

VEGETATION COMPOSITION AND STRUCTURE AS INDICATORS OF
ECOLOGICAL INTEGRITY WITHIN URBAN PARKS OF HALIFAX REGIONAL
MUNICIPALITY

by

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Abstract

Conserving biodiversity is fundamental to promoting ecological integrity in urban environments. As a type of protected area, urban parks are important places for the conservation of indigenous plant communities within an otherwise inhospitable matrix of anthropogenic infrastructure. However, their vegetation is influenced by a number of stressors associated with management, recreation, fragmentation, and disturbance. In the present study, the influence of these stressors was examined within urban parks of Halifax Regional Municipality, Nova Scotia. Compositional and structural measures of vegetation were examined within habitat-types, identified and grouped according to the relative intensities of management activities, recreation, and hurricane disturbance to which they were subject. Gradients in vegetation composition as a function of distance from the edge of anthropogenically maintained forest boundaries and recreational trails were also examined. Plant communities within the urban parks were found to vary considerably in character, and ranged from natural forest remnants dominated by communities of native taxa to structurally simpler anthropogenic ones comprised mostly of exotics. Within remnants of natural forest, land-use legacies and edge influences significantly affected vegetation, particularly by increasing the prominence of exotics. Exotic plants were not more abundant within sites that were severely disturbed by the hurricane event compared to more intact ones. Randomization tests suggest that forest boundaries and trails act in an additive manner to affect vegetation composition, and that they exert a distance-of-edge-influence, on measures of exotic plants, of 40-60 m and at least 4-6 m, respectively. These results may be used to help enhance ecological integrity within urban ecosystems by directing naturalization efforts within anthropogenic habitats and providing guidelines for the conservation of interior forest conditions.

List of Abbreviations

CCA – Canonical Correspondence Analysis

CWD – Coarse Woody Debris

DBH – Diameter at Breast Height

DCA – Detrended Correspondence Analysis

DEI – Distance of Edge Influence

df – Degrees of Freedom

EI – Ecological Integrity

F – F-values

HRM – Halifax Regional Municipality

INSPAN – Indicator Species Analysis

IQR – Interquartile Range

IV – Indicator Value

MSE – Mean Square due to Error

MST – Mean Square due to Treatment

P – P-values

PFG – Plant Functional Group

SE – Standard Error

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Chapter 1: Introduction

Parks have long been regarded as an important aspect of city life (Welch 1991; Jones and Wills 2005). However, their intended function, and subsequent design and management, has changed considerably with time in response to the perceived social problems of the day (Cranz 1982). As such, what may be considered to constitute a “park” is varied and adaptable. For the purposes of this document, urban parks are considered to be municipally-managed green spaces in cities that offer outdoor recreational opportunities to city residents. Although urban parks have traditionally expressed various ideas about nature, they were not intended to address ecological issues (Cranz and Boland 2004). However, as ecological concerns increase in importance, a new approach to park design and maintenance is required.

Urbanization

Urbanization is a dominant demographic trend and an important component of global land transformation (Pickett et al. 2001). The urban growth rate worldwide is approximately twice that for the total population. Between 1960 and 2009, the number of people living in urban areas globally has grown from about 1 billion to more than 3 billion; and year 2007 marked the first time in world history that more than half of the population lived in cities. This trend is expected to intensify so that by the year 2050, more than 6 billion people or almost 70% of the global population will live in urban areas (United Nations 2007). In Canada, approximately 80% of the country’s population lives in cities and this is expected to increase to 88% by 2050 (United Nations 2007). In contrast, only 13% of Canadians lived in cities in 1851 (Statistics Canada 2007). Patterns of urbanization are also prominent within Nova Scotia – while rural counties in the province are becoming less populated, urban areas are growing, largely because of immigration. Between 2001 and 2006, Halifax Regional Municipality (HRM) experienced a population increase of 3.8% - more than six times that experienced by the province as a whole (Statistics Canada 2007), and this growth is predicted to continue (HRM 2006a).

Urbanization causes profound changes to local ecosystems, the most obvious being the destruction and fragmentation of natural habitat, which are considered the main threats to biodiversity worldwide (Saunders et al. 1991; Drinnan 2005; Freedman 2009). The conversion of natural habitat to urban land-use types involves severe disturbance, the creation of large amounts of impervious surfaces, and intense management regimes (Bryant 2006). The resulting infrastructure is often the most geographically ubiquitous human influence (McKinney 2006) and is typically impoverished of native species, dominated by exotic taxa, and lacks many of the structural or functional characteristics of the natural ecosystems it replaced (Kowarik 1990; Freedman et al. 1996; Turner et al. 2005). Such conditions are maintained by ongoing management activities whose horticultural initiatives typically favor exotics and disrupt natural successional processes by imposing frequent disturbance events (e.g. through activities such as mowing) (Niemelä 1999). Furthermore, pollution by toxic chemicals, heat, nutrients, noise, and biological pathogens is often relatively high in urban areas compared to more rural ones (Pickett et al. 2001; Freedman 2009).

Natural habitats that persist within the urban landscape are highly fragmented. Fragmentation influences vegetation and structure within forest remnants by reducing their area, increasing their isolation, and causing a proliferation of edges (Kupfer et al. 2006). Studies examining the effects of fragmentation on the species richness of urban forest remnants have found that smaller and more isolated remnants have fewer plant species, as is predicted under the theory of island biogeography (Davis and Glick 1978; Bastin and Thomas 1999; Guirado et al. 2006). In addition, edge influences act to increase the risk of species extirpation and encourage the colonization of non-forest and exotic taxa (Godefroid and Koedam 2003a; Guirado et al. 2006). Urban forest remnants are also subject to management practices, such as mowing and plantings (Hobbs 1988), and recreational activities (Bhuju and Ohsawa 1998), both of which may stress indigenous communities.

The ecological effects of urbanization are more permanent than those of other forms of anthropogenic habitat loss. For example, although forests in northeastern North America are regenerating following agriculture and logging, most urban areas continue to grow in size (McKinney 2002). Furthermore, the effects of urbanization may increase in

severity with time: studies that have examined temporal changes in urban floras have found that native species richness declines while that of exotics increases (Drayton and Primack 1996; Chocholoušková and Pyšek 2003; Standley 2003; DeCandido and Gargiullo 2004; Tait et al. 2005). Unfortunately, this means that people who live in cities are continually exposed to habitats which are not “natural” in character. As such, people are becoming increasingly unfamiliar with and disconnected from the native ecological environment (Noss 2004; Turner et al. 2004; McKinney 2006).

Ecological Integrity

Ecological integrity (EI) is a holistic concept that encompasses other ecological notions, such as biodiversity, ecosystem and environmental health, sustainability, naturalness, wildness, stability, and resilience (Freedman 1993; Noss 1995; Andreasen et al. 2001; Turner and Beazley 2004). Although value judgments are inherent in the concept, there is a consensus that ecosystems with high levels of EI are subjected to relatively low intensities of anthropogenic stressors and are comprised of components of a naturally self-organizing system (as opposed to being maintained by human activities). Such attributes include a variety of compositional, structural, and functional measures, including (Freedman 2009):

- relatively high resistance and resilience to changes in the intensity of environmental stressors,
- richness of indigenous biodiversity,
- complexity in structure and function,
- presence of large species and top predators,
- controlled nutrient cycling (i.e., nutrient capital is not “leaked” to the ambient environment), and
- the ecosystem is self-maintaining, and does not require anthropogenic management to conserve its key attributes.

Because EI is being widely adopted as a criterion for management and conservation initiatives, there is a need for clear operational definitions. To this end, Parks Canada identifies EI as being (Parks Canada Agency 2000):

"...a condition that is determined to be characteristic of its natural region and likely to persist, including abiotic components and the composition and abundance of native species and biological communities, rates of change and supporting processes."

Whether (or not) urban areas can reflect aspects of a system with EI is an important question for people who live in cities, and is becoming increasingly relevant at a global scale as more of the world becomes urbanized (Noss 2004). Although EI is increasingly being adopted as an umbrella concept that guides the stewardship of protected areas and initiatives in ecologically sustainable resource use, it has not yet been fully embraced for the design and management of urban areas.

Indicators

Quantitative measures of the condition of ecosystems are required to make concepts such as EI operational. As such, ecological indicators that convey information on EI are sought. For the purpose of this document, ecological indicators are defined as "measurable characteristics of the structure (genetic, population, habitat, and landscape pattern), composition (genes, species, populations, communities, and landscape types), or function (genetic, demographic/life history, ecosystem, and landscape disturbance processes) of ecological systems" (Niemi and McDonald 2004) that relay information on ecological processes. However, to serve as indicators of EI they must convey information on anthropogenic stress.

Ideally, ecological indicators should be easily measured, have a known sensitivity to particular stressors, forecast changes that can be mitigated by management actions, be integrative of a range of conditions, and have good accuracy and precision of response (Dale and Beyeler 2001). A major challenge in the use of ecological indicators is the need to detect a response to variations of intensity of anthropogenic stressors (Karr 2004) against a background of natural variability (Frost et al. 1992). There is no perfect indicator of EI, and so trade-offs must be made between desirable features, costs, and feasibility (Dale and Beyeler 2001). The utility of particular indicators varies over spatial and temporal scales, and it is therefore important to understand these contexts when they are being established or used (Simberloff 1998). Because of deficiencies of any particular

indicator, a set of complementary ones is necessary (Noss 1990), especially for reporting on multifaceted concepts such as EI.

The selection of ecological indicators is based on knowledge of the ways that they reflect key environmental stressors and/or their effects on species, communities, or other aspects of ecosystems. They are relatively simple measures in comparison to the complexity of ecosystems. Many components of vegetation may be helpful as indicators of EI depending on the context - including particular species, groups of taxa defined by common trait(s), community indices, stand and landscape-level features, and multimetric indices (LaPaix et al., 2009).

Indicators of EI may be identified analytically by testing candidate measures along gradients of one or more stressors that are associated with ecological changes. Particular stressors may be used as surrogates for EI if they are identified as dominant factors influencing ecological responses within a given context. For example, measures of anthropogenic disturbance are commonly used to arrange sites along a general gradient of EI to which candidate indicators are assessed (e.g., Kimberling et al. 2001; DeKeyser et al. 2003). Measures that respond in a unimodal fashion across a gradient of EI are particularly useful indicators. Because of the complexity of ecosystems and the holistic nature of the EI concept, it can be difficult to establish indicators of EI in a purely quantitative way. As such, information from observational and experimental studies, coupled with insight from ecological theory and knowledge of ecoregional regimes of processes and stressors, are helpful for evaluating candidate measures.

Urban Parks

As one type of protected area, urban parks are important for promoting ecological integrity within cities. For example, they may contain remnant natural areas that act as refugia for many native species (McDonnell and Pickett 1990; Bastin and Thomas 1999; Drinnan 2005). In highly fragmented landscapes, networks of natural-habitat patches, of varying quality, may be crucial to the survival of populations of native plants (Bastin and Thomas 1999). Moreover, by exposing large numbers of people to natural or semi-natural habitats and their species, urban parks have the potential to foster a greater sensitivity to environmental issues by increasing awareness of and appreciation for native biodiversity

and healthy ecosystems (Sebba 1991; Rohde and Kendle 1994; Chiesura 2004; Noss 2004; Turner et al. 2005). In addition, they are important for conserving the quality of water, air, and noise within cities (Drinnan 2005).

Despite their ecological importance, the history of urban parks in America reveals more concern with social problems. According to Cranz (1982) park design has gone through a series of stages – from pleasure grounds (1850-1900), to reform parks (1900-1935), to recreational facilities (1930-1965), and to open space systems (1965+). Each of these phases reflects shifts in what were considered the more urgent social problems of the time (such as public health, social reform, assimilation, and recreation) and corresponding changes in park design. More recently, a fifth park model has been identified, which may be termed the “sustainable” or “ecological” park (Cranz and Boland 2004). The major functions of this park type is contributing to the ecological sustainability of cities and helping to improve quality of life in doing so. According to Cranz (2004), the “sustainable” park emerged in the late 1990s and is characterized by three general attributes: (1) self-sufficiency with regard to material resources and maintenance, (2) contributes to solving larger urban problems outside of park boundaries, and (3) creates new standards for aesthetics and landscape management in parks and other urban landscapes. The emergence of this park type has been a response to the inability to treat ecological and social issues independently in today’s cities.

Within HRM, the primary goal of urban parks is the provision of outdoor recreation (HRM 2006a). However, they are also considered valuable for the conservation of natural ecosystems within the broader context of environmental stewardship (HRM 2006b), and so are managed to conserve biodiversity, maintain cultural heritage, and promote the “quality of life” of urban residents (HRM 2006a). However, the design and intended function of particular parks varies considerably. They range from small areas designed primarily for recreation to relatively large spaces aimed at protecting important natural or cultural resources (HRM 2006a).

Urban parks also provide unique opportunities for ecological research. For example, certain ecological processes (e.g., exotic invasions) may be more prevalent within urban areas than rural ones (Niemelä 1999). As such, knowledge gained from studies in urban areas may provide important insights that can be applied to more rural

systems. In this context, remnant forest patches within urban ecosystems provide excellent opportunities to investigate long-term anthropogenic impacts on forest ecosystems (McDonnell and Pickett 1990). However, compared to other disciplines there has been relatively little biodiversity research directed at urban parks and to the field of urban ecology in general (Cornelis and Hermy 2004; Drinnan 2005). Unfortunately, poor documentation of biodiversity within cities limits possibilities for applying ecological knowledge to urban planning (Niemelä 1999).

