); Fort Needham west (I); Hemlock Ravine (J); McBeth inland (K); McBeth riverside (L); Point Pleasant Park (M); Shubie Park (N); St. Mary's Boat Club forest (O); St. Mary's Boat Club waterside (P). Dimensions with categorical scores are not depicted.

birdsong, sounds such as high-speed traffic and industrial activity can be pervasive. Many components of the soundscape impinge on an urban forest site, and are not a product of the site itself. Sounds are thus one of the biggest obstacles to increasing perceived naturalness in the urban forest. Even old-growth sites visited in this study were punctuated by anthropogenic sounds. This is unfortunate, as urban residents identify the soundscape of birds and leaves rustling as one of the delights of a natural urban forest (Peckham et al. 2013). The soundscape is an important dimension to consider as it may help us set priorities for naturalization. Sites dominated by anthropogenic sounds may not be good candidates for naturalization if we are interested in creating spaces where humans can disconnect with the built city and feel immersed in the natural world.

5.4.2 Correlations Among Dimensions

There were seven relationships among dimensions with a correlation equal to or greater than 0.8; we were interested in relationships that may be highly correlated to a point of redundancy, and not simply significance (Table 4). If dimensions do not add any new information, they are inefficient to analyze. Furthermore, we might assume that in increasing the naturalness of one, the naturalness of the other automatically comes along; it is thus inefficient to treat and manage them separately.

The scores of canopy differentiation and canopy cover had a correlation of 0.9. Canopy differentiation was a categorical score, derived from the grouping of trees by diameter into three classes and assessing the proportions between classes. It is thus less informative (or more derived) than the canopy cover score, which is a continuous dimension determined from aerial photographs. Our results suggest that future naturalness assessments need not consider canopy differentiation as a separate category – we found that sites with high canopy cover have trees of a variety of sizes, and support conditions in which young seedlings are establishing. The lower canopy differentiation scores of more-open sites potentially relate to the openings in the canopy that allow ground flora such as grass to outcompete young trees. This relationship is captured in another of the highly correlated pairs of dimensions, seedling density and canopy cover.

Not all strongly correlated variables suggest redundancies that could be streamlined. The relationship between the native proportion of tree canopy individuals and the native proportion of basal area was exponential (r=0.9). While the naturalness scores may have had a high correlation, this relationship was

Table 4 Framework dimensions with correlations greater than 0.8 with associated R^2 values

	Native proportion of tree canopy	Tree canopy concordance with FEC	Tree density	Native proportion of ground flora	Seedling density	DWD volume	Canopy differentiation
Native proportion of basal area	0.91 (R^2=0.82)						
Native proportion of natural regeneration		0.9 (R^2=0.8)					
Interior potential				0.81 (R^2=0.65)		0.89 (R^2=0.79)	
Canopy cover			0.86 (R^2=0.74)		0.89 (R^2=0.79)		0.9 (R^2=0.81)

weakened when the actual basal area values were considered (r=0.41). Understanding the canopy composition thus does not lead to an understanding of the canopy structure and native dominance in the site. Basal area represents the relative dominance of native trees within the site; if the native trees are all in small diameter classes, the site's dominant structural elements (e.g. whether it acts as a visual attractant for birds) are supplied by non-native species. Larger and thus presumably older trees may also provide greater contributions to regeneration; if non-native trees dominate the reproductive component of the community, this affects future naturalness of the site.

Similarly, tree FEC concordance had a correlation of 0.9 with the native proportion of the natural regeneration. FEC concordance was calculated by determining whether any trees present in the site were not listed with the site's vegetation type, and thus sites with fewer discordant canopy species had fewer non-native up-and-coming trees. Despite the high correlation, we still consider it important to determine the native proportion of seedlings to understand the site's ability to perpetuate natural conditions, since many alien invasive species are highly dispersive and were found as seedlings in our most-natural, old-growth sites. Would we discard FEC concordance due to the high correlation between the two naturalness dimensions? It may be that the non-native species were driving down the concordance scores, and that in their absence, sites had fairly natural communities. FEC concordance is one of the weaker variables in our framework, due the high variability in natural communities and our inability to determine soil types, part of the classification, in many urban regions. However, it speaks to the assemblage of species in a way that other dimensions do not. We still think it is important to consider if species could co-occur, and to plan plantings around natural assemblages.

Lastly, the native proportion of the ground flora was strongly correlated with the site's interior potential (r=0.81). Larger sites had more-natural ground cover. However, this may be an artifact of our site selection – the same may go with the relationship between the volume of CWD and interior potential. We did not choose any large, less-natural sites, though they are certainly present in urban areas; our largest sites were also our most-natural sites. There is limited naturalization potential for most large, sparsely treed urban areas, which often have particular social functions related to being large and open (e.g. sports fields; baseball diamonds).

5.4.3 Similarities in Management Directions

Based on how sites scored on all dimensions, we find that management actions would fall into three broad groups. The first group, "Stand Initiation", consists of three sites scoring at the low end of almost every naturalness dimension. The three sites fall outside of the popular conception of a forest: one lacked trees all together, one had twelve, and one had fourteen trees. Two of the sites were small (less than 1000 m^2) and bounded by roads or development; however, no Stand Initiation site was entirely covered by trees (the highest canopy coverage value was 34% for the fourteen trees). Even when trees were present, these sites scored low on canopy compositional and structural dimensions – the trees were all non-native species, or all the same age, or provided patchy canopy cover. The dominance of non-native species in the understory was also characteristic of Stand Initiation sites.

The second group, "Stand Transformation", consists of sites with a potpourri of scores: they received middling scores on many of the naturalness dimensions, high scores on a few, and scores of 0 on others. These sites scored relatively more highly on dimensions relating to tree canopy composition and structure. Often, a native canopy with several layers and high ground coverage was present. The sites scored lower on dimensions relating to ground flora and shrub characteristics.

Lastly, the "Monitoring" sites are a group of six sites that received top scores on at least half their dimensions, and generally only received a score of 0 on one dimension, typically the soundscape dimension. Monitoring sites had been characterized as highly natural or old-growth prior to the study by both researchers and urban forest managers (LaPaix and Freedman 2010). Half of the Monitoring sites had a soundscape dominated by non-anthropogenic sounds, three of the only four to be characterized as such. It may be that Monitoring sites would not score as highly on dimensions not assessed in this study – those dimensions relating to faunal communities, for example. While we primarily characterized the vegetation, and considered it a template for the other dimensions, we did not assess whether this is indeed the case.

5.4.4 Management Implications

There may be overall categories for management activities. For example, the environmental organization Green Seattle Partnership has four phases for managing urban forests, depending on the site's naturalness along gradients of the native proportion of the tree canopy and the ground cover of invasive species: (1) invasive plant removal; (2) planting and secondary invasive removal; (3) plant establishment (native species); and (4) long-term monitoring and maintenance (Green Seattle Partnership 2006). These activities decrease in intensity and labour investment as forests become more natural and there are decreased costs associated with their management. It should be noted that Green Seattle Partnership's framework was constructed around the threat of invasive species such as ivy, and around increasing forest regeneration to sustain the urban canopy for centuries. These phases have some overlap with the three groupings of sites we saw emerge. Stand Initiation sites require invasive plant removal in the understory and subsequent planting, often of trees. Stand Transformation sites may also require the removal of invasive species, but are also sites where we are focused on the establishment and survival of existing native species. Lastly, Monitoring sites align with Green Seattle Partnership's fourth phase.

Overall management activities may be similar among sites that grouped together (Appendix). However, we believe it is better to determine naturalization goals and actions based around individual site assessments. The required actions to increase the naturalness of the ground flora and shrub layer of Stand Transformation sites will be context-specific. Fort Needham (west) has almost an entirely native suite of ground flora species; its FEC shrub discordance may relate more to the dominance of non-native species in the canopy. Its closed canopy may allow native species indicative of forest conditions to establish; it is the canopy composition that needs shifting. Sites such as St. Mary's Boat Club (forest) would require an intensive invasive removal program before introducing a native understory, whereas Fort Needham (east) is mostly bare with sparse blackberry (*Rubus* spp.) shrubs, but with nearby and encroaching Japanese knotweed (*Fallopia japonica* (Houtt.) Ronse Decr.) that would require monitoring.

To increase the native proportion of the tree canopy, in terms of **individuals** and **basal area**, and the **native proportion of the natural regeneration** we would both remove non-native trees and plant native seedlings or calliper trees. This removal in established stands should attempt to avoid disrupting the relatively shadier forest floor conditions already in place that may favour the growth of forest understory species or later successional canopy species. Selecting native species should be done with the site's **FEC vegetation type**, if available, to recreate natural assemblages.

The action to increase **tree** and **seedling density** is simply to plant more trees. Any decreases in density from removing non-natives to increase canopy naturalness should be taken into account. Similarly, increasing **canopy coverage** can be accomplished by planting more trees. Planting many young trees would hopefully increase the naturalness of the **canopy differentiation**, by moving sites towards a reverse-J diameter distribution (where young trees are the most abundant).

Increasing the **native component of the ground flora** should entail consulting the FEC for the site, if available, to determine the appropriate species composition for plantings. Locally sourced stock is preferable to maintain regional gene pools. This would also increase **FEC concordance of the shrub layer.**

Increasing the **volume of coarse woody debris** could occur through girdling trees or mechanically pushing trees over. This may be perceived as a contentious activity in parks, particularly if naturalization is an overtly stated goal. Residents may not equate removing trees with increasing naturalness. Deadwood could also be brought in from off-site, or simply left within the park when it falls.

Some dimensions will be difficult to manipulate. Dimensions such as **interior potential** and **soundscape** are products of the surrounding environment and not necessarily of the site itself. Interior potential could be increased through expanding the park's area, or the naturalized area of the park. This action may be limited in its application. Creating dense buffers of vegetation could mute some anthropogenic noises, but some low and/or loud sounds might be pervasive. Other dimensions, such as whether **large/old trees** are present, cannot be directly manipulated.

Invasive species should be removed and suppressed to the greatest degree possible. However, this may be a Sisyphean task, due to the dispersive qualities of invasive species.

Only sixteen of the thirty-eight framework dimensions from Toni and Duinker (2015) were considered within this study. We assessed dimensions relating primarily to the vegetation. This focus is not uncommon in restoration studies, which often adopt a "build it and they will come" approach (Palmer et al. 1997). We view vegetation as a template, and assume that other components of naturalness, such as native faunal communities or particular lichen guilds, will be attracted to and/or supported by areas in which native vegetation (both compositionally and structurally) has been re-established. For instance, most plant-eating insects are highly specialized and can only feed on plants with which they share a co-evolutionary history (Blair and Launer 1997; Tallamy 2004). In the absence of this food base, only insects able to switch hosts or with low host specificity would be successful in urban areas; correspondingly, generalist invertebrate species are common in urban forests (Ruiz-Jaén and Aide 2006).

Baseline measurements should consider (a) tree species and sizes, to assess present naturalness and ability to foster natural processes; (b) the native proportion of regeneration, to assess future naturalness were the site to be minimally managed; and (c) whether invasive species are present in the canopy or understory, which threaten present and future naturalness. Present naturalness is not an indicator of whether or not invasive species are present, as we found invasive species in untreed fields and old-growth sites alike. The native proportion of natural regeneration had only one strong correlation with any another dimension – we thus need to assess it independently of the other dimensions. Lastly, we restate our belief that in the absence of natural canopy composition and structure, which acts as a template, the other dimensions cannot achieve natural states either.

While not an inherent component of our framework, we believe that the framework could be used both to prioritize sites for naturalization and determine management directions. If municipalities are uncertain which sites are of interest, they could assess sites using our framework and decide which dimensions are most of interest. Managers may be interested in various qualities of the urban forest – some may have strong invasive removal programs, whereas others may want to target sites farther along naturalness dimensions to recreate habitat for certain more-sensitive species. If municipalities have already determined which sites are of naturalization interest, the framework would help pinpoint individual management goals and actions to increase naturalness.

5.5 Conclusion

If urban forest managers were interested in moving ecosystems to a more-natural state, then they would need a structure to guide decision-making. The Toni and Duinker (2015) framework provided a useful way to conceptualize and assess the urban forest. The framework structure of breaking naturalness into its constituent parts provided clear potential management directions and goals. We believe that the majority of the subset of dimensions we considered was a useful starting point for naturalness assessments. Additionally, our application of the framework found that some of the dimensions we considered might ultimately be of no interest to managers. Dimensions may be important conceptual components of naturalness, but unnecessary or inefficient to assess and manage, such as canopy differentiation.

Further applications of the framework would help clarify initial trends. While our sites fell into three groups, it may be that this was an artefact of our site selection biases. There are surely sites that fall between the groups we delineated, thus forming more of a continuum. In that case, suggestions for overall management goals would be less of interest. Similarly, we only considered a subset of the 38 dimensions; assessing other dimensions might have yielded different groupings for our sites. The untested dimensions are linked to urban-rural gradients within the literature, but their potential application to naturalization decision-making remains unclear. Some may provide good guidance but be difficult to assess, such as the proportion of expected native fauna that were absent. The absence of faunal species would still be worthwhile to determine, whether in terms of establishing reintroduction programs or taking into account the absence of their effects on other members of the community. Some dimensions not assessed by us, such as soil characteristics, would play an influential role in the site's trajectory and potential.

Naturalizing the urban forest will only increase in importance as more and more people move to urban areas, and as our activities and needs further encroach on remaining natural areas. Unless we change the ways in which we create and manage our cities, we will gradually lose both the natural world and our connection to it. Connections to the natural world are fostered in more-natural parts of the city, and these may be the first and foremost connections urban residents have (Peckham et al. 2013). If we have never come across a moccasin flower in bloom, it is already lost to us. Knowledge breeds understanding, and understanding in turn promotes care. Without these connections, how can we be aware when that flower is lost from the rest of the forest, or advocate for its persistence?

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CHAPTER 6 CONCLUSION

Urbanization has replaced the previous landscape with a novel ecosystem in which native species and processes may or may not be able to establish and occur. Many components of a typical urban site impede the spontaneous establishment of native species: the grass may inhibit seed establishment, the soil seed bank may be depleted after years of being a lawn, or the area may not necessarily be connected to potential seed sources or source populations (Hallett 2007). In the absence of human intervention, urban areas may not immediately tend towards naturalness of their own accord (Doody et al. 2010). The species composition and ecological processes of urban greenspace are heavily influenced by direct and indirect human impacts; an example is the abundance of invasive alien species that establish in unmanaged lots. High degrees of naturalness are possible alongside human development, though, as found in a third of the sites sampled in this study. My framework for urban forest naturalness is an accumulation of knowledge to date on how urban forests differ from their natural, non-urban counterparts. Through understanding the individual components of naturalness, they become easier to assess, create, and manage.

Applications of the framework found three groups of sites that either needed initiation, transformation, or monitoring. Sites in which there were no trees, or in which trees are few and non-native, require initiation or the planting of native trees. Those sites with a fairly continuous tree canopy and understory of native and non-native species require stand transformation. This occurs through removing non-native species, creating conditions in which native species can become self-sufficient, and fostering natural processes. Lastly, sites that scored highly on almost all naturalness dimensions (often old-growth stands) require monitoring to determine whether non-native species or processes are not encroaching on the stand.