Purpose and Outline of Thesis

To maintain biodiversity within urban ecosystems, Niemelä (1999) identifies three major steps for understanding ecological patterns and processes: (1) describe the nature that exists, (2) obtain knowledge about ecological processes that are important for urban ecosystems, and (3) design management schemes based on knowledge of these processes. Within that context, the purpose of this study is to identify and describe elements of vegetation structure and composition that are sensitive to a suite of anthropogenic stressors, and to use this information to provide insight into ecological processes which may be used for promoting EI within urban parks. Chapter 2 explores relationships among plot-based measures of vegetation, management activities, fragmentation, and natural disturbance within urban parks. In doing so, a number of potentially valuable indicators of EI are identified. Chapter 3 applies knowledge of the relationship between exotic plants and fragmentation to quantify the extent of edge influences within urban forest remnants. As such, this chapter demonstrates how indicators may be used to address practical questions related to ecological processes and the design and management of protected areas. Specific objectives for Chapters 2 and 3 are outlined in their relevant chapter introductions. Chapter 4 provides further discussion of the EI concept within an urban context, particularly the use of an indicator approach for its quantification.

Chapter 2: Influences on Vegetation in Urban Parks

Abstract

Urban parks are important places for the conservation of biodiversity within cities, but their vegetation is influenced by a number of anthropogenic stressors. This study took an exploratory approach to examining the influence of management, land-use legacies, natural disturbance (from a major hurricane), and fragmentation-related factors on compositional and structural indicators of vegetation within urban parks of Halifax Regional Municipality, Nova Scotia. Study sites were selected using a stratified random sampling procedure, based on the size of urban parks. Plots of 10 m x 10 m were randomly distributed throughout the sites and used to quantify plant composition, forest structural attributes, and environmental variables. Variation in composition was described using species and plant functional groups, which were identified by combining information on growth form, life history, and biogeographical status. Plant communities within the studied urban parks varied considerably in character, ranging from remnants of natural forest dominated by an array of native taxa, to structurally simple anthropogenic ones comprised mostly of exotics. Historical use and edge influences (from trails and forest boundaries) significantly affected vegetation within remnants of natural forest, particularly by increasing the prominence of exotic taxa. The intensity of hurricane disturbance was also important for constructing plant communities but was not found to promote exotics. These results may be used to help enhance ecological integrity within urban ecosystems, particularly by directing naturalization efforts within anthropogenic habitats.

Introduction

Urban parks are important for providing recreational and educational opportunities to city residents and often also serve to preserve cultural resources. As a type of protected area, they are also potentially important places for the conservation of indigenous biodiversity within a matrix of human infrastructure. They may be particularly valuable in this respect if they harbor remnants of natural habitat. Such relatively natural protected areas provide a number of important ecological services, including the

provision of habitat for native species. Furthermore, by exposing large numbers of people to semi-natural habitats, parks also help to increase awareness of and appreciation for native biodiversity and healthy ecosystems (Sebba 1991; Rohde and Kendle 1994; Chiesura 2004; Noss 2004).

Anthropogenic habitats may also help to conserve natural values, particularly if they are managed with concepts of naturalization in mind. Urban naturalization initiatives attempt to achieve a pleasing aesthetic, but unlike conventional horticultural practices, they favour native rather than exotic plants (Freedman 2009). Native plants are generally considered more ecologically appropriate than alien ones because they may grow better under local environmental conditions, be more hardy and disease resistant, be less likely to become invasive, support associated native species, contribute to protecting the global store of biodiversity, and are important for conserving the character of regional landscapes (Kendle and Rose 2000; MacQuarrie and Lacroix 2003; Sanz-Elorza et al. 2006; Freedman 2009). These are among the reasons why dominance of communities by native species is considered a key attribute of ecological integrity. As such, many municipalities (including Canadian ones) are actively supporting naturalization efforts in urban parks and other land-use types (Ingram 2001).

Unfortunately, natural-forest remnants within urban parks are often severely fragmented and therefore have large edge-influenced: interior area ratios. Edges influence vegetation composition and structure by creating gradients of disturbance (Harper et al. 2005), resource availability (Gehlhausen et al. 2000), human activity (Guirado et al. 2006), and species' propagules (Cadenasso and Pickett 2001). For example, non-forest species are more frequent (Honnay et al. 2002; MacQuarrie and Lacroix 2003; Guirado et al. 2006) and wind damage to trees often more severe close to the edge of a forest boundary than in the interior (Harper et al. 2005; Mascarúa López et al. 2006). Similarly, recreational trails create edges within natural habitats and are well-known to influence species composition (Adkison and Jackson 1996; Bhujju and Ohsawa 1998; Dickens et al. 2005). For example, they promote synanthropic species (those inhabiting anthropogenic habitats) by acting as corridors for their dispersal and by providing suitable microhabitat (Benninger-Truax et al. 1992; Parendes and Jones 2000).

Despite the ecological importance of urban parks, there have been few investigations of their vegetative character or other aspects of their biodiversity. Although several studies have addressed relationships among plant species richness and the spatial patterning of park features (Hermy and Cornelis 2000; Cornelis and Hermy 2004; Li et al. 2006), such knowledge has limited importance for conservation planning without accompanying information on the identities and relative abundances of the constituent species (including whether they are native or alien). Some other studies have examined changes in vegetation for individual parks over decades (Loeb 1992; Zipperer and Zipperer 1992; Drayton and Primack 1996; DeCandido 2004). The detailed information that may be gained from these studies provides insight into the role of anthropogenic stressors in influencing plant communities. However, results from such studies lack generalization. As such there is a need for greater understanding of how human stressors influence biodiversity within urban.

The present study describes the influence of some dominant anthropogenic stressors on vegetation within urban parks of HRM. More specifically, an exploratory approach was taken to address the following questions:

1. How does vegetation structure and composition vary among habitats that have been subject to different management regimes?
2. How is plant composition within natural forest remnants influenced by fragmentation?

In order to provide context for the role of the anthropogenic factors in affecting plant communities, the influence of an event of severe natural disturbance¹ was also investigated.

Methods

Study Area

The study area (approx. 188 km²) comprises the portion of the urbanized landscape of HRM, Nova Scotia (approx. center at 44° 39' N, 63° 34' W) that is located

¹ Hurricane Juan caused variable amounts of damage to forests within HRM during 2003. Although moderately to highly disturbed areas within some urban parks were subject to clean-up operations, the event(s) is considered here to be predominantly natural, rather than anthropogenic.

within the Eastern Interior Ecodistrict of the Acadian Ecozone (Neily et al. 2003). The ecodistrict is characterized by an undulating to gently rolling topography. Meguma Group quartzite and slates comprise the bedrock of the study area. Soils are of the Halifax, Bridgewater, and Wolfville series and are composed of well-drained, often stony loams from quartzite, slate, and shale/sandstone, respectively (MacDougall et al. 1963). The depth of the till throughout the ecodistrict varies from <1 to 10 m and averages <3 m (Neily et al. 2003).

Stands of forest within the Eastern Interior Ecodistrict are predominantly conifer-dominated, with *Picea rubens* (red spruce) and *Picea mariana* (black spruce) dominating stands with well-to-imperfectly drained to poorly drained soils, respectively. However, the natural forest composition throughout the ecodistrict is varied and reflects the depth of the soil profile and other factors (Neily et al. 2003). For example, shade-intolerant hardwoods such as *Acer rubrum* (red maple) and *Betula papyrifera* (white birch), along with scattered *Pinus strobus* (white pine) and an understory dominated by ericaceous shrubs, are prominent on shallow soils. In contrast, shade-tolerant species, including *Tsuga canadensis* (eastern hemlock) and *Fagus grandifolia* (American beech), may be found on deeper, well-drained sites such as those located on the crests and upper slopes of hills or drumlins (Neily et al. 2003). The dominant natural disturbances affecting the forests within the ecodistrict are wildfires and hurricanes (Neily et al. 2003) and much of the area is presently in a regenerative state following a major hurricane (Juan) in September 2003.

Selection of Study Sites

Study sites were identified with assistance from an HRM GIS layer (HRM 2005) which depicts “municipally owned parks and other parks that the municipality has an interest in” (King 2007). Because this layer was comprised of a variety of land-use types, the list was shortened to exclude land parcels which did not include “park” in their title and/or whose primary function was the provision of sites for fire stations, libraries, schools, the servicing and storage of maintenance equipment, or activities of the Department of National Defense. The remaining sites were then overlaid with a GIS layer depicting boundaries for NSDNR’s Ecological Land Classification. The list of parks was

then limited to those located within the Eastern Interior Ecodistrict (NSDNR 2007), the dominant ecodistrict for the Halifax metro area. From the refined data set ($n = 157$ sites), a stratified random sampling procedure, based on the size of urban parks, was then used to select 24 study sites (Figure 1 and Table 1). Area was used to stratify the selection because it was expected to provide a range of urban parks in terms of the types and intensities of management and recreational activities. Parks were assigned into one of nine geometrically increasing size classes (<0.25 ha, $0.25-<0.5$ ha, $0.5-<1$ ha, $1-<2$ ha, $2-<4$ ha, $4-<8$ ha, $8-<16$ ha, $16-<32$ ha, and >32 ha); three parks were randomly selected from the smallest six categories, and two were selected from each of the others. The location of park boundaries was primarily based on information in the GIS layer, but after ground-truthing several were modified to improve accuracy. ArcMap 9.2 (ESRI 2006) was used for all GIS analyses.

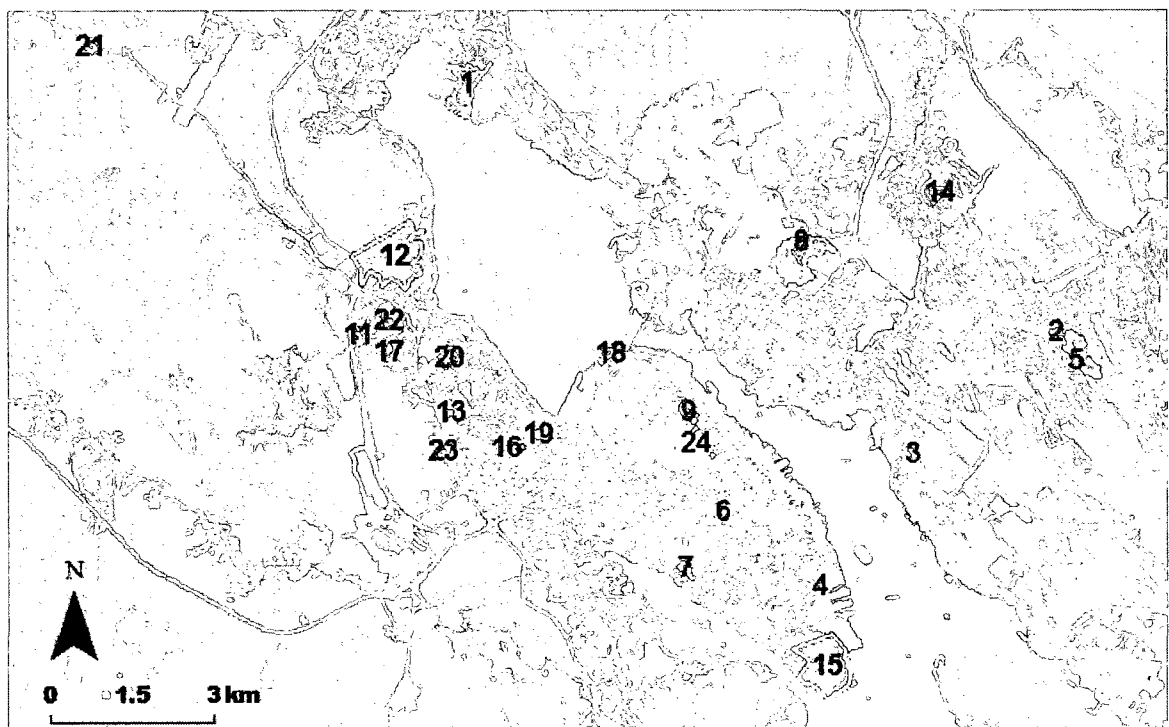


Figure 1: Locations of the 24 urban parks that were studied. Park names are provided in Table 1. Landsat image from CCRS (1989).

Table 1: Area (ha) of semi-natural and anthropogenic habitats within the urban parks that were studied.

Number	Park	Semi-natural forest	Anthropogenic habitat	Total area
1	Admiral's Cove Park	28.5	0.00	28.5
2	Alder Piper Park	0.55	0.49	1.04
3	Arnold Whitworth Park	0.26	0.27	0.53
4	Barrington Street Park	0.00	0.08	0.08
5	Bell Lake Park	10.0	0.49	10.5
6	Cogswell Park	0.00	0.29	0.29
7	Conrose Park	0.00	2.66	2.66
8	Cyril Smith Park	24.5	0.00	24.5
9	Fort Needham Memorial Park	0.84	4.70	5.54
10	Fuller Terrace Park	0.00	0.07	0.07
11	Glenbourne Park	1.46	2.70	4.16
12	Hemlock Ravine	91.4	0.20	91.6
13	Lincoln Cross Park	0.18	0.09	0.27
14	Montebello Park	0.00	2.31	2.31
15	Point Pleasant Park	65.9	10.1	76.1
16	Randall Avenue Park	0.21	0.19	0.39
17	Remington Court Park	0.56	0.24	0.80
18	Seaview Memorial Park	0.00	5.06	5.06
19	Titus Smith Park	0.00	0.90	0.90
20	Tremont Plateau Park	4.93	3.3	8.24
21	Uplands Park	0.00	1.77	1.77
22	Wedgewood Park	1.83	0.10	1.93
23	Willett Street Park	2.97	0.12	3.09
24	Young/Kaye Park	0.00	0.14	0.14

Sampling Design

Sampling plots of 10 m x 10 m were randomly distributed within *semi-natural* and *anthropogenic* habitats of the selected urban parks. *Anthropogenic* habitats were identified as those which were converted to an anthropogenic land-use and/or were subject to frequent management activities that disrupted natural successional processes, such as mowing. Conversely, *semi-natural* habitat was loosely defined as those whose natural regeneration processes are more-or-less unhindered by anthropogenic management. Within each of the selected parks, the *semi-natural* and *anthropogenic* habitats were mapped on a GIS layer using information available from air photos, park

maps, and site visits. Within each park, the locations of survey plots were then identified using GIS to generate random coordinates, with replication being determined by the size of the respective habitats (Table 2). In order to reduce bias associated with changes in percent cover estimates throughout the growing season, the temporal sequence of sampling days was randomly determined.