Test applications only considered a subset of the framework dimensions. Some untested dimensions are important conceptual components of naturalness but may be difficult to assess. Determining the correlations or lack thereof between tested and untested dimensions would increase the knowledge available to urban forest managers. For instance, understanding which dimensions are most closely linked to the presence or absence of native fauna would provide priorities for site management if providing habitat is a goal. Some studies have found that the amount of vegetation is more important than the positioning of and linkages between greenspace for native bird richness (Sandström et al. 2006). Managers interested in providing habitat for native birds would thus focus on increasing tree density and canopy coverage rather than connectivity.

It is also worth understanding how perceived naturalness relates to ecological naturalness. Naturalization in the form of fallen branches and unmown grass can look messy, and removing expansive, well-established invasive species can leave bare swaths in formerly green parks. Attempts to increase ecological naturalness can negatively impact the attractiveness or perceived naturalness of a site (Gobster 1999; Dallimer et al. 2012; Gaston et al. 2013). Some studies have assessed whether the positive feelings urban residents associate with time spent in natural urban areas are linked to the ecological qualities of those spaces (Fuller et al. 2007; Dallimer et al. 2012). Understanding which dimensions of the framework are most closely linked to perceived naturalness might provide priorities for managers, as greater public support may be available for projects that align with elements of perceived naturalness.

In conclusion, naturalizing the urban forest will only increase in importance as more people move to urban areas, and our activities and needs further encroach on remaining natural areas. Naturalization can combat the global loss of biodiversity through providing a small increase in habitat available for native species, but, more importantly, could foster conservation-minded surban residents. If natural forests were accessible to urban residents, they could function as teaching tools, through structured and unstructured activities, and provide less-ecologically impoverished experiences. Humans in urban areas can be disconnected from the non-human world (e.g. Doody et al. 2010); naturalization can help forge a re-connection. Urban residents could participate in naturalization projects, or simply have the opportunity to walk through a landscape relatively uncontrolled by humans and grow accustomed to native species. This re-connection could translate into greater concern for natural areas, and promote a conservation ethic in urban residents.

Naturalization is further important in ever-expanding cities due to its contribution to urban liveability (Chiesura 2004; Hale et al. 2015). Trees, the most prominent component of a natural urban forest, provide numerous economic services such as stormwater reduction, pollution absorption, reduced cooling and heating costs for buildings, and an increased longevity of infrastructure such as asphalt (cf. Duinker et al. 2015; Hale et al. 2015). While some of these services could be provided by any and all trees, native trees provide social and ecological services that non-native species cannot, relating to the particular time and place of the city. Conversely, there may be some services that can only be provided by non-native species: residents might enjoy the flowers of a magnolia tree in spring. Other services can be provided by native and non-native alike. Through increasing urban forest naturalness, we increase habitat for native species and create spaces that foster human health and well-being.

Cities may be conceived as human habitat first and foremost, but we receive physical, emotional, and spiritual benefits from the non-human world. We ignore naturalness at our peril. A more-natural urban forest contributes to more liveable and sustainable cities. Many urban residents are interested in and appreciate naturalness, and favour managing urban areas as habitat for native species, even if they cannot identify these species (Doody et al. 2010). Naturalization thus serves direct human interests and the well-being of the broader public. Creating more-natural cities also breaks down the divide between humans and nature, enabling a new conceptualization wherein humans and naturalness co-exist, rather than the former compromising the latter. As cities become more integrated into their surroundings, ideally so do their human occupants. Naturalization helps intertwine the non-human world with one’s sense of home, and this feeling of being a part of the natural world is of increasing importance as wild spaces and species are disappearing. We will only protect what we love and can only love what we know. Naturalization is a small step, but an important one, towards caring.

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APPENDIX SITE DESCRIPTIONS

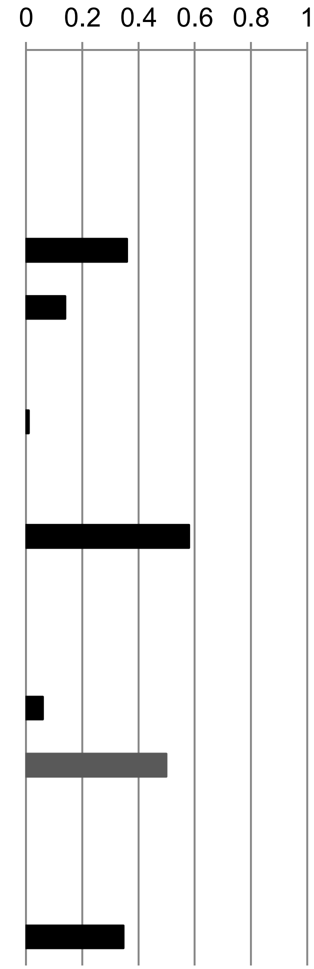
Ardmore Park

Ardmore Park is located at the intersection of Almon Street and Oxford Street in the west end of Halifax. The park contains a basketball court, a fenced-off overgrown horseshoe pit, a small manicured garden with benches and walkways, and a mowed grassy area with a scattering of non-native trees. There is continual vehicle traffic on the two streets, and the sparse canopy provides little shade. It is not an ideal place to linger, and only three people in total were seen in the park during three field visits.

Opportunities: The noisy environment means the park may have low social value for providing an escape or relief from an urban setting. There may be few social goals to balance with ecological goals. Planting native trees should be the priority for the site. This would increase the naturalness score of many of the dimensions. Ceasing mowing and planting a native understory in the fenced-off portion could be an interesting naturalization experiment.

Challenges: The small size of the park means some dimensions of both perceived and ecological naturalness will be unlikely to increase. The soundscape is dominated by anthropogenic noise, and edge effects will penetrate into the entire site. These edge effects will promote the establishment of many non-native or invasive alien species, many of which do well in exposed areas. Since the park is located at a fairly busy intersection, it may be undesirable to establish a dense understory that reduces sightlines. Increasing canopy naturalness may be the primary goal for much of the park.

Dimension	Score
Native tree proportion	0
Native basal area	0
Tree canopy FEC concordance	0
Tree cover IAS	0.36
Tree density	0.14
Large/old	0
Native ground flora	0.01
Shrub layer FEC concordance	0
Ground flora IAS	0.58
Native natural regeneration	0
Seedling density	0
CWD volume	0.06
Canopy differentiation	0.5
Soundscape	0
Potential for interior conditions	0
Canopy coverage	0.35



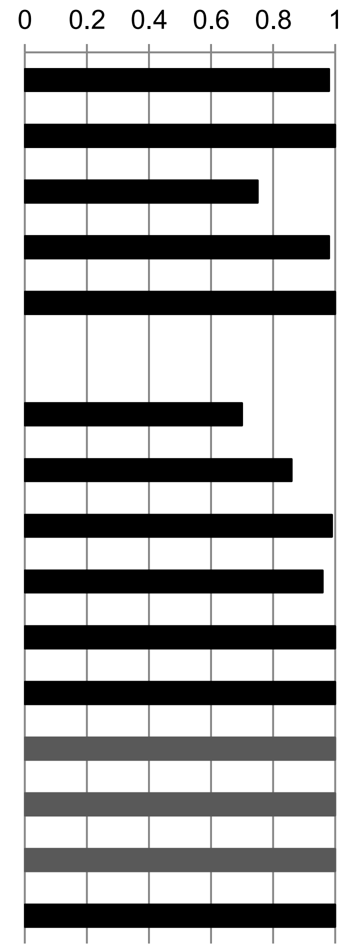
Assiniboine Forest

Assiniboine Forest is in the largest city park in Winnipeg, covering 283 hectares. The park contains a zoo, conservatory, numerous sports fields, several sculpted gardens and ponds, a children's playground, and extensive forested areas with a network of paved and unpaved trails. The sections south of Corydon Avenue are mostly forested and relatively undeveloped, and are used primarily by walkers and cyclists. Assiniboine Forest is surrounded by residential and commercial developments on three sides; its southern edge borders a high-speed road separating it from the South Wilkes rural residential area.

Opportunities: The sections south of Corydon Avenue have a high potential for both naturalization, due to the low levels of development. Current recreational uses align well with stronger ecological naturalization goals. Assiniboine Forest primarily requires monitoring to ensure that invasive species are not interfering with the establishment of native species or natural ecological processes.

Challenges: The high levels of use, in particular by dogs, may be a detriment to some naturalization goals. Pedestrians act as a dispersal source for non-native species, or dogs might harass small animals. Additionally, the area supports many white-tailed deer (*Odocoileus virginianus*), which can be a nuisance for nearby residents and hazards for drivers. Reducing the white-tailed deer presence may in fact be of interest. Similarly, increasing the number of other large faunal species may be unrealistic due to the surrounding residential developments.

Dimension	Score
Native tree proportion	0.98
Native basal area	1
Tree canopy FEC concordance	0.75
Tree cover IAS	0.98
Tree density	1
Large/old	0
Native ground flora	0.7
Shrub layer FEC concordance	0.86
Ground flora IAS	0.99
Native natural regeneration	0.96
Seedling density	1
CWD volume	1
Canopy differentiation	1
Soundscape	1
Potential for interior conditions	1
Canopy coverage	1



Source: City of Winnipeg

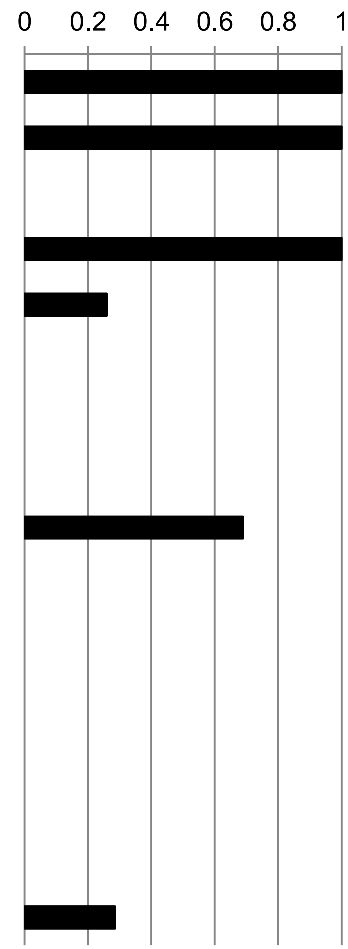
Barrington & Cornwallis

This site is a small triangular patch of grass and trees at the intersection of two major roads just north of downtown Halifax. The patch is isolated from other parks or forested areas within the urban environment, apart from the forested area overrun with invasive species running alongside Barrington to the north.

Opportunities: The canopy in place already consists solely of a native species, sugar maple (*Acer saccharum*). Through continuing to plant native tree species of a variety of sizes, many other naturalness dimensions could be increased (seedling density, tree canopy coverage, canopy differentiation). Planting a native understory may be possible, but the small size of the site means edge effects will be pervasive and enable the establishment of shade-intolerant non-native species.

Challenges: Only some dimensions of naturalness will be appropriate for this site. Its small size, distance from other natural green spaces and non-urban forests, and location within a built-up portion of the city will limit what species could occur here. A nearby roadside slope is unmanaged and heavily invaded with species such as white poplar (*Populus alba*) and Japanese knotweed (*Fallopia japonica*), making their establishment in a naturalizing site highly probable.

Dimension	Score
Native tree proportion	1
Native basal area	1
Tree canopy FEC concordance	0
Tree cover IAS	1
Tree density	0.26
Large/old	0
Native ground flora	0
Shrub layer FEC concordance	0
Ground flora IAS	0.69
Native natural regeneration	0
Seedling density	0
CWD volume	0
Canopy differentiation	0
Soundscape	0
Potential for interior conditions	0
Canopy coverage	0.29



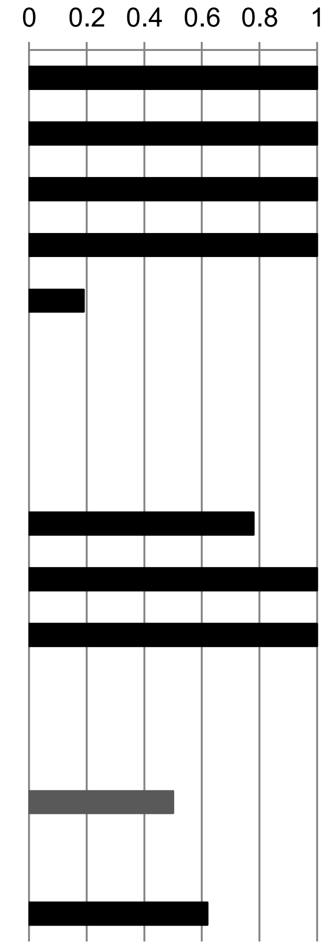
Churchill Drive

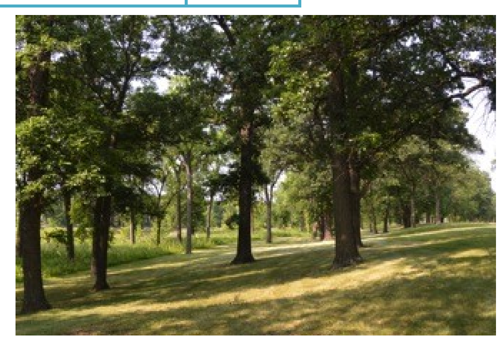
Churchill Drive is a road bordered on one side by a greenway running along the Red River that is used extensively for recreation. On the other side is a residential neighbourhood. The greenway contains a multipurpose path running through the riverbank forest; further 'monkey trails' or unauthorized bike trails run right along the riverbank. Portions of the greenway are sparsely treed or have no trees at all, and the mowed understories are used for activities such as picnicking.

Opportunities: Naturalization could include the sparse planting of more native trees to increase stand density without obstructing sightlines and the site's 'parkland' feel while increasing canopy closure and differentiation. Naturalness could be increased through removing non-native and invasive alien species from the understory while planting native species. However, a naturalized understory may be undesirable due to conflicts with the area's recreational value.

Challenges: Many framework dimensions are limited by the area's recreational use. A shrub layer, large volume of coarse woody debris, or high seedling density may be undesirable. While the site had a high score for seedling density, these were all very small seedlings that would be removed in subsequent mowing or were growing immediately adjacent to a tree trunk. Churchill Drive is much larger than the area sampled. Naturalization projects would need to be fairly uniform at a large scale or would create a disjointed, messy aesthetic for the region.

Dimension	Score
Native tree proportion	1
Native basal area	1
Tree canopy FEC concordance	1
Tree cover IAS	1
Tree density	0.19
Large/old	0
Native ground flora	0
Shrub layer FEC concordance	0
Ground flora IAS	0.78
Native natural regeneration	1
Seedling density	1
CWD volume	0
Canopy differentiation	0
Soundscape	0.5
Potential for interior conditions	0
Canopy coverage	0.62





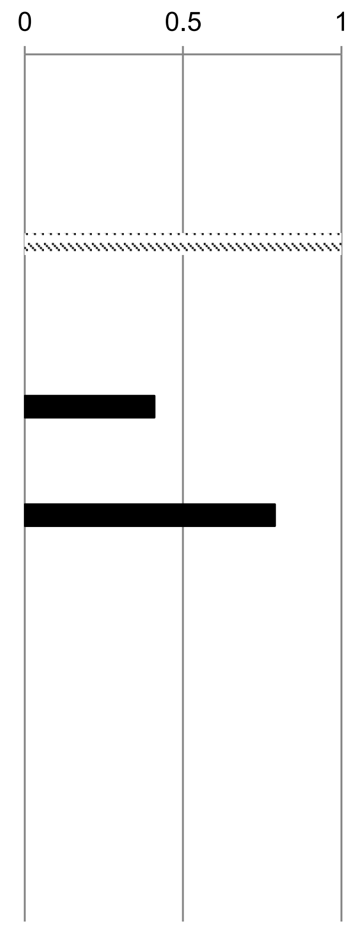
Don Bayer Sports Park (Burnside)

Don Bayer Sports Park is located in Burnside Industrial Park. The assessed site was an untreed field running alongside Burnside Drive towards its intersection with Akerley Blvd, both major, high-traffic roads. The site is separated from the sports fields in Don Bayer Sports Park by a damp, shrubby ditch. The site is underlain by rocky soil and contains many non-native and invasive species. There is little evidence of anyone ever spending time in this field, except for one set of off-road vehicle tracks through the middle of the field, despite the fact that it is an excellent wild strawberry patch.