Table 2: Number of plots per size increment within the general habitat-types.

Size (ha)	Semi-natural habitats	Anthropogenic habitats
< 0.25	2	1
0.25 - 0.5	4	2
0.5 - 1	6	3
1 - 2	8	3
2 - 4	10	4
4 - 8	12	4
8 - 16	14	5
16 - 32	16	5
> 32	18	6

Field Methods

Plot Establishment

A GPS was used in the field to identify the center of the plots (within an accuracy of 5 m). If the randomly derived plot coordinates resulted in any part of the plot crossing an ecotone associated with a forest boundary or trail, the plot was re-established to a distance where this was not a concern (typically 1-4 m) at an angle perpendicular to the identified ecotone. The sides of the square plots were positioned so that they were parallel and perpendicular to the slope of the terrain. In addition to the randomly located plots, four others were placed within stands of forest that had regenerated following a previous conversion to an anthropogenic land-use type. Three of these were within forest habitat described as containing “old field patches” in Point Pleasant Park (Neily et al. 2004) - their locations were haphazardly selected from a map prior to site visitation. Another plot, within Hemlock Ravine Park, was positioned at a location known to have been previously

occupied by a small shed (as determined by a conversation with local residents during summer 2007).

Data Collected

Field work was performed during the summer of 2007. Vegetation composition and structure was quantified within each of the study plots² (n = 214). Trees >10 m in height were identified to species and their cover was estimated using an ocular tube (radius 8.5 cm and length 14.5 cm) at five locations within each plot (center and 0.5 m inside each corner). All other vascular plants rooted within the plot and <10 m in height were identified to species and assigned a value based on a visual estimation of their foliage cover. Separate estimates were made for woody species within the “shrub” (<2 m) and “canopy” (2-10 m) layers. Cover values >10% were estimated to the nearest 1% but those <10%, <1%, and <0.1% were recorded to the nearest 0.5%, 0.1%, and 0.01% increment, respectively. A clinometer was used to identify the 10 m height mark. Non-vascular taxa with a percent cover >0.01 growing on the ground were identified to genus and assigned a cover value. Estimates of total non-vascular, herbaceous, shrub, 2-10 m canopy, and >10 m canopy cover were made. The species and DBH of all live trees and snags within plots having a diameter >5 cm were also recorded. In addition, plot borders were used as transects along which the diameter of all intersecting pieces of coarse woody debris (CWD) >4 cm in diameter were recorded.

Vascular plants were identified using *Roland's Flora of Nova Scotia* (Zinck 1998), *Flora of New Brunswick* (Hinds 2000), *Manual of Vascular Plants of the Northeastern United States and Adjacent Canada* (Gleason and Cronquist 1991), *Manual of Cultivated Plants* (Bailey 1973) and the *Flora of North America: North of Mexico* series (Flora of North America Editorial Committee 1993+). Taxonomic nomenclature follows Kartesz (1999), although several ecologically / morphologically distinct taxa not recognized by this reference were maintained (e.g., *Deschampsia cespitosa* ssp. *parviflora*). Bryophyte nomenclature follows the *Moss Flora of the Maritime*

² In order to facilitate any future comparison with other data sets, field protocols were developed with reference to those used to conduct vegetation surveys by the Nova Scotia Department of Natural Resources during the development of its Forest Ecosystem Classification (Neily 2006).

Provinces (Ireland et al. 1982), while that for lichens adheres to the *Lichens of North America* (Brodo et al. 2001).

Information on recent disturbance, fragmentation, and past and present anthropogenic influences was recorded for each plot. All plots were assigned a disturbance class based on a visual inspection of the effects of hurricane Juan. Low, moderate, and high disturbance designations were assigned based on the proportion of dominant trees that had been severely damaged by the hurricane, with <25%, 25-75%, and >75% being used as the respective criteria. The presence and character of on-going management activities and historical human influences, as evidenced by the vegetation, substrate, and remnants of built structures, were recorded. In addition, distance and aspect to the nearest anthropogenically maintained trail and forest boundary (within 50 m of the plot center), as well as trail width, were recorded for plots within semi-natural habitats. For plots >50 m away from such trails and forest boundaries, the relevant measures were made using GIS techniques after field work. Park reports, maps, and orthophotos were used in conjunction with field notes on evidence of historical influences to identify/confirm forested sites that had been converted to an anthropogenic land-use type (including fields, building sites, and other areas cleared for human use) within the 20th century.

Habitat Types

Information on management, recreation, disturbance, and site history was used to classify plots into one of seven “habitat types”. Semi-natural habitats were categorized as either *tertiary* forest or *low*, *moderately*, or *highly* disturbed primary/secondary³ forest. *Tertiary* forest is defined here as stands that have regenerated from a conversion⁴ to an anthropogenic land-use, as evidenced by site visits and/or historical records. *Low*, *moderately*, and *highly* disturbed primary/secondary forest designations were based on previously described hurricane disturbance scores (low, moderate, or high disturbance). Conversely, plots within anthropogenic habitats were assigned to one of three classes

³ Secondary forests are those which are regenerating from a major stand-replacing disturbance (e.g., fire, windthrow, or timber harvest) whereas primary ones have reached an “old growth” condition. As far as known, the criteria for reference to “tertiary” forests in this document are unique.

⁴ The exact nature and timing of the conversions is unknown but records indicate that they either took place within, or were maintained into, the 20th century.

according to the intensity of management and recreation to which they were subject: *derelict*, *horticultural*, or *intensive recreation*. *Horticultural* sites were characterized by understory vegetation maintained through horticultural practices (mowing, planted flower beds, etc.) and provide passive recreational services to city residents. *Intensive recreation* habitats are similar to *horticultural* ones but are managed specifically for active recreational activities (i.e. sports fields). *Derelict lands* were identified as habitats whose vegetation was not evidently subject to frequent management practices, but that are at an early successional stage as a result of a recent abandonment from an anthropogenic land-use type. The habitat types are considered to be subject to varying degrees of anthropogenic stress, as is depicted when they are assigned qualitative scores representing their exposure to human activities (Table 3).

Table 3: Variation in selected anthropogenic stressors among habitat types (0 = negligible, 1 = low, 2 = moderate, 3 = high).

Habitat type	Intensity of present management activities	Recency of land-use conversion / maintenance	Intensity of present recreational activities	Total score
Low-disturbance forest	1	0	1	2
Moderately-disturbed forest	1	0	1	2
Highly-disturbed forest	1	0	1	2
Tertiary forest	1	1	1	3
Derelict land	1	2	1	4
Horticultural	3	3	2	8
Intensive recreation	3	3	3	9

Plant Functional Groups

Information on the growth form, life history, and biogeographical status of species was combined to identify plant functional groups (PFGs). These plant characteristics were expected to form PFGs that are sensitive to the measured environmental stressors (management, fragmentation, and disturbance). Growth form was regarded as an important plant characteristic associated with horticultural activities (i.e., it was expected that certain growth forms would be preferred more than others). Life history strategy was

considered to respond to variations in the intensities of both natural and anthropogenic disturbance mechanisms. Furthermore, species of differing biogeographical status were considered to respond in opposing ways to stressors associated with fragmentation, management, and disturbance (both natural and anthropogenic).

A designation of “exotic” or “native” was assigned to vascular species to represent their biogeographical status. Exotic taxa were defined as those whose presence is a result of an accidental or intentional introduction beyond their natural range by human activities (Richardson et al. 2000b). In this context, all species introduced to Nova Scotia were considered exotic. In addition, species with native populations in the province were considered exotic if they were restricted to regions and habitats not surveyed in the study (i.e. extralimitals) and if introduced genotypes are known to be common elsewhere (e.g., *Poa pratensis* and *Thuja occidentalis*). This treatment is consistent with other definitions used in a park management context (see Westman 1990). For species with indigenous and non-indigenous populations occupying similar habitats, biogeographical status was assigned based on their relative commonness (e.g. *Achillea millefolium* considered native whereas *Prunella vulgaris* was considered exotic). References used to determine the biogeographical status of taxa include the regional floras previously listed for plant identification purposes as well as the S-ranks of the Atlantic Canada Conservation Data Center (ACCDC 2007).

Life history and growth form were assigned based on information in Gleason and Cronquist (1991) and the USDA online plant database (USDA 2008). Growth form categories include: lichen, bryophyte, graminoid, forb, pteridophyte, woody vine, subshrub, shrub, and tree. Life-history status was assigned to vascular plants only and included annual, biennial, and perennial designations.

Analyses

Univariate and multivariate techniques were used to determine relationships among management regimes, historical influences, fragmentation-related factors, and variation in vegetation composition and structure. Univariate analyses were performed with SPSS 15.0 (SPSS Inc. 2006) and Microsoft Excel (Microsoft Corporation 2002)

whereas PC-ORD 4.41 (McCune and Mefford 1999) was used for the multivariate techniques.

Measures of vegetation structure and diversity within each of the habitat types were summarized by computing their median and interquartile range (IQR) values (these statistics were used to present the data because of problems with normality). Structural measures included non-vascular ground cover, herbaceous cover, shrub (<2 m) cover, 2-10 m canopy cover, >10 m canopy cover, basal area of live trees (m²/ha), basal area of snags (m²/ha), CWD (m²/ha), live tree density (stems/ha), and snag density (stems/ha). Measures of diversity included species richness (the number of taxa in a plot) and Shannon's index (H) computed as:

$$H = -\sum_{i=1}^n (p_i * \ln(p_i))$$

where p_i is the cover of species i relative to the total cover of all species. Kruskal-Wallis tests were applied to identify variables that differed significantly amongst management groups. These were followed by multiple comparisons amongst specific habitat types using Mann-Whitney U-tests with Bonferroni correction.

Variation in vegetation composition among habitat types was examined using the Indicator Species Analysis (INSPAN) of Dufrene and Legendre (1997). This method was used to compute indicator values (IVs) for individual taxa and PFGs for the *semi-natural* and *anthropogenic* habitat designations as well as the more specific habitat types. IVs convey the degree to which a species or PFG has an affinity for a particular grouping, relative to the others (higher IVs imply greater association). The analyses were based on plot - species / PFG matrices. Single cover values for each species or PFG were obtained by summing cover estimates within all strata (the >10 m strata was incorporated using the average of the five cover estimates for each species). IVs are calculated using a combination of relative abundance and relative frequency:

$$IV = A_{ij} * B_{ij} * 100$$

where A_{ij} is the mean cover of species or PFG i in the sites of group j compared to all groups in the study ($A_{ij} = \text{cover}_{ij} / \text{cover}_i$) and B_{ij} is the relative frequency of occurrence of species or PFG i in the sites of group j ($B_{ij} = \text{Nsites}_{ij} / \text{Nsites}_j$). The IV values may be interpreted to identify “indicators” which are the “most characteristic species (or PFG) of

each group, found mostly in a single group of the typology and present in the majority of the sites belonging to that group”⁵(Dufrene and Legendre 1997). Monte Carlo permutations (1000) were performed to test for significant relationships between IVs and their habitat types. “Important” indicators were identified as those with an IV > 25% and a highly significant association ($p < 0.01$) to a particular habitat type.

A Detrended Correspondence Analysis (DCA) was performed in order to obtain a visual interpretation of variation in species composition amongst the habitat types. This analysis was based on the same species-site matrix as the INSPAN analysis. DCA, a unimodal method of ordination, was used because the lengths of the ordination axes were greater than 3 standard deviations (Jongman et al. 1995). The DCA was performed on square-root transformed species cover values in order to reduce the influence of dominant taxa on the analysis. Additionally, cover values of rare species (those with a frequency < 1/5 of the commonest species) were down-weighted in proportion to their frequency. Axes were rescaled using 26 segments and a rescaling threshold of 0. The strength of the DCA axes was measured using an after-the-fact coefficient of determination between relative Euclidean distance in the unreduced species space and Euclidean distance in the ordination space (as recommended by PC-ORD).

Indirect gradient analysis was used to examine the influence of fragmentation-related factors on vegetation. To do this, a DCA was performed on the understory composition (<2 m in height) of semi-natural forest plots ($n = 151$). Understory composition was used because it was expected to provide a more sensitive measure of vegetation response than an analysis that included the overstory. This DCA was performed with the same settings as outlined for the previous ordination, and was found to be a statistically suitable method for presenting variation in species composition among plots (i.e., unimodal method is appropriate). Spearman rank-correlation coefficients (ρ) were calculated among the first three DCA axes, disturbance classes (an ordinal variable), historical influences (dummy variable), and the fragmentation-related factors (distance and aspect to trails and boundaries, trail width). Because edge influences within the

⁵ This type of indicator is fundamentally different than those used for assessing EI. However, indicators of particular environmental states, such as are provided here, may be used to relay information on changes in ecological conditions, and in this sense they may be used indirectly as EI indicators.

region are more pronounced for forests that face south than north (Harper et al. 2005), aspect measurements were transformed to represent degrees departure from south.

Canonical Correspondence Analysis (CCA) was used to examine relationships among understory vegetation composition within semi-natural forests and important fragmentation-related factors. The CCA was performed using a PFG-plot matrix. Plot scores along the CCA axes were constrained to be linear combinations of the environmental variables identified as highly significant ($p < 0.01$) along either of the first two DCA axes. The distance measurements were log-transformed because trails and boundaries were expected to have a more pronounced influence on vegetation composition when in close proximity (following rationale outlined in Jongman et al. 1995). Disturbance and historical influences were incorporated into the model in order to provide context. Monte Carlo permutations (1000) were used to test the strength between the two matrices. Because the inclusion of moderately to strongly intercorrelated variables may yield unstable CCA results (Jongman et al. 1995), the variables employed were tested first for multicollinearity using Spearman rank coefficients, but were found to be only weakly correlated (maximum $\rho = 0.27$). Standardized canonical coefficients and intraset correlations were calculated between the environmental variables and the CCA axes.

Prior to the multivariate analyses, outliers were identified using the “Outlier Analysis” in PC-ORD. Outlier plots were identified as those with a Euclidean distance measure > 2 s.d. away from the mean. Thirteen and seven outliers were identified for the all-plot and semi-natural-plot matrices, respectively (based on species), and DCAs were performed on data sets with and without them. In the end, outliers were not removed because doing so did not make the results more interpretable. Furthermore, outliers did not appear to reflect measurement error.

Results

Vegetation Structure

All structural variables varied significantly among habitat types (Table 4). In general, all measures except herbaceous cover were higher within semi-natural habitats

than anthropogenic ones, suggesting that management activities result in a simplification of vegetation structure. Low structural variability was particularly pronounced for derelict lands and sites managed for intensive recreation. Some structural attributes for horticultural habitats were highly variable. Although median values were low, IQR values suggest that canopy cover (2-10 m and >10 m), live tree basal area, and tree density within horticultural plots were sometimes similar to that of moderately-to-highly disturbed forest remnants. Conversely, shrub cover, snag basal area, snag density, and CWD were consistently impoverished in all anthropogenic habitats compared to semi-natural ones.

Differences in structure were observed among semi-natural habitats. Measures of overstory structure (canopy cover, basal area, density of live trees) and non-vascular cover showed a general decrease with disturbance intensity, whereas CWD, herb cover, and shrub cover increased. Snag density decreased with higher disturbance whereas snag basal area increased, indicating that snags in disturbed sites are generally larger (which may, in part, reflect management efforts to conserve large snags for their benefits to wildlife). Some structural attributes of tertiary forests varied considerably from other semi-natural habitats (Table 4). In particular, snag density and basal area were low, measures of overstory structure (including canopy cover >10m, basal area, and tree density) were more similar to moderately disturbed forests, and their understory (cover of non-vascular, herbaceous, and shrub layers) is more like highly disturbed ones despite tertiary forests being relatively un-disturbed (10 were classified as “low disturbance” and 2 as “moderately-disturbed”).

Vegetation Composition

General Observations of Natural History

Within the 2007 study plots, a total of 367 vascular taxa were recorded (327 of which were identified to their species epithet), as well as 28 bryophyte and 2 lichen genera. Several rare-to-uncommon vascular species⁶ were located within or at the edge of semi-natural forest remnants, including *Carex foenea* (dry-spike sedge), *Carex swanii*

⁶ Specimens of rare or uncommon taxa have been submitted to one of the following herbaria: Dalhousie University, Nova Scotia Museum of Natural History, or Acadia University.

(swan sedge), *Hieracium paniculatum* (panicled hawkweed), and *Viola sagittata* (arrow-leaved violet). Of these, *C. swanii* may be considered the most provincially-uncommon - it has been assigned a rank of “S2?” by the ACCDC whereas the others are considered S3 or S3/S4 (the “?” reflects the uncertainty of the rank, however) (ACCDC 2007). *C. swanii* was recorded within three parks during the present study - Wedgewood, Alder Piper and Bell Lake. In addition, the rare moss *Tetraplodon angustatus* (S1), which is associated with dung of carnivores (Ireland et al. 1982), was encountered in Willett St. Park during the summer of 2006.

Of the vascular taxa, 157 are exotic (at least in part). Several of the exotics, although known to reside in the province, had not been previously reported within the Halifax area, including *Erysimum hieracifolium* (European wallflower) and *Epipactis helleborine* (helleborine). In particular, *E. hieracifolium* had previously only been collected once in Nova Scotia, on gypsum at Heatherdale, Cape Breton (Zinck 1998). This species was encountered in tertiary forest at the edge of an old battlement in Point Pleasant Park.

Table 4: Median (IQR) for measures of vegetation structure and diversity and P-values from Kruskal–Wallace tests of significance amongst habitat types. Habitat types with a shared letter (e.g., a) are not significantly different for the associated attribute, as identified by pairwise Mann-Whitney multiple comparison tests (experiment-wise alpha <0.05). Codes for habitat types are as follows: LD – low-disturbance forest; MD – moderately-disturbed forest; HD – highly-disturbed forest; Tert. – tertiary forest; Der. – derelict lands; Hort. – horticultural; and Rec. – intensive recreation.

	Semi-natural forest				Anthropogenic habitats			
	LD (n=100)	MD (n=22)	HD (n=17)	Tert. (n=12)	Der. (n=9)	Hort. (n=45)	Rec. (n=9)	p
STRUCTURAL								
Non-vascular cover	7.0 (21) ab	10 (13) a	2.0 (3.8) c	2.0 (5.5) bc	0.00 (1.5) cd	0.20 (2.5) cd	0.00 (0.01) d	<0.01
Herbaceous cover	20 (32) d	20 (31) cd	70 (55) b	45 (50) bc	98 (28) ab	99 (7.0) a	100 (1.3) a	<0.01
Shrub cover (<2m)	35 (45) a	45 (51) a	40 (48) a	30 (37) ab	7.0 (13) bc	1.0 (8.0) cd	0.00 (0.00) d	<0.01
Canopy cover (2-10m)	65 (34) a	35 (30) b	10 (11) c	75 (36) a	0.01 (7.5) cd	10 (40) bc	0.00 (0.00) d	<0.01
Canopy cover (>10m)	50 (41) a	31 (17) b	0.00 (11) d	24 (22) abc	0.00 (0.00) d	6.0 (23) cd	0.00 (0.00) d	<0.01
Live tree basal area (m ² /ha)	28 (15) a	21 (13) a	0.86 (5.3) b	18 (14) a	0.00 (0.10) b	0.00 (18) b	0.00 (0.00) b	<0.01
Snag basal area (m ² /ha)	2.2 (4.5) a	2.4 (6.1) ab	0.00 (14) abc	0.00 (1.2) bd	0.00 (0.00) cd	0.00 (0.00) d	0.00 (0.00) cd	<0.01
CWD (m ² /ha)	1.8 (4.8) b	7.9 (12) a	18 (18) a	0.66 (1.8) bc	0.00 (0.00) cd	0.00 (0.00) d	0.00 (0.00) cd	<0.01
Tree density / ha	2200 (1500) a	1100 (630) b	100 (650) cd	750 (2200) abc	0.00 (50) de	0.00 (250) de	0.00 (0.00) e	<0.01
Snag density / ha	250 (500) a	200 (230) ab	0.00 (200) bc	0.00 (75) bd	0.00 (0.00) cd	0.00 (0.00) d	0.00 (0.00) cd	<0.01
COMPOSITIONAL								
Species richness	24 (15) ab	26 (14) ab	29 (11) ab	32 (15) a	27 (11) ab	20 (14) b	9.0 (5.0) c	<0.01
Diversity (Shannon's index)	2.0 (0.57) a	2.0 (0.36) a	2.4 (0.85) a	2.2 (0.38) a	2.6 (0.71) a	1.8 (0.47) a	1.2 (0.39) b	<0.01

Management influences

The distribution of plots within multidimensional space, as presented by the DCA, allows an interpretation of relationships among vegetation composition, disturbance, and management regimes. The first axis (eigenvalue 0.843, 55.8% of variance) arranged sites along a general gradient of anthropogenic influence, as identified by the relative position of the habitat types (Figure 2). Arranged in order of increasing human influence were semi-natural forests subject to varying intensities of disturbance, tertiary forests, derelict lands, horticultural sites, and areas managed for intensive recreation (sports fields). The second axis (eigenvalue 0.293, 6.5% of variance) of the DCA (Figure 2) is interpreted to represent, a gradient of natural disturbance as is indicated by the separation of semi-natural forest plots according to their disturbance class. Anthropogenic plots exhibited little variation along the second axis, suggesting that the influence of management practices overwhelms that of natural processes, thereby promoting biotic homogenization. The third axis (not shown) had an eigenvalue of 0.192 and accounted for 4.2% of the variance.

Species richness and diversity varied among the habitat types (Table 4). Sites managed for intensive recreation had significantly lower species richness and diversity than any other habitat type. Species richness was greatest in tertiary forest, whereas diversity was comparably high within derelict lands, highly disturbed, and tertiary forest. The richness and diversity of horticultural sites was similar to that of low-to-moderately disturbed forests. Derelict lands and tertiary forests are comprised of high abundances of both exotic and native species, whereas other habitat types were dominated more intensely by one or the other (Table 5).

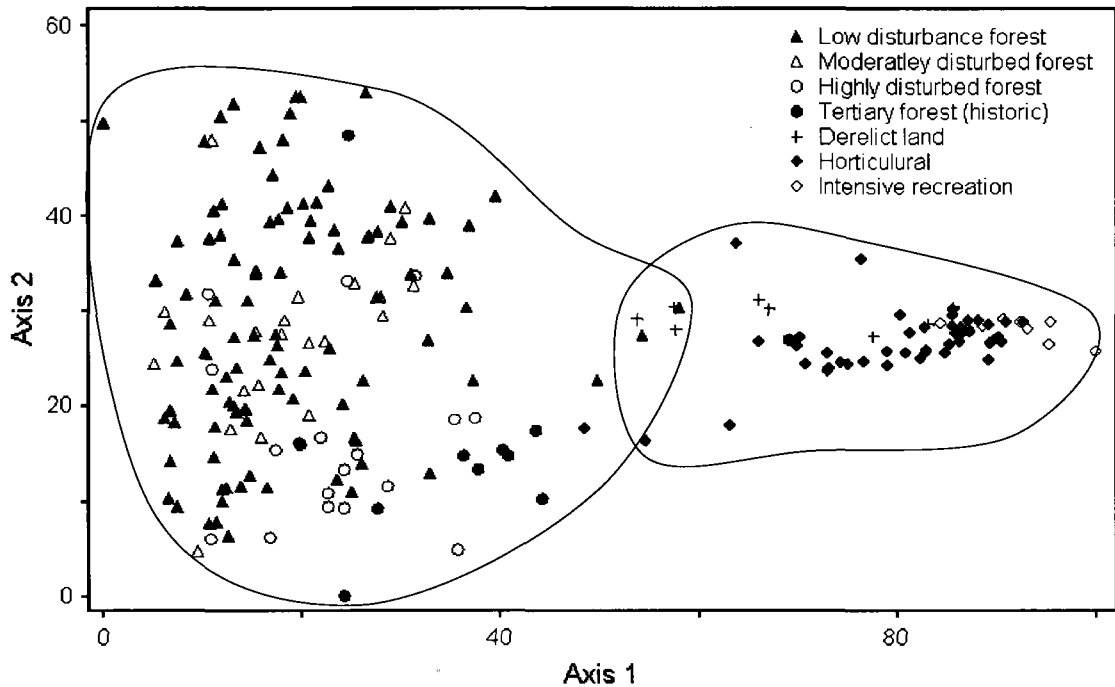


Figure 2: DCA of study plots (n = 214) showing variation in species composition (overstory + understory) among habitat types. Lines are drawn around semi-natural and anthropogenic plots.

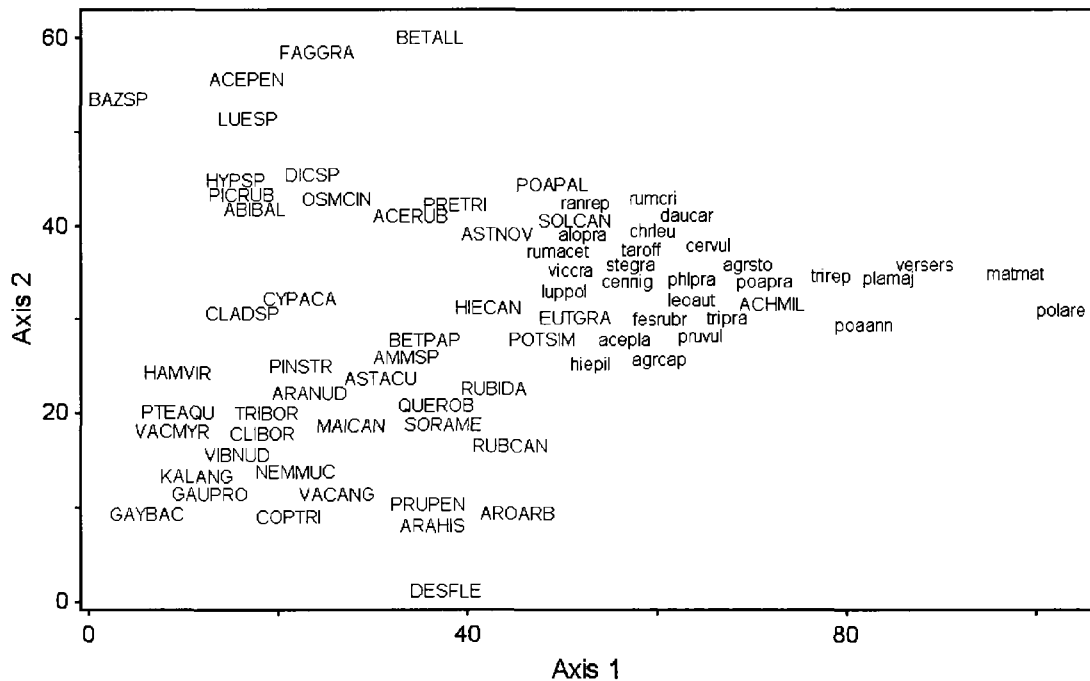


Figure 3: DCA of 75 Indicator taxa (IV > 25, p < 0.01). Codes based on the first three letters of genus and species epithet; exotic species are in lowercase; complete species names are provided in Table 7.