Opportunities: The site's location in an industrial park, alongside two major roadways, suggests it is not a well-visited place. Traffic noise is loud and continual, the field is thick with prickly vegetation, and there is no shade. There would be few recreational or cultural demands for the space to balance with ecological goals. Many native seedlings could be planted, and large patches of invasive species removed without concern for aesthetics.

Challenges: The adjacent high-traffic road presumably acts as a vector for the seeds of many invasive species. Any removal of current plants would need to be followed up with immediate planting and monitoring to ensure that invasive species are not establishing. Many years will be required for sufficient canopy cover to be in place for many forest understory species to establish. The area might not achieve high degrees of perceived naturalness due to the noise from the nearby roads.

Dimension	Score
Native tree proportion	0
Native basal area	0
Tree canopy FEC concordance	0
Tree cover IAS	N/A
Tree density	0
Large/old	0
Native ground flora	0.41
Shrub layer FEC concordance	0
Ground flora IAS	0.79
Native natural regeneration	0
Seedling density	0
CWD volume	0
Canopy differentiation	0
Soundscape	0
Potential for interior conditions	0
Canopy coverage	0



Flinn Park (A)

Flinn Park is located at the west end of Quinpool Avenue next to the railway track. The park is set a little ways off the road on sloping land adjacent to a residential area and the rail line. Much of the park consists of untreed mowed lawn with paths or play structures. Trees and more-natural areas are found primarily on the sloped portions of the park; Flinn Park (A) is the eastern portion of a group of trees near where the main sidewalk intersects with MacDonald Street.

Opportunities: There is already a well-developed, fairly native tree canopy in place in Flinn Park. However, the natural regeneration is less native than the current canopy – removing the non-native species, and planting native species that align with the FEC type, could help foster a more-natural site. Increasing the amount of coarse woody debris would be a simple way to both increase the score on that naturalness dimension and increase the habitat available for different kinds of native species.

Challenges: The small size of the site means that edge effects, and thus the potential for conditions favourable to the many shade-intolerant non-native species, will be pervasive. The understory of the site is currently dominated by invasive alien species (63%). Intensive removal and subsequent monitoring efforts would be necessary.

Dimension	Score
Native tree proportion	0.79
Native basal area	0.94
Tree canopy FEC concordance	0.33
Tree cover IAS	0.9
Tree density	1
Large/old	0
Native ground flora	0.1
Shrub layer FEC concordance	0.1
Ground flora IAS	0.37
Native natural regeneration	0.7
Seedling density	1
CWD volume	0.22
Canopy differentiation	1
Soundscape	0
Potential for interior conditions	0
Canopy coverage	1

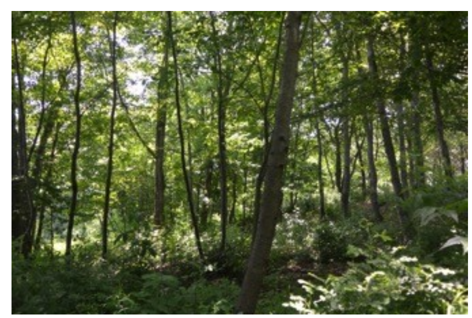
Flinn Park (B)

Flinn Park is located at the west end of Quinpool Avenue next to the railway track. The park is set a little ways off the road on sloping land adjacent to a residential area and the rail line. Much of the park is untreed, mowed lawn with paths or play structures. Trees and naturalized areas are found primarily on the slopes of the park; Flinn Park (B) is the western portion of a group of trees along the main sidewalk, and is crisscrossed by well-used several footpaths. Garbage was frequently encountered during field visits.

Opportunities: Interestingly, the forest canopy in Flinn Park B is less natural than in Flinn Park A, despite the fact that they are immediately adjacent stands separated only by ten to fifteen metres. Increasing the canopy naturalness is a good priority for Flinn Park B. The non-native trees could be cut down to increase the volume of coarse woody debris while simultaneously planting native tree species. It would be important to ensure that tree removal does not create too-large openings. Otherwise, shade-intolerant forest species would be promoted.

Challenges: Similarly to Flinn Park A, the small size of the stand means that naturalness can only be fostered to a certain point. The site cannot support interior conditions, and edge effects will penetrate throughout the site. There is a large presence of invasive alien species in the stand; their removal and subsequent monitoring would require high commitment by managers.

Dimension	Score
Native tree proportion	0.65
Native basal area	0.46
Tree canopy FEC concordance	0.25
Tree cover IAS	0.96
Tree density	1
Large/old	0
Native ground flora	0.08
Shrub layer FEC concordance	0.12
Ground flora IAS	0.66
Native natural regeneration	0.63
Seedling density	1
CWD volume	0.09
Canopy differentiation	1
Soundscape	0.5
Potential for interior conditions	0
Canopy coverage	0.88



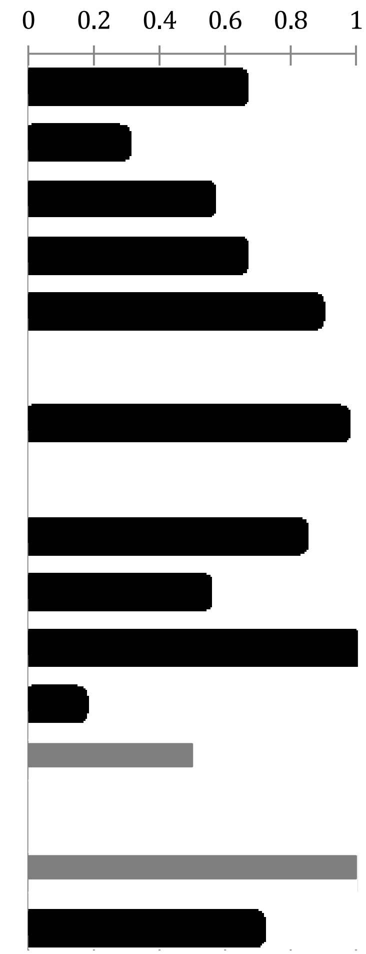
Fort Needham Park (west)

Fort Needham Park is found in the Hydrostone neighbourhood of Halifax's north end. The park has an important place in local history as one can stand on top of the hill and look down to the location in the harbour where two ships collided in the Halifax Explosion of 1917, and across the neighbourhood that burnt following the explosion. A memorial statue is located at the top of the hill, and is accessible by footpaths. The forest to the west of the memorial is dense and has a well-developed understory of native plants. Several trampled footpaths cross through the forest, and garbage is frequently encountered.

Opportunities: The site has a dense native understory, which contrasts strongly with and serves as a guide for the understory of Fort Needham Park (east) on the other side of the slope. The shrubs are not entirely concordant with the species list from the FEC type; however, this may be because the canopy is made up of many non-native species. Shifting the canopy composition through planting more native seedlings would increase the nativeness of the natural regeneration while also increasing the native proportion of tree canopy individuals and basal area.

Challenges: The site borders a residential area and a fairly busy road. Edge influences will penetrate into the majority of the stand, and the footpaths will facilitate invasion by many alien species. Some alien invasive species were found along the paths just outside of the sampled area, such as multiflora rose (*Rosa multiflora*). Since the native ground cover is so dense, it may be difficult for invasive alien species to establish; ensuring that the native ground cover remains may be an important component of site management.