Table 5: Percent contribution of exotics to species richness and structural variables (values not provided for measures of snags and CWD due to challenges in some cases in identifying the species). Codes for management-types are as follows: LD – low-disturbance forest; MD – moderately-disturbed forest; HD – highly-disturbed forest; Tert. – tertiary forest; Der. – derelict lands; Hort. – horticultural; and Rec. – intensive recreation.

	Semi-natural forest				Anthropogenic habitats		
	LD	MD	HD	Tert.	Der.	Hort.	Rec.
Herbaceous cover	6.7	13	0.9	32.3	78.5	95.5	99.9
Shrub cover (<2m)	2.5	0.1	0.5	23.2	29.3	78.3	-
Canopy cover (2-10m)	1.6	0.2	2.1	42.3	17.5	77.4	-
Canopy cover (>10m)	0.4	1.6	0.0	45.1	-	71.1	-
Live tree basal area (m ² /ha)	1.9	0.6	0.1	51.0	0.0	58.8	-
Tree density / ha	1.6	0.3	0.2	39.6	0.1	56.6	-
Species richness	8.9	8.9	4.5	28.2	66.4	82.8	96.4

A total of 20 PFGs were constructed from the data, the majority of which were useful indicators of management regimes (Table 6). In addition, 75 indicator species were identified, 21 and 33 of which were “important” indicators (IV>25% and $p<0.01$) of semi-natural and anthropogenic habitat-types, in general (Table 7). Semi-natural forests were primarily comprised of native vascular and non-vascular taxa. In order of their corresponding indicator values, native shrubs, trees, bryophytes, pteridophytes, perennial forbs, perennial graminoids, lichens, and subshrubs were associated with these habitats. In contrast, exotic vascular taxa dominated the anthropogenic habitats and bryophyte and lichen cover were low. PFGs significantly associated with anthropogenic habitats ($p<0.01$), include exotic annual graminoids, perennial forbs, annual forbs, trees, and annual graminoids. Although no alien PFGs were significantly associated with the semi-natural forests, exotic climbers were more prominent within these habitats than anthropogenic ones.

Vegetation composition varied considerably amongst the types of semi-natural habitats. The prominence of native trees, shrubs, bryophytes, pteridophytes, and annual forbs decreased with disturbance intensity, whereas perennial forbs, perennial graminoids, and subshrubs obtained their greatest IV values in highly-disturbed forests (Table 6).

Only one species, *Acer rubrum*, was found to be an important indicator of low-disturbance forests (*A. rubrum* is also common in regenerating forests, however, and as such, its IV value here likely represents greater cover (abundance) in low-disturbance forests). In contrast, 6 and 12 species were identified as important indicators of moderately and highly-disturbed sites, respectively. Indicators of moderately disturbed forests included native trees such as *Abies balsamea*, bryophytes such as *Hypnum* sp., and *Cladonia* lichens. Within highly-disturbed forests, a number of native forbs such as *Aralia hispida* were identified as indicators as well as native trees (e.g., *Prunus pensylvanica*), subshrubs (e.g. *Rubus idaeus*), shrubs (e.g. *Gaultheria procumbens*), and a grass (*Deschampsia flexuosa*). Variation in the number of indicators identified for the disturbance classes likely reflects heterogeneity in the species composition of intact forests and the homogenizing influence of disturbance processes on them.

Exotic taxa were of little prominence in semi-natural habitats, except within tertiary forests where they accounted for 23-51% of vegetation structure and 28% of species richness (Table 5). Of the PFGs, exotic trees and native perennial graminoids were significantly associated to tertiary forests, where they received IV values of 53% and 42%, respectively. The exotic tree *Acer platanoides*, several native shrubs (e.g., *Photinia pyrifolia*), and the native perennial forb *Hieracium canadense* were identified as indicators of tertiary forests.

Important differences in vegetation composition among the anthropogenic habitat types are also evident. Although common in all anthropogenic habitats, exotic forbs were particularly prominent in derelict lands. Exotic trees and perennial graminoids were highest in horticultural sites, and annual graminoids were greatest in sites managed for intensive recreation. Among the habitat types, 16 species were associated with derelict lands, 8 with horticultural habitats, and 5 with recreational ones. Although most of these were exotic forbs and graminoids, several native indicators were also identified. Examples of exotic indicators include *Centaurea nigra* and *Phleum pratense* within derelict lands; *Festuca rubra* ssp. *rubra* and *Leontodon autumnalis* within horticultural sites; and *Matricaria discoidea* and *Poa pratensis* within recreational ones. Although several native indicator herbs, such as *Poa palustris* and *Stellaria graminea*, were

identified for derelict lands, only one (*Achillea millefolium*) was identified for horticultural habitats and recreational sites had none.

Table 6: Percent indicator values for PFGs amongst habitat types. Codes for habitat types are as follows: Ftot. – semi-natural forests (in general); LD – low-disturbance forest; MD – moderately-disturbed forest; HD – highly-disturbed forest; Tert. – tertiary forest; Atot. – anthropogenic habitats (in general); Der. – derelict lands; Hort. – horticultural; and Rec. – intensive recreation.

PFGs	Semi-natural forest					Anthropogenic habitats			
	Ftot.	LD	MD	HD	Tert.	Atot.	Der.	Hort.	Rec.
NATIVE VASCULAR PLANTS									
Forbs									
Annuals	20*	11	1	0	11	0	2	0	0
Biennials	1	0	1	2	2	4	11	0	0
Perennials	63**	8	9	33**	18	24	27	3	0
Graminoids									
Annuals	0	0	0	0	0	2	0	2	0
Perennials	55**	4	4	37	42**	7	2	1	0
Pteridophytes									
Annuals	73**	29*	15	15	3	1	5	0	0
Subshrubs	30**	2	4	46**	6	3	9	0	0
Shrubs	94**	31**	21	13	28	0	0	0	0
Trees	93**	36**	27	13	18	2	1	1	0
EXOTIC VASCULAR PLANTS									
Forbs									
Annuals	1	0	0	0	0	59**	32**	13	25
Biennials	3	0	0	0	6	11	29**	1	0
Perennials	2	0	0	0	9	95**	35**	31	21
Graminoids									
Annuals	0	0	0	0	1	34**	1	6	26*
Perennials	1	0	0	0	2	98**	29	36**	31
Subshrubs	0	0	0	0	0	2	0	2	0
Climbers	13	8	1	0	6	9	14	3	0
Shrubs	1	1	0	0	0	14*	9	9	0
Trees	5	0	0	0	53**	42**	0	27	0
NON-VASCULAR PLANTS									
Bryophytes	78**	36**	31	8	10	10	1	6	0
Lichens	50**	8	45**	5	0	0	1	0	0

** p<0.01, * p<0.05

Table 7: Percent indicator values for species among habitat types (for those with $p < 0.01$ and $IV > 25\%$). The first letter of the “Plant Functional Group” designates biogeographical status: native (N) or exotic (E); second letter identifies life history: annual (A), biennial (B), or perennial (P). Note that biogeographical status is only provided for vascular taxa and life history is only given for herbaceous taxa (if not provided, assume native and perennial status). Codes for habitat types are as follows: Ftot. – semi-natural forests (in general); LD – low-disturbance forest; MD – moderately-disturbed forest; HD – highly-disturbed forest; Tert. – tertiary forest; Atot. – anthropogenic habitats (in general); Der. – derelict lands; Hort. – horticultural; and Rec. – intensive recreation. Species are arranged to reflect a decreasing association within semi-natural habitats and an increasing association with anthropogenic ones (in general), as determined by the IV values.

Spp CODE	Spp name	Plant Functional Group	Semi-natural forest					Anthropogenic			
			Ftot.	LD	MD	HD	Tert.	Atot.	Der.	Hort	Rec.
ACERUB	<i>Acer rubrum</i>	NTree	90**	32**	31	16	10	1	1	0	0
DICSP	<i>Dicranum sp.</i>	Bryophyte	83**	28	32**	15	7	0	0	0	0
PICRUB	<i>Picea rubens</i>	NTree	81**	27	35**	14	5	0	0	0	0
HYPSP	<i>Hypnum sp.</i>	Bryophyte	80**	33	37**	7	5	0	0	0	0
AMMSP	<i>Amelanchier sp.</i>	NShrub	79**	24	21	7	28**	0	0	0	0
VIBNUD	<i>Viburnum nudum</i>	NShrub	73**	20	15	24*	13	0	0	0	0
BETPAP	<i>Betula papyrifera</i>	NTree	70**	19	20	23*	11	1	0	1	0
MAICAN	<i>Maianthemum canadense</i>	NPForb	65**	13	8	29**	17	0	0	0	0
LUESP	<i>Leucobryum sp.</i>	Bryophyte	56**	23	36**	3	2	0	0	0	0
TRIBOR	<i>Trientalis borealis</i>	NPForb	54**	11	13	17	11	0	0	0	0
KALANG	<i>Kalmia angustifolia</i>	NShrub	51**	20	14	5	13	0	0	0	0
PTEAQU	<i>Pteridium aquilinum</i>	NPteridophyte	51**	16	17	16	3	0	0	0	0
CLADSP	<i>Cladonia sp.</i>	Lichen	51**	13	30**	12	3	0	0	0	0
VACMYR	<i>Vaccinium myrtilloides</i>	NShrub	50**	16	23*	12	2	0	0	0	0
GAUPRO	<i>Gaultheria procumbens</i>	NShrub	50**	11	14	26**	4	0	0	0	0
VACANG	<i>Vaccinium angustifolium</i>	NShrub	49**	9	15	7	29**	0	0	0	0

Spp CODE	Spp name	Plant Functional Group	Semi-natural forest					Anthropogenic			
			Ftot.	LD	MD	HD	Tert.	Atot.	Der.	Hort	Rec.
ABIBAL	<i>Abies balsamea</i>	NTree	47**	11	38**	12	0	0	0	0	0
QUEROB	<i>Quercus rubra</i>	NTree	47**	20*	12	7	10	1	0	1	0
ARANUD	<i>Aralia nudicaulis</i>	NPForb	45**	8	11	25*	6	0	0	0	0
FAGGRA	<i>Fagus grandifolia</i>	NTree	44**	26*	23	0	0	0	0	0	0
SORAME	<i>Sorbus americana</i>	NTree	41**	5	3	31**	22	1	1	0	0
PINSTR	<i>Pinus strobus</i>	NTree	37**	12	5	10	11	0	0	0	0
PRETRI	<i>Prenanthes trifoliata</i>	NPForb	37**	6	7	24*	7	1	0	0	0
NEMMUC	<i>Nemopanthus mucronata</i>	NShrub	34**	12	4	11	7	0	0	0	0
CYPACA	<i>Cypripedium acaule</i>	NPForb	32**	7	7	2	19	0	0	0	0
ASTACU	<i>Oclemena acuminata</i>	NPForb	32**	7	1	32**	6	0	0	0	0
CLIBOR	<i>Clintonia borealis</i>	NPForb	32**	3	10	32**	2	0	0	0	0
GAYBAC	<i>Gaylussacia baccata</i>	NShrub	30**	11	12	1	9	0	0	0	0
OSMCIN	<i>Osmunda cinnamomea</i>	NPteridophyte	29**	17	5	4	1	0	0	0	0
HIECAN	<i>Hieracium canadense</i>	NPForb	29	2	6	8	38**	8	5	1	0
BAZSP	<i>Bazzania sp.</i>	Bryophyte	28**	17	15	0	1	0	0	0	0
HAMVIR	<i>Hamamelis virginiana</i>	NShrub	27**	16	11	1	0	0	0	0	0
BETALL	<i>Betula alleghaniensis</i>	NTree	27**	21*	2	0	2	1	1	0	0
ACEPEN	<i>Acer pensylvanicum</i>	NTree	26**	11	18*	1	0	0	0	0	0
PRUPEN	<i>Prunus pensylvanica</i>	NTree	19**	0	5	63**	0	0	0	0	0
DESFLE	<i>Deschampsia flexuosa</i>	NPGraminoid	17**	0	1	25**	24	0	0	0	0
RUBIDA	<i>Rubus idaeus ssp. strigosus</i>	NSubshrub	16*	1	3	43**	1	1	4	0	0
RUBCAN	<i>Rubus canadensis</i>	NSubshrub	15*	1	4	28**	1	1	4	0	0
COPTRI	<i>Coptis trifolia</i>	NPForb	12*	1	1	26**	0	0	0	0	0

Spp CODE	Spp name	Plant Functional Group	Semi-natural forest					Anthropogenic				
			Ftot.	LD	MD	HD	Tert.	Atot.	Der.	Hort	Rec.	
ARAHIS	<i>Aralia hispida</i>	NPForb	10*	0	2	57**	0	0	0	0	0	
AROARB	<i>Photinia pyrifolia</i>	NShrub	7	0	0	0	40**	0	0	0	0	
ASTNOV	<i>Symphyotrichum novi-belgii</i>	NPForb	7	1	0	2	9	19	40**	2	0	
EUTGRA	<i>Euthamia graminifolia</i>	NPForb	6	1	1	2	5	7	47**	0	0	
POAPAL	<i>Poa palustris</i>	NPGraminoid	0	0	0	0	0	4	30**	0	0	
rumacet	<i>Rumex acetosa</i>	EPForb	0	0	0	1	0	4	30**	0	0	
luppol	<i>Lupinus polyphyllus</i>	EPForb	0	0	0	0	0	5*	33**	0	0	
rumcri	<i>Rumex crispus</i>	EPForb	0	0	0	0	0	7*	39**	0	0	
SOLCAN	<i>Solidago canadensis</i>	NPForb	1	0	1	0	1	10*	54**	0	0	
viccra	<i>Vicia cracca</i>	EPForb	0	0	0	0	0	12**	53**	0	0	
daucar	<i>Daucus carota</i>	EBForb	0	0	0	0	0	12**	37**	1	0	
chrleu	<i>Leucanthemum vulgare</i>	EPForb	0	0	0	0	0	16**	40**	1	0	
acepla	<i>Acer platanoides</i>	ETree	1	0	0	0	26**	18**	0	11	0	
alopra	<i>Alopecurus pratensis</i>	EPGraminoid	0	0	0	0	0	19**	34**	1	3	
POTSIM	<i>Potentilla simplex</i>	NPForb	0	0	0	2	3	19**	28**	4	0	
agrcap	<i>Agrostis capillaris</i>	EPGraminoid	0	0	0	0	12	27**	0	24**	2	
matmat	<i>Matricaria discoidea</i>	EAForb	0	0	0	0	0	28**	0	3	57**	
hiepil	<i>Hieracium pilosella</i>	EPForb	1	0	1	0	2	29**	1	18	5	
versers	<i>Veronica serpyllifolia ssp. serpyllifolia</i>	EPForb	0	0	0	0	0	29**	1	18*	9	
polare	<i>Polygonum arenastrum</i>	EAForb	0	0	0	0	0	30**	0	11	22**	
poaann	<i>Poa annua</i>	EAGraminoid	0	0	0	0	1	33**	2	10	18*	
phlpra	<i>Phleum pratense</i>	EPGraminoid	0	0	0	0	2	41**	46**	4	10	

Spp CODE	Spp name	Plant Functional Group	Semi-natural forest					Anthropogenic				
			Ftot.	LD	MD	HD	Tert.	Atot.	Der.	Hort	Rec.	
ACHMIL	<i>Achillea millefolium</i>	NPForb	0	0	0	0	0	41**	1	43**	0	
pruvul	<i>Prunella vulgaris</i>	EPForb	0	0	0	0	0	41**	0	39**	3	
cennig	<i>Centaurea nigra</i>	EPForb	0	0	0	0	1	42**	62**	7	0	
stegra	<i>Stellaria graminea</i>	EPForb	0	0	0	0	0	46**	55**	13	0	
agrsto	<i>Agrostis stolonifera</i>	EPGraminoid	0	0	0	0	1	46**	12	24**	4	
ranrep	<i>Ranunculus repens</i>	EPForb	2	1	0	0	2	49**	38**	16	2	
tripra	<i>Trifolium pratense</i>	EPForb	0	0	0	0	2	60**	22	24*	9	
cervul	<i>Cerastium fontanum</i> ssp. <i>vulgare</i>	EPForb	0	0	0	0	0	69**	8	47**	8	
fesrubr	<i>Festuca rubra</i> ssp. <i>rubra</i>	EPGraminoid	0	0	0	0	4	74**	35	37**	2	
leoaut	<i>Leontodon autumnalis</i>	EPForb	0	0	1	0	1	75**	6	58**	6	
plamaj	<i>Plantago major</i>	EPForb	0	0	0	0	1	78**	5	34	42**	
trirep	<i>Trifolium repens</i>	EPForb	0	0	0	0	0	85**	4	40	44**	
taroff	<i>Taraxacum officinale</i>	EPForb	1	0	1	0	2	90**	27	31**	28	
poapra	<i>Poa pratensis</i>	EPGraminoid	0	0	0	0	0	91**	16	32	44**	

** p<0.01, * p<0.05

Fragmentation Related Influences

The DCA performed on the semi-natural forested plots (ordination not shown) had eigenvalues of 0.457, 0.281, and 0.172, respectively, which accounted for 32.5%, 19.8%, and 6.5% of the variance, respectively. The first axis is significantly correlated with a number of anthropogenic factors, including distance to the forest boundary ($\rho = 0.44$), distance to trail ($\rho = -0.38$), and historical use (0.27) (Table 8). The second DCA axis is associated with trail width ($\rho = -0.29$), disturbance ($\rho = -0.27$), and historical use (-0.24).

Table 8: Spearman correlation coefficients (ρ) between DCA axes, disturbance, historical use, and fragmentation-related factors.

Variables	Axes1		Axes2		Axes3	
Historical use	0.27	***	-0.24	**	0.39	***
Distance to forest boundary	-0.44	***	-0.11		-0.23	**
Boundary aspect ($^{\circ}$ from S)	-0.04		0.01		0.14	
Distance to trail	-0.38	***	-0.04		0.07	
Trail aspect ($^{\circ}$ from S)	0.13		0.09		-0.01	
Trail width	0.20	*	-0.29	***	0.08	
Disturbance	0.17	*	-0.27	***	-0.13	

*** $P < 0.001$, ** $p < 0.01$, * $p < 0.05$

Table 9: Standardized canonical coefficients and intraset correlations for disturbance and selected anthropogenic variables. The canonical coefficients represent the unique contribution of individual variables to the CCA axes, whereas the intraset correlations convey the simple correlation between variables and axes.

Variables	Standardized canonical coefficients			Intraset correlations		
	Axes 1	Axes 2	Axes 3	Axes 1	Axes 2	Axes 3
Historical influences	-0.25	0.04	0.13	-0.71	0.19	0.67
Distance to forest boundary	0.20	0.00	0.11	0.64	-0.24	0.63
Distance to trail	0.13	-0.03	0.03	0.53	0.19	0.36
Trail width	-0.02	-0.06	0.01	-0.3	-0.47	0.21
Disturbance	-0.06	-0.22	0.00	-0.11	-0.97	-0.01

Relationships among anthropogenic factors and vegetation composition discovered in the DCA were further supported by the CCA. Eigenvalues for the CCA axes were 0.151, 0.058, and 0.028, which accounted for 11%, 4.2% and 2% of the variation in the PFG matrix, respectively. Land-use legacies and edge influences constructed the first CCA axis, whereas the second was predominantly one of disturbance (Table 9 and Figure 4). Monte Carlo permutations found the eigenvalues and species-environment correlations (0.721 and 0.578) to be highly significant ($p < 0.01$).

As can be discerned from the ordination of PFGs, exotic taxa of a variety of life-history strategies and growth forms are generally associated with greater degrees of anthropogenic influence (Figure 5). However, exotic PFGs vary with regard to their relationship to anthropogenic influences. Exotic annual graminoids and trees were arranged at the far left side of the first axis, suggesting that they are highly associated with close proximity to anthropogenic edges and sites historically subject to high amounts of human activities. In contrast, exotic shrubs were arranged at the far right-hand side of the first axis, indicating that they are not particularly associated with those anthropogenic factors and may be prominent at sites subject to minimal amounts of human influence. Similarly, although native PFGs were generally associated with lower degrees of human influence, relationships vary depending on other traits. For example, native biennial and annual forbs are positioned farther left along the first axes than most other native PFGs, indicating that they are promoted by the measured anthropogenic stressors.

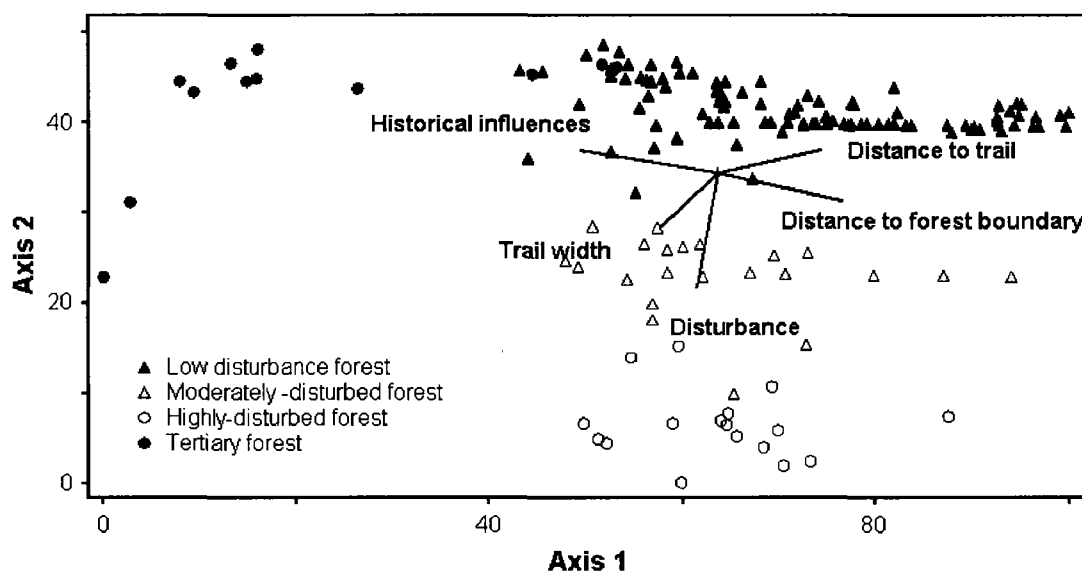


Figure 4: CCA joint plot ordination of semi-natural forested plots. Analysis based on PFGs within the understory. The length and direction of the vectors represent the strength and association of the environmental variables in explaining variation in species composition within semi-natural forest habitats.

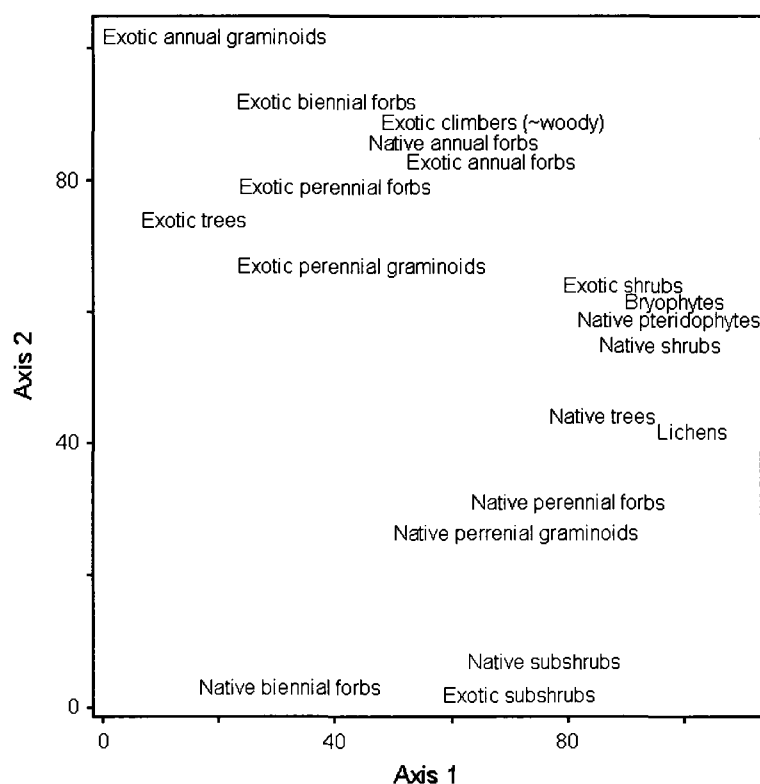


Figure 5: CCA ordination of understory PFGs within semi-natural forested ecosystems.

Discussion

Vegetation Structure and Composition

Plant communities within urban parks varied considerably in their structural character. They ranged from remnants of original forest which have relatively well-developed structural attributes, to the comparatively simple construct of some anthropogenic habitats that are actively managed. The relatively low structural diversity observed within many managed habitats reflects an obvious degradation of their ecological integrity and is promoted by the predominant horticultural aesthetics of the region and time period. In particular, the well-developed herbaceous layer and low cover of non-vasculars, shrubs, and tree canopy reflects the desire for well-tended grass lawns, which has been a dominant horticultural preference across North America and elsewhere during the past century (Freedman 2009).

However, anthropogenic habitats within urban parks vary in their structure and therefore in the ecological processes they support. For example, greater variation in basal area of trees in horticultural sites compared to derelict or recreational habitats reflects their superior potential to sequester carbon. Studies within the urban matrix of HRM and elsewhere have similarly observed a high carbon storage potential of older stands of urban forest, which are characterized by low stem densities but large trees (McPherson and Rowntree 1989; Rowntree and Nowak 1991; Freedman et al. 1996; Turner et al. 2005). Similarities between tertiary forests and primary / secondary ones suggest that if left alone, successional processes will recover much of the structural character lost as a result of management practices, although land-use legacies may persist.