Dimension	Score
Native tree proportion	0.65
Native basal area	0.29
Tree canopy FEC concordance	0.55
Tree cover IAS	0.65
Tree density	0.88
Large/old	0
Native ground flora	0.96
Shrub layer FEC concordance	0
Ground flora IAS	0.83
Native natural regeneration	0.54
Seedling density	1
CWD volume	0.16
Canopy differentiation	0.5
Soundscape	0
Potential for interior conditions	1
Canopy coverage	0.7



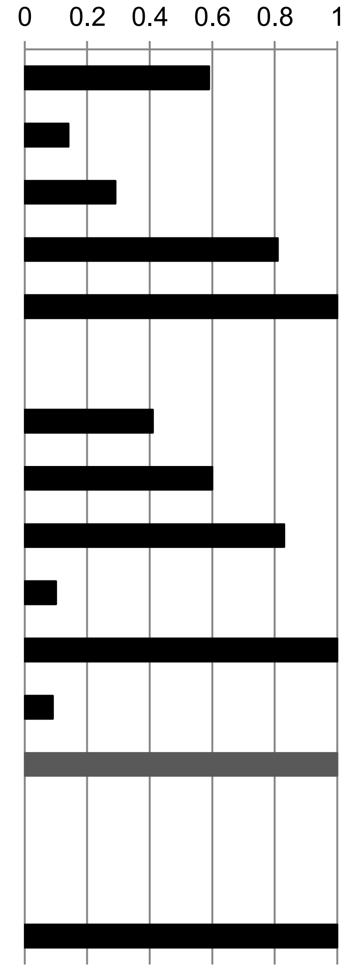
Fort Needham Park (east)

Fort Needham Park is found in the Hydrostone neighbourhood of Halifax's north end. The park has an important place in local history as one can stand on top of the hill and look down to the location in the harbour where two ships collided in the Halifax Explosion of 1917, and across the neighbourhood that burnt following the explosion. The treed region on the eastern slope of the memorial primarily consists of alien tree species, and contains a patch of the highly invasive shrub Japanese knotweed (*Fallopia japonica*). A trampled footpath cuts through the stand, and garbage is abundant in the sparse understory. The forest borders a residential area on two sides and other portions of the park.

Opportunities: The removal of Japanese knotweed and seedlings of alien tree species should be priorities for the site, coupled with the planting of native seedlings and ground flora. The primary aim for this site should be to shift the species composition of the plant community.

Challenges: Norway maple (*Acer platanoides*) and Norway spruce (*Picea abies*) dominate the canopy; it will take time to transition the site to a more-natural stand structure. Long-term monitoring and removal will be necessary to control Japanese knotweed and continued planting of native seedlings may be necessary over several years. The costs for naturalization will be initially high for this site. Its location within the city, its small size, and the constant human traffic also limit its potential for more-natural, undisturbed conditions.

Dimension	Score
Native tree proportion	0.59
Native basal area	0.14
Tree canopy FEC concordance	0.29
Tree cover IAS	0.81
Tree density	1
Large/old	0
Native ground flora	0.41
Shrub layer FEC concordance	0.6
Ground flora IAS	0.83
Native natural regeneration	0.1
Seedling density	1
CWD volume	0.09
Canopy differentiation	1
Soundscape	0
Potential for interior conditions	0
Canopy coverage	1



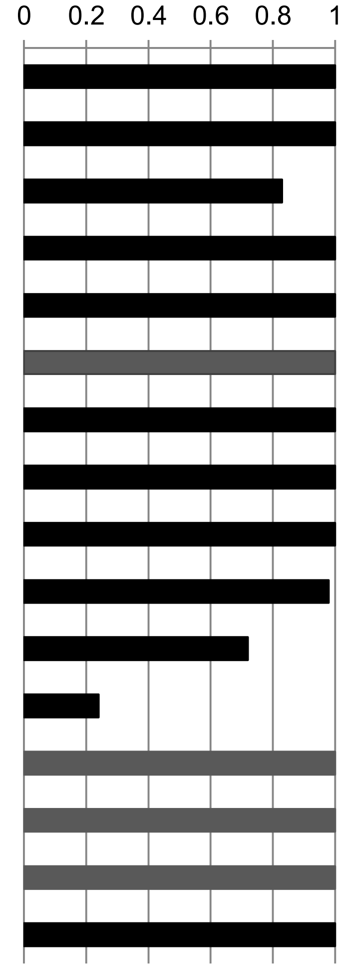
Hemlock Ravine

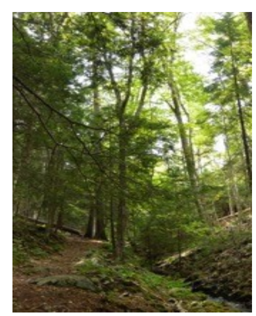
Hemlock Ravine is a 200-hectare park located north of the Halifax peninsula, just off the Bedford Highway. Originally owned by the Lieutenant Governor of Nova Scotia in the 1700s, the park is now jointly owned and maintained by the HRM and the Province of Nova Scotia. It is surrounded by residential areas and major roadways, which provide a constant low level of noise. The park has several access points with parking lots along its eastern edge, but trail usage is minimal during the daytime (only two individuals were encountered during three field visits).

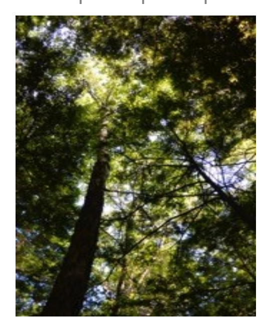
Opportunities: The large size of the park and the position of the site within the park make high degrees of naturalness possible. Human encroachments into the site seem to be few. The main dimension that could be improved is the volume of coarse woody debris – the average volume in non-urban spruce-hemlock stands is ~25 m^3/ha, which is even higher than the minimum against which our deadwood scores were calibrated. Deadwood might be removed if it interferes with the path network, but could be left in piles throughout the remainder of the site.

Challenges: Dimensions that were not assessed within our study may be difficult to increase, such as those relating to faunal communities. However, its vegetation communities seem to be highly natural. A concurrent study on old-growth characteristics in urban settings found that urban stands can achieve the same vegetational composition and structure as non-urban old-growth.

Dimension	Score
Native tree proportion	1
Native basal area	1
Tree canopy FEC concordance	0.83
Tree cover IAS	1
Tree density	1
Large/old	1
Native ground flora	1
Shrub layer FEC concordance	1
Ground flora IAS	1
Native natural regeneration	0.98
Seedling density	0.72
CWD volume	1
Canopy differentiation	1
Soundscape	1
Potential for interior conditions	1
Canopy coverage	1







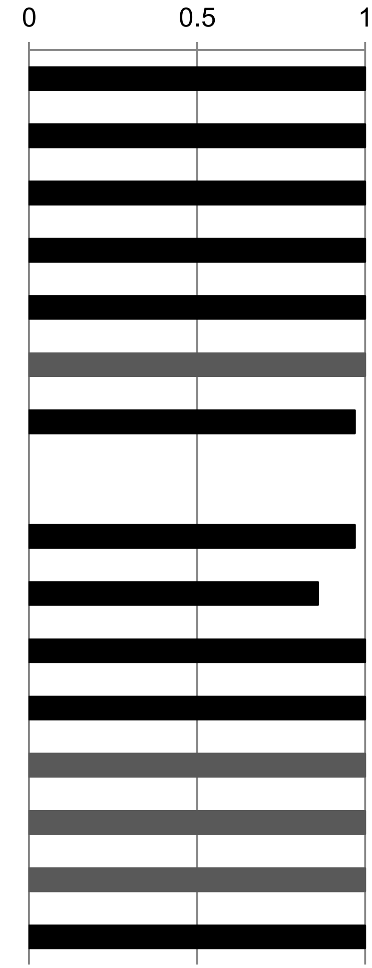
McBeth Park (inland)

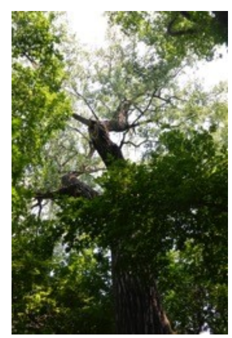
McBeth Park is located in the North Main region of Winnipeg along the banks of the Red River. The portions of the park farther away from the river are not necessarily submerged during spring flooding; the ground flora form a thick carpet and there are large piles of downed woody debris, presumably deposited by seasonally high water. The park is bordered on three sides by residential development. There are formal and informal paths throughout; semi-permanent structures have been erected along the park boundaries by nearby residents, and one of the largest trees has been the victim of arson. McBeth Park is believed to contain the oldest and largest trees in Winnipeg.