Vegetation is well-known to reflect the occurrence of human activities after they have subsided and natural succession is allowed to take place. For example, a study of plant communities in urban habitats in Phoenix found that those with a history of farm use had 43% fewer woody plant genera than those which had never been cultivated (Hope et al. 2003). Similarly, a study on the Yucatan Peninsula found that forests which had regenerated over old Mayan ruin sites were not as species rich, had greater mean basal area of stems, and contained different plant species than other forest sites (White and Hood 2004). Such influences may extend for considerable periods of time - for example,

Duffy and Meier (1992) found that some understory herbs in mixed Appalachian forests had not recovered from clearcutting after 87 years. Results from this study demonstrate that tertiary forests have particularly high abundances of exotic taxa, are relatively impoverished of snags and CWD, and have certain understory attributes (such as non-vascular cover, prominence of perennial graminoids) more closely resembling that of highly disturbed, rather than intact, forests. Although evidence of conversion to an anthropogenic land-use type may persist for extended periods of time, studies within eastern North America have demonstrated that their strength does decline with successional development. For example, a chronosequence study performed on old-field deciduous forests in Ohio found that exotics (as well as annuals and biennials) declined with site age (Vankat and Snyder 1991). As such, despite land-use legacies (such as the persistence of a few key exotic invaders), many forest attributes may be considered quite resilient to human perturbations given sufficient time for natural successional processes to operate.

Urbanization is one of the most biologically homogenizing human activities (McKinney 2006). The replacement of local native species with exotic ones causes the floras of cities in different biogeographical regions to become more similar (i.e., beta diversity is reduced) (Kühn and Klotz 2006; McKinney 2006; Schwartz et al. 2006). However, at more local scales, urban areas are often quite biologically diverse due to species introductions (Sax and Gaines 2003). For example, results from this study and elsewhere (e.g. Hope et al. 2003; Turner et al. 2005) demonstrate that specific urban habitat-types may have similar, or greater, alpha diversity than more natural ones. Some anthropogenic habitat-types (i.e., areas managed for intensive recreation) do have low alpha diversity, however. Low diversity in such habitats may reflect exceptionally high degrees of anthropogenic stress, caused by management (mowing) and recreational (trampling) activities. Regardless of the alpha diversity of specific habitat-types, the homogenizing effect of management activities on beta diversity is evident when variation among anthropogenic sites is compared to that of semi-natural ones.

The high prominence of exotic taxa within urban ecosystems may be attributed to two general factors: (a) the extensive importation of exotic species, and (b) the provision of favorable habitat and management for their establishment and persistence (McKinney

2006). The importation of exotic plants may be intentional (such as for agricultural, forestry, or horticultural purposes) or accidental (for example, via ballast water or contaminated seed stock) (Freedman 2009). Disturbance, in general, is regarded as an important factor promoting the invasion of habitats by exotics (Alpert et al. 2000; Davis et al. 2000). Disturbances may alter environmental conditions such that previously well-adapted native species may be put at a competitive disadvantage with exotics (Byers 2002). However, as noted by others (Simberloff 1997; McKinney 2006), the anthropogenic nature of urban habitats may be what promotes their dominance by exotics, rather than disturbance per se. Additional physical changes in urban environments that may encourage exotics have been identified (see Sukopp 2004). For example, the heat-island effect (the universal tendency for ambient mean temperatures in urban areas to be higher than in the surrounding landscape) allows exotics to inhabit higher latitudes and climates than those where they are naturally found (Kowarik 1990). In addition, urban soils generally have high alkalinity (from the extensive use of concrete and other lime-based materials), which promotes the growth of plants requiring high soil pH (Gilbert 1991).

Forests that were subject to severe hurricane disturbance were not more invaded by exotic taxa than intact ones. Although disturbance (in general) is widely recognized as an important factor assisting species introductions, resource availability is also a key determinant (Alpert et al. 2000; Davis et al. 2000). For example, Davis et al. (2000) propose a general theory of invasibility whereas plant communities become more susceptible to invasion whenever there is an increase in the amount of available resources. This conceptual model holds that competition is less important in recently disturbed environments in which the already-established vegetation is not likely to be utilizing all available resources (due to both reduced resource usage and increased resource levels), and in which colonization by new species is therefore common (Davis et al. 2000). If one is to embrace this theory, then it might be reasoned that forest invasibility within the study area is limited by a resource which was not greatly promoted by the hurricane-induced disturbance and /or that is effectively sequestered by remaining vegetation. Although Luken (2003) suggests that the low prominence of exotics within the understories of many forests of northeastern North America is primarily due to low light

availability, this resource was presumably abundant within highly-disturbed forests (due to removal of the overstory). As such, exotics may be more limited by nutrient availability (particularly nitrogen, phosphorous, of calcium) than light. Given the highly fragmented nature of the study area and the fact that many exotics encountered during the study are wind-dispersed, it seems unlikely that dispersal limitations account for the low abundance of exotics within the area's forests.

Species with short life-history strategies may also be promoted by anthropogenic influences. The success of annuals and biennials in anthropogenic habitats likely reflects the intensity of stress to which vegetation is subjected. For example, they may be able to avoid drought stress associated with compact soils because they have the capability of completing their life cycle during short periods when water availability is high.

Management Opportunities

If maintaining native biodiversity is to be a goal of urban planning, then the setting aside of green areas, such as parks, is important. For example, rare species are often found in urban parks and other kinds of protected areas, where they are particularly associated with habitats that have not been subject to high intensities of anthropogenic development (Kendle and Forbes 1997; Godefroid 2001; Godefroid and Koedam 2003b). Such associations strengthen the call to protect indigenous communities within the urban landscape and stress the need for knowledge of ecology and natural history to guide the design and management of parks. However, the high human density and need for recreational areas within cities makes the development of strictly protected areas difficult. As such, management initiatives focused on improving ecological integrity must be developed in union with those for more utilitarian services. One reasonable management goal could be to provide adequate habitat for indigenous plant communities in forest remnants and assist in the dispersal of their constituent species through more intensely managed ones. In this context, the restoration (at least in part) of highly modified habitats may be important for conserving regional biodiversity if such initiatives provide appropriate environmental conditions for the dispersal of native forest species.

Urban parks represent an obvious starting point to promote naturalization within the urban environment due to (a) their opportunities for education, (b) the presence of

native plant communities from which indigenous species may be sourced, and (c) established social perceptions regarding their role as places of conservation value. Although there may be little room for improving the vegetative integrity of sites managed for intensive recreation, those serving more passive human activities (i.e. “horticultural” sites) may benefit from a variety of naturalization efforts. The principal means of restoration and naturalization is the cultivation of native species in ways that develop semblances of natural community types. Although exotic species are undesirable in this context, they may contribute to important ecological functions (Kendle and Rose 2000). As such, strict “native only” policies may not be the most useful means of promoting EI. Furthermore, a number of philosophical, ecological, and technical issues complicate the identification of exotic species (Kendle and Rose 2000) and management resources directed at their complete eradication may quickly become exhausted. In addition to promoting the prominence of native species, efforts which encourage structural heterogeneity and diversity of growth forms in ways that mimic the character of more natural ecosystems, may also serve to increase EI within cities.

Conclusion

This study has identified patterns in vegetation composition and structure within urban parks in association with gradients of anthropogenic stress caused by management activities and fragmentation. Vegetative communities subject to intense management regimes lack many of the structural attributes of more natural ones including amounts of tree basal area, woody debris, and non-vascular, shrub, and tree canopy cover. Conversely, they have a much higher herbaceous cover than less intensely managed habitats and are more dominated by exotic taxa. Plant communities within forests that are currently subject to minimal amounts of management activities are affected by secondary processes associated with this stressor. That is, semi-natural forests are affected by adjacent anthropogenic communities via edge influences and by historic land-use practices. In particular, these factors promote exotic plants. Such patterns reflect past and current aesthetic ideals as well as the desire to provide areas for intensive recreational activities. However, these forces are considered here to represent a degradation in EI, an effect that can be partly mitigated by naturalization efforts within urban parks.

Chapter 3: Relationships among Anthropogenic Edges and Exotic Plants within Urban Forest Remnants

Abstract

Anthropogenic edges are well-known to promote the spread of exotic plants within forested ecosystems. The influence of edges on vegetation composition is particularly important in highly fragmented areas such as urban environments. In urban areas, remnants of natural habitat are surrounded by a matrix of human infrastructure and are often subject to high intensities of recreational use. This study examined gradients in exotic cover as well as the richness of alien and native taxa as a function of distance from the edges of anthropogenically maintained forest boundaries and recreational trails within selected urban parks of Halifax Regional Municipality, Nova Scotia. Specific study objectives were (a) to describe plant composition at different distances from trails and forest boundaries, (b) to test for the simultaneous influence of both edge types on trail vegetation, and (c) to estimate the distance-of-edge-influence to which forest boundaries and trails influence plant composition. Both 2 m and 10 m wide plots (distributed amongst 11 urban forest remnants) were employed in order to describe edge influences at multiple spatial scales. Gradients in trail vegetation were investigated among three distance-from-forest-boundary increments: 10-50, 50-100, and >100 m. Randomization tests were used to determine if trail vegetation was simultaneously influenced by both edge types and to quantify distance-of-edge-influences. For both forest boundaries and trails, exotic cover as well as the richness of alien and native taxa decreased with greater distance from the edge. Forest boundaries and trails acted in an additive manner to affect vegetation composition and exerted a distance-of-edge-influence of 40-60 m and at least 4-6 m, respectively. These results are important for designing urban forest remnants which conserve interior forest through appropriately sized patches and trail densities.

Introduction

Urban forest remnants provide important habitat for native species but their structure and composition is known to be influenced by a number of fragmentation-related factors (Hobbs 1988; Matlack 1993a; Bastin and Thomas 1999; Godefroid and Koedam 2003b; Guirado et al. 2006). In particular, they may be comprised of proportionally high amounts of edge-influenced habitat compared to more contiguous forested systems. Edge influence may be defined as “the effect of processes (both abiotic and biotic) at the edge that result in a detectable difference in composition, structure, or function near the edge, as compared with the ecosystem on either side of the edge” (Harper et al. 2005). Urban forest remnants are subjected to two prominent anthropogenic edge influences – those from the surrounding urban matrix (i.e., at the forest boundary) and those within the forest, formed by recreational trails (which in some cases also serve as maintenance roads) (Figure 6).

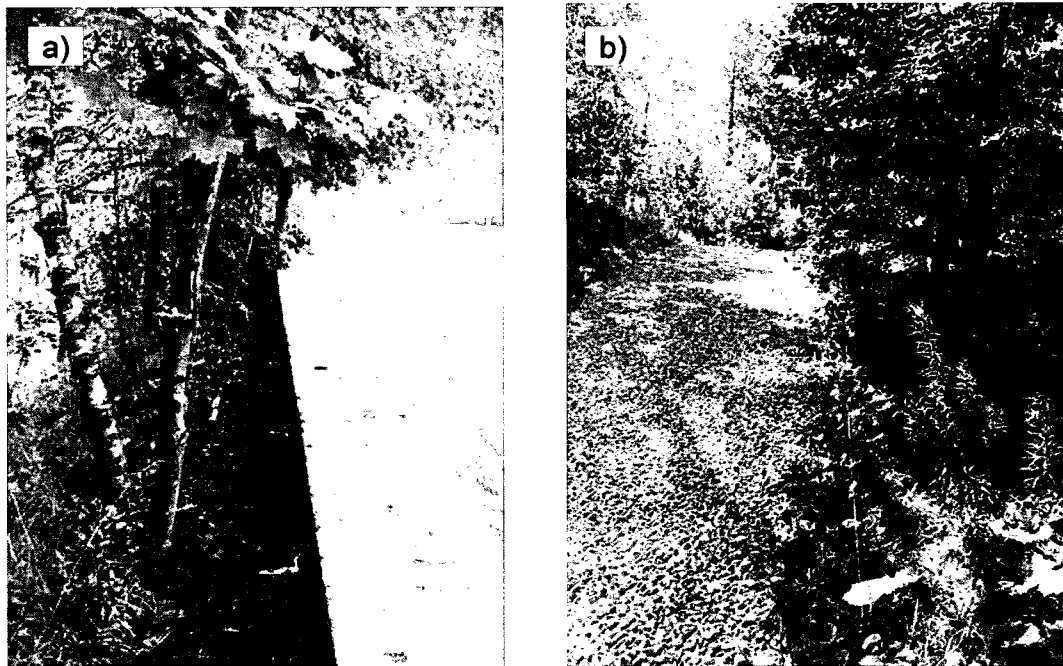


Figure 6: Anthropogenic edges formed by (a) a forest boundary and (b) a recreational trail.

Edges are associated with gradients in vegetation composition. In particular, alien and synanthropic species (those inhabiting anthropogenic habitats) are often more

abundant close to a forest boundary (Brothers and Spingarn 1992; Fraver 1994; Burke and Nol 1998; Honnay et al. 2002; MacQuarrie and Lacroix 2003). Habitats adjacent to trails also tend to have a greater abundance and richness of alien taxa than do forest interiors (Benninger-Truax et al. 1992; Bhuju and Ohsawa 1998; Godefroid and Koedam 2004; Baret and Strasberg 2005; Dickens et al. 2005). Many additional vegetation responses to edges have been observed. For example, increased amounts of ruderals, disturbance indicators, nitrogen-demanding species, and plants associated with high soil pHs have been associated with trails (Godefroid and Koedam 2004). Conversely, certain native taxa are less abundant close to forest boundaries (McDonnell and Pickett 1990; Burke and Nol 1998; Harper et al. 2004). Such patterns in vegetation reflect differences in the availability of resources (Gehlhausen et al. 2000), human frequentation (Guirado et al. 2006), and numbers of propagules from non-forest species (Cadenasso and Pickett 2001). Edge influences generally decrease with depth into the forest and are affected by a multitude of local and regional environmental variables. The large number of factors involved results in edge influences being largely site-specific, although some regional trends may be identified (see Harper et al. 2005).