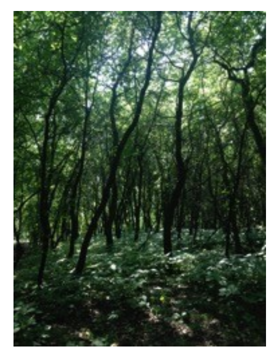
Opportunities: The inland region scored higher on several naturalness dimensions than the riverside portion of McBeth Park, due to the amount of deadwood and potential for interior conditions. Potential naturalization actions are few – the few non-native seedlings and ground flora species could be removed, and subsequent monitoring could occur to ensure that native seedlings are establishing.

Challenges: This portion of the park does not seem to be seasonally inundated; changes in the river levels may affect the site indirectly through decreasing the success of nearby, riverside plants which could act as seed sources or barriers to edge conditions. The trees may be affected by trampling – many of the large trees are right off the footpaths, and the soil is packed down around their roots. It may be difficult to limit this trampling, as people are drawn towards being near (and potentially photographing being near) large trees.

Dimension	Score
Native tree proportion	1
Native basal area	1
Tree canopy FEC concordance	1
Tree cover IAS	1
Tree density	1
Large/old	1
Native ground flora	0.97
Shrub layer FEC concordance	0
Ground flora IAS	0.97
Native natural regeneration	0.86
Seedling density	1
CWD volume	1
Canopy differentiation	1
Soundscape	1
Potential for interior conditions	1
Canopy coverage	1







McBeth Park (riverside)

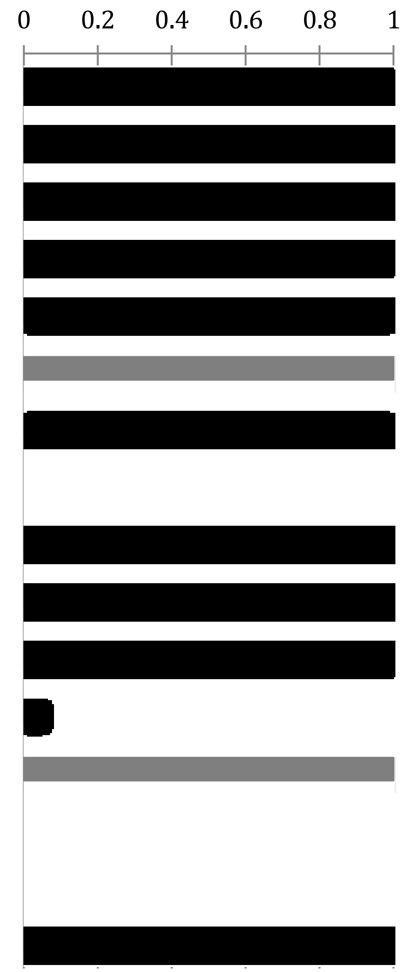
McBeth Park is located in the North Main region of Winnipeg, Manitoba. It is located along the banks of the Red River, north of the large and popular Kildonan Park. The annual flooding and periodic large floods have shaped the vegetation communities growing along the river. In recent years, more-intense floods have become more frequent, and a greater area of the park is submerged for greater periods of time.

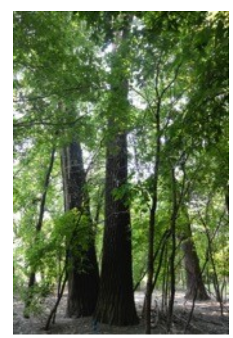
The park is bordered on three sides by residential development. There are formal and informal paths throughout; semi-permanent structures have been erected along the park boundaries by nearby residents; and one of the largest trees has been the victim of arson. McBeth Park is believed to contain the oldest and largest trees in Winnipeg.

Opportunities: The age and size of the cottonwoods (*Populus deltoides*) makes this park a unique place in Winnipeg. Naturalization activities, even if they limit human encroachment into some areas, may find a lot of support due to the positive values many residents may associate with the persistence of large, old trees.

Challenges: The changing frequency and duration of the flooding cycle may fundamentally alter which plants are able to establish and grow within the park. It may be difficult to increase shrub FEC concordance with increased flooding if young plants are too stressed during submerged periods in spring; young canopy members may also die off if flooding is too stressful.

Dimension	Score
Native tree proportion	1
Native basal area	1
Tree canopy FEC concordance	1
Tree cover IAS	1
Tree density	1
Large/old	1
Native ground flora	1
Shrub layer FEC concordance	0
Ground flora IAS	1
Native natural regeneration	1
Seedling density	1
CWD volume	0.06
Canopy differentiation	1
Soundscape	0
Potential for interior conditions	0
Canopy coverage	1





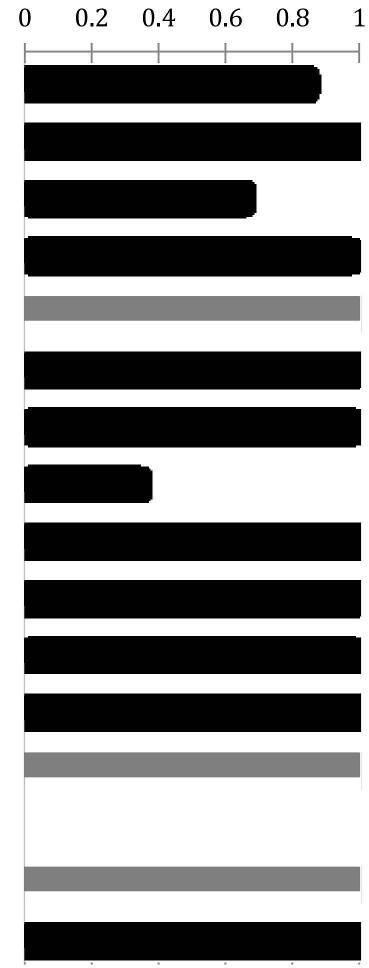
Point Pleasant Park

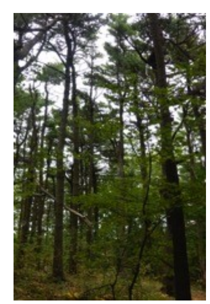
Point Pleasant Park is a 77-hectare park located on the easternmost point of the Halifax peninsula. It became a park in 1866 when it was leased by the city from the federal government, an arrangement that persists to this day. Prior to that, the park provided many resources to settler and Aboriginal populations. In 2003, a large percentage of the canopy was blown down during Hurricane Juan. An extensive replanting of the forest (101 000 trees) occurred several years later; much of the park is regenerating with a human hand. Several pockets of old-growth forest survived the hurricane, and are located in the western reaches of the park. The park is extremely popular for walking and as an off-leash dog park, particularly on the weekends. The park is traversed by many trails, some covered with crusher dust, others trampled. A container loading facility is located along the northeastern perimeter of the park, and is often very noisy.

Opportunities: The site we assessed is only a small area located within the much larger park. Interior conditions would be possible in some portions of the site. Many species poorly adapted to the urban environment might be able to establish and live within this park due to its high naturalness, large size, and semi-isolation from dense urbanization. There are opportunities to alter the understory composition through planting seedlings and native shrubs to increase the FEC concordance.

Challenges: The soundscape of the park cannot be altered, and it is the strongest detractor from the site's perceived naturalness. Urban residents are unlikely to linger in this area during periods of activity at the container facility.

Dimension	Score
Native tree proportion	0.86
Native basal area	1
Tree canopy FEC concordance	0.67
Tree cover IAS	0.99
Tree density	1
Large/old	1
Native ground flora	1
Shrub layer FEC concordance	0.36
Ground flora IAS	1
Native natural regeneration	1
Seedling density	1
CWD volume	1
Canopy differentiation	1
Soundscape	0
Potential for interior conditions	1
Canopy coverage	1





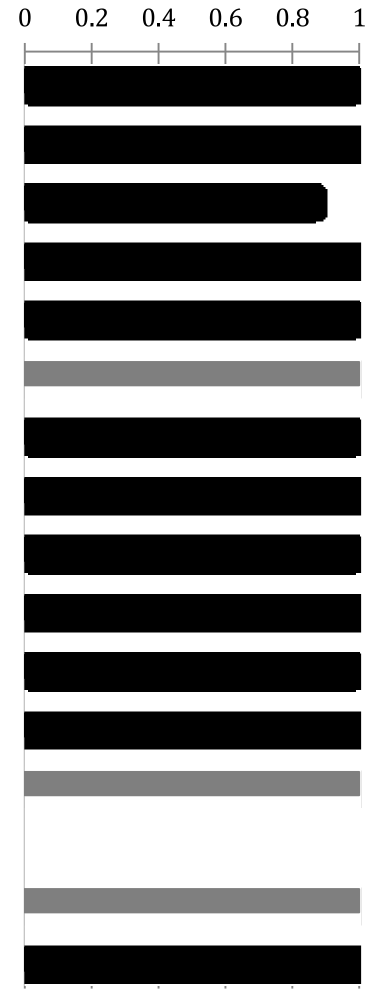
Shubie Park

Shubie Park is a 40-acre park located alongside Highway 118 and between Lake MicMac and Lake Charles along the Shubenacadie Canal. The park was formerly part of an estate that acted as a woodlot for the navy, providing trees for shipmasts. Portions of the estate became industrial developments, and undeveloped regions were set aside as municipal parks in the 1960s and 1970s. Portions of the park are used for beach access, camping, and an interpretive centre on the Shubenacadie Canal. Walkers and dogs are very common on the park's many trails. The nearby highway provides a constant hum of traffic.

Opportunities: The site scores highly on almost all the assessed naturalness dimensions. Changing the soundscape of the site is unlikely. Ensuring that the site continues to be an old-growth hemlock forest would require a comparison between its current conditions and the understory/seedling composition in non-urban old-growth stands. Increasing the site's naturalness may require determining what is absent, rather than what is present

Challenges: The site is very small, occupying little more than the area of the three plots that were assessed, and is bisected by a well-used walking path. Interior conditions would be impossible to foster. High levels of exposure and disturbance from the site's size and positioning could negatively affect some processes (e.g. the recruitment of some species).

Dimension	Score
Native tree proportion	1
Native basal area	1
Tree canopy FEC concordance	0.88
Tree cover IAS	1
Tree density	1
Large/old	1
Native ground flora	1
Shrub layer FEC concordance	1
Ground flora IAS	1
Native natural regeneration	1
Seedling density	1
CWD volume	1
Canopy differentiation	1
Soundscape	0
Potential for interior conditions	1
Canopy coverage	0.88



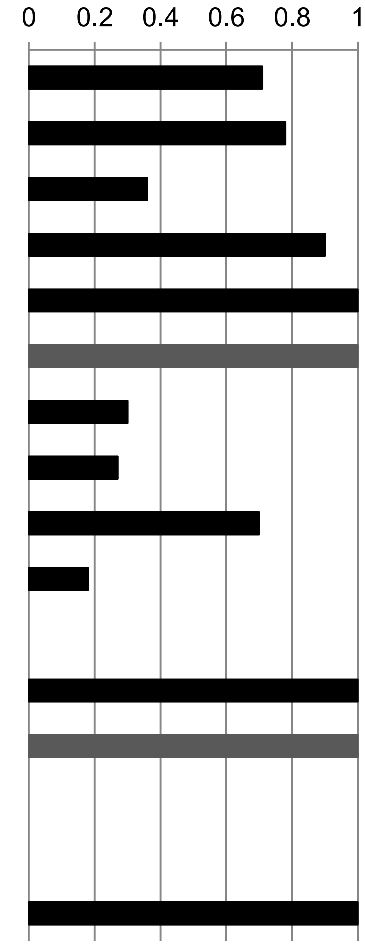
St. Mary's Boat Club (forest)

St. Mary's Boat Club, built in 1905, was renovated and reopened by the Halifax Regional Municipality (HRM) in 1991. There is a single footpath along the perimeter of the forested region to the southeast of the boat club, and a picnic table by the waterfront. One portion of the path runs through a dense raspberry thicket, and another portion runs down a steep slope and is overgrown with *Aegopodium*; the trail thus does not appear to be well used. The stand is bordered by posh residential properties on two sides, mowed grass along the north edge, and the ocean.

Opportunities: A primary task for increasing the site's naturalness would be removing the many invasive ground flora species and planting native ground flora in their stead. While seedlings are abundant, less than 20% are native species. Removing the alien seedlings while planting native species would increase the naturalness of the site. Human use appears to be limited; few recreational objectives would need to be balanced against ecological targets.

Challenges: Alien and/or invasive ground flora and seedlings occupy much of the understory. Naturalization would thus need an extensive removal, replanting, and monitoring program to increase the naturalness of the understory.

Dimension	Score
Native tree proportion	0.71
Native basal area	0.78
Tree canopy FEC concordance	0.36
Tree cover IAS	0.9
Tree density	1
Large/old	1
Native ground flora	0.3
Shrub layer FEC concordance	0.27
Ground flora IAS	0.7
Native natural regeneration	0.18
Seedling density	0
CWD volume	1
Canopy differentiation	1
Soundscape	0
Potential for interior conditions	0
Canopy coverage	1



St. Mary's Boat Club (waterfront)

St. Mary's Boat Club, built in 1905, was renovated and reopened by the Halifax Regional Municipality (HRM) in 1991. The property surrounding the boat club itself is used recreationally for activities such as dog-walking and private functions. The grassy waterfront portion to the north of the club is popular for weddings. The boat club is surrounded by posh residential development.

Opportunities: The naturalness of the wooded portion of the site could be increased through planting seedlings of native tree species, particularly those that would increase the site's FEC concordance. While seedling density is high, less than half is composed of native species. Planting native seedlings would increase stand density, native basal area, native tree proportion, and native natural regeneration. Tree planting in the grassy regions could be coupled with both invasive removal, as many invasives were found in the untreed portions of the site, and planting of native ground flora species. Those species with attractive flowers could be favoured, due to the adjacent area's popularity for private functions and thus potential resistance to understory establishment.

Challenges: The area's popularity for private functions such as weddings may mean there is a lack of interest in re-establishing an understory, or an initial resistance to a growing canopy that would block sightlines. Mowed lawn may be preferable. Past community tree-planting initiatives were unsuccessful, as a nearby resident uprooted many seedlings to maintain a clear view of the ocean.

Dimension	Score
Native tree proportion	0.7
Native basal area	0.47
Tree canopy FEC concordance	0.43
Tree cover IAS	0.93
Tree density	0.4
Large/old	1
Native ground flora	0.18
Shrub layer FEC concordance	0
Ground flora IAS	0.67
Native natural regeneration	0.38
Seedling density	1
CWD volume	0.07
Canopy differentiation	1
Soundscape	1
Potential for interior conditions	0
Canopy coverage	0.76