The distance-of-edge-influence (DEI) may be defined as the depth into the adjacent community over which there is a significant edge influence (Harper et al. 2005). DEI estimates associated with forest boundaries vary considerably and range from 3 to 120 m for measures of vegetation structure and composition (Brothers and Spingarn 1992; Fraver 1994; Matlack 1994; Burke and Nol 1998; Honnay et al. 2002; MacQuarrie and Lacroix 2003). In contrast, the influence of trails on adjacent plant communities is generally considered to be restricted to within several meters (Dale and Weaver 1974; Cole 1987; Benninger-Truax et al. 1992; Dickens et al. 2005). However, this may be much greater at points of interest, where users may extend their explorations away from the trail edge (Cole 1987).

Edge influences in urban ecosystems may be particularly important ecologically. For example, Moran (1994) found that forests adjacent to residential land-uses had more introduced species, higher species richness, and showed greater similarity with the non-forest habitats than those next to agricultural lands. Forests adjacent to urban ecosystems may be more altered than those next to other land-use types as a result of higher amounts

of recreational activities at their edges (Moran 1984; Guirado et al. 2006). As such, the DEI on plant composition within urban forest remnants may be greater than for rural ones (Matlack 1993a). Furthermore, due to extensive habitat loss and fragmentation, small forest patches may be comprised mostly of edge-influenced habitats and may be simultaneously influenced by multiple edges (Kapos 1989; Fernandez et al. 2002; Fletcher 2005; Harper et al. 2007). Conversely, however, some types of edge influences may be less pronounced within urban settings depending on the intensity of other anthropogenic stressors (i.e. that may override edge influences).

Despite the importance of edges in urban ecosystems, they have been subject to few studies compared to those conducted within forest-management or agricultural contexts (but see Godefroid and Koedam 2003b; Howard et al. 2004; Guirado et al. 2006; Hamberg et al. 2008). Furthermore, only one study (Hamberg et al. 2008) has estimated a DEI for the response of vegetation within urban forest remnants. The relative lack of such studies may in part be attributable to the complexity of urban environments, which makes the description of edge influences difficult (Murcia 1995).

Understanding the influence of edges on forest composition is important for the conservation of native plant communities. Because exotic taxa are known to be strongly promoted by edges, they may serve as important indicators of this stressor. In this context, they may be particularly useful in areas with high habitat heterogeneity, such as the Acadian Forest Region, where detecting gradients in the abundances of native species or structural attributes is complicated by large degrees of variation. In addition, understanding processes of exotic plant invasions is important because they are known to negatively affect native species (Standish et al. 2001) and to alter ecological processes (Gordon 1998; Brooks et al. 2004). As such, this study focuses on measures of exotic taxa with the following specific objectives: (a) describe plant composition at different distances from trails and forest boundaries, (b) test for the simultaneous influence of both edge types on trail vegetation, and (c) estimate the distance-of-edge-influence to which forest boundaries and trails influence plant composition.

Methods

Study Area

This study takes place within urban parks of HRM, Nova Scotia. The study area (approximate center of study area located at 44° 39' N, 63° 34' W and area of 188 km²) is located within the urbanized landscape of the Eastern Interior Ecodistrict of the Acadian Ecozone, as defined by Neily et al. (2003). This ecodistrict has an undulating to gently rolling topography and bedrock consisting of Meguma Group quartzite and slate. Soils within the study area are predominantly composed of the Halifax, Bridgewater, and Wolfville series, which consist of well-drained, often stony, loams derived from quartzite, slate, and shale/sandstone, respectively (MacDougall et al. 1963). The depth of till throughout the ecodistricts varies from <1 to 10 m and averages <3 m (Neily et al. 2003).

Forests within the Eastern Interior Ecodistrict are varied and reflect such factors as the depth of the soil profile, drainage, disturbance regime, and site aspect. Those encountered during the study varied considerably in seral stage and composition but were mostly mixed woods with coniferous and angiosperm content. The most prominent trees were *Acer rubrum* (red maple), *Picea rubens* (red spruce), and *Pinus strobus* (white pine), although areas dominated by *Fagus grandifolia* (American beech), *Tsuga canadensis* (eastern hemlock), and *Quercus rubra* (red oak) were encountered. *Betula papyrifera* (white birch) was common throughout most of the forests while *Larix laricina* (eastern larch), *Picea mariana* (black spruce), and *Betula alleghaniensis* (yellow birch) were uncommonly encountered as scattered individuals. Much of the forest within the study area was subject to varying intensities of disturbance by a hurricane (Juan) in September 2003.

Forests within the region have been subject to a range of anthropogenic stressors. Nova Scotia's forests, in general, have been harvested and managed for up to four centuries, and as a result few stands have escaped human influence (Loo and Ives 2003). The most prominent change in forests throughout the province has been a shift in the relative abundance of successional stages and associated changes in structure and composition, brought about by forest management and agricultural activities. These practices have resulted in the average forest age decreasing from about 200 years (in pre-

settlement times) to 55 years today; a decrease in the abundance of late-successional, shade-tolerant tree species; and an increase in early-successional, shade-intolerant ones (Loo and Ives 2003). Most of the forest stands within the parks have likely been harvested at one time, but relatively few appear to have regenerated from a previous conversion to an anthropogenic land-use type (such as for agriculture, residential, or industrial purposes). Forest remnants within the urbanized landscape are surrounded by a variety of anthropogenic land-use types such as residential developments, sports fields, and transportation infrastructure. Maintained recreational trails are common throughout most of the forest remnants and are on average approximately 3 m in width.

Sampling Design and Data collected

Study sites were identified from the larger subset of 24 urban parks within HRM previously selected in Chapter 2 which used a stratified (on total park size) random sampling procedure. Those which were selected for inclusion in this study ($n = 11$) contained natural forest remnants >0.5 ha in size (Table 10). “Natural forest remnants” are defined for the purposes of this investigation as those which appear to have regenerated by natural successional processes and that have not been previously converted to an anthropogenic land-use type.

Trail, Boundary, and Reference Plots

Plots were used to quantify the vegetation at the edges of forest boundaries and trails, as well as in reference conditions (see Figure 7). Only anthropogenically created forest boundaries were targeted, such as those formed by residential lots, roads, or sport fields. Boundary plots were distributed without regard to edge orientation or type of adjacent land use because of limited opportunities to replicate according to these factors (i.e. the character of forest boundaries were highly varied). This study focused on primary trails, defined as those created and maintained in a purposeful manner by park management as evidenced by the addition of gravel or other management of the terrain. Narrow footpaths that are formed as a result of spontaneous recreational use within forested ecosystems and are not maintained by park management initiatives were not studied. Because vegetation gradients alongside trails were hypothesized to vary

according to their depth into the forest, trail plots were distributed among three distance-from-boundary intervals: 10-50 m, 50-100 m, and >100 m. Boundary and trail plot locations were determined by identifying edge intervals using park maps and then using a random number table in the field to identify the distance (m) and side (left or right) along these intervals where the plot centers were to be positioned. Reference conditions were defined as areas >100 m from a forest boundary and >10 m from a primary trail. Reference plot locations were randomly determined by identifying suitable areas (i.e., based on previously described spatial requirements), overlaying a grid (the dimensions of which varied according to the size of the area), and using a random number table to select grid squares, the midpoint of which served as the plot center.

The number of plots within reference conditions, alongside boundaries, and adjacent to trails (within each distance-from-boundary intervals) within a particular forest remnant was determined by its size; 1, 2, and 3 plots were distributed within <2 ha, 2-8 ha, and >8 ha remnants, respectively. However, due to limitations in the availability of appropriate sites (caused by variation in the size and shape of the forest remnants, distribution of primary trails throughout, and character of the park borders), the desired numbers of plots within individual parks were not always obtainable. For example, only 4 parks (Admiral's Cove, Cyril Smith, Hemlock Ravine, and Point Pleasant) contained forest >100 m from a forest boundary, and in one of them (Admiral's Cove) primary trails were not present (Table 10). In contrast, boundary plots were obtainable in all parks except one (Remington Court) whose borders were either not represented by a forest edge or were too close to trails to be included in the study.

Each plot was 10 m x 10 m in size and was subdivided into five 2 m x 10 m subplots (Figure 7). For plots situated along an edge, subplots were located 0-2 m, 2-4 m, 4-6 m, 6-8 m, and 8-10 m from the edge. An additional 2 m x 10 m subplot was placed at a distance of -2 to 0 m from the edge. The 0 m edge position was identified with reference to a combination of factors, including the presence or absence of altered substrate (e.g., gravel, artificial bank), mowed vegetation, trampling effects, and the locations of tree boles. Reference plots were oriented so that the long axis of the subplots was perpendicular to the aspect of the site. In order to limit the confounding influence of edges that were not targeted, plots were >10 m away from additional forest boundaries

and trails. All plots were >20 m away from others and located within relatively intact forest, which is defined as that not highly disturbed (< 75% of dominant trees blown down, see Methods section in Chapter 2). Due to high heterogeneity in the character of forests throughout the study area and the desire to maximize the number of replicates, effort was not made to control for forest “type,” as may be defined by factors such as seral stage, dominant tree composition, soil, or drainage properties.

Data were collected during the summer of 2007. All herbaceous and woody vascular plants within the understory (<2 m in height) of the 2 m x 10 m subplots were identified to species and assigned a percent cover value (based on a visual estimation). Woody species within the overstory (>2m) were only assigned cover estimates at the 10x10 m scale. A single cover value was estimated for each species in the 2-10 m strata, whereas values for taxa >10 m in height were obtained by averaging measurements taken at each plot corner and center (total of 5 measurements) using an ocular tube. Percent cover values were estimated as the percentage of the ground surface covered by the outline of the photosynthetic crown. Cover values >10% were recorded to the nearest 1%, whereas those <10%, <1%, and <0.1% were recorded to the nearest 0.5%, 0.1%, and 0.01% increment respectively.

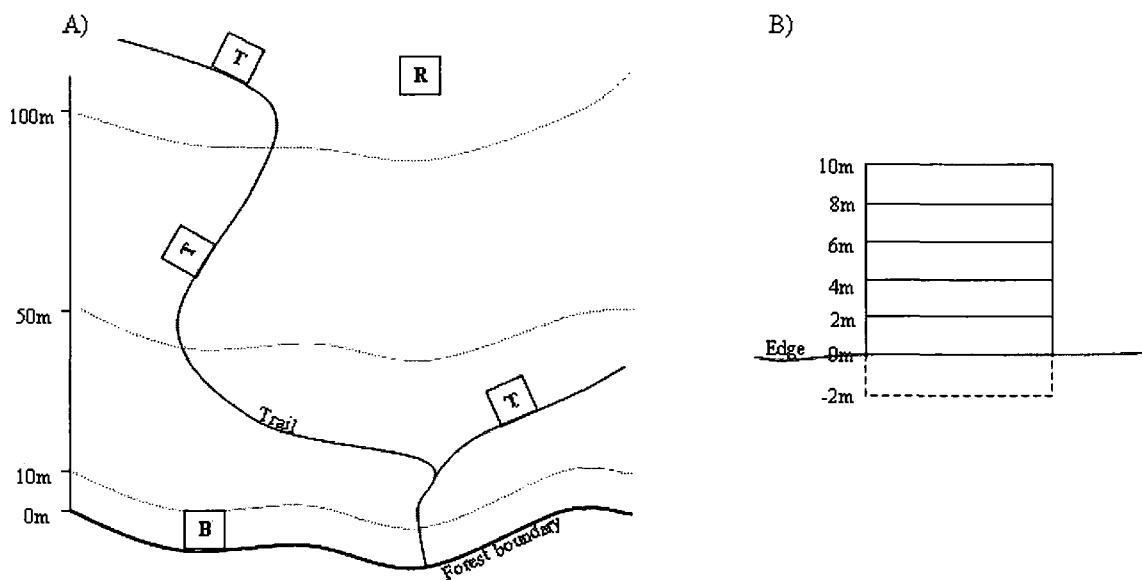


Figure 7: (a) Hypothetical layout of 10 m x 10 m forest boundary (B), trail (T) and reference (R) sampling plots – dotted lines represent the 10-50 m, 50-100 m, and >100 m distance from the forest boundary intervals used to stratify the trail plots; and (b) plot layout in relation to the edges of trails or forest boundaries – plots were subdivided into 2 m x 10 m subplots with an additional subplot located -2 to 0 m from the edge (trail and boundary plots only).

Table 10: Size of forest remnants and the number of boundary, trail, and reference sampling plots.

Park	Size of forested area (ha)	Treatments					
		Trail			Boundary	Reference	Total
		10-50m	50-100m	>100m			
Admiral's Cove	29	0	0	0	3	3	6
Alder Piper	0.6	1	0	0	1	0	2
Bell Lake	20	0	0	0	3	0	3
Cyril Smith	25	3	3	3	3	3	15
Glenbourne	1.5	1	0	0	1	0	2
Hemlock Ravine	91	3	3	3	3	3	15
Point Pleasant	66	3	3	3	3	3	15
Remington Court	0.6	1	0	0	0	0	1
Tremount Plateau	4.9	2	1	0	2	0	5
Wedgewood	1.5	1	0	0	1	0	2
Willett Street	3	2	2	0	2	0	6
Total		17	12	9	22	12	72

Random Plots

To examine gradients in vegetation composition at further distances from forest boundaries, plots were distributed randomly throughout the study area (see Methods section of Chapter 2). These plots were also 10 m x 10 m in size, but they were not subdivided into subplots, unlike those used to target forest boundaries, trails, and reference conditions. Furthermore, they were distributed without regard for their proximity to forest edges or historical land-use. The number of these plots within a forest remnant was determined by its size; with 6, 8, 10, 12, 14, 16, and 18 plots being distributed throughout areas of 0.5-1, 1-2, 2-4, 6-8, 8-16, 16-32, and >32 ha, respectively.

Vegetation composition within the plots was characterized using the same protocols as previously described for *trail*, *boundary*, and *reference* plots except percent cover values for taxa within the understory (<2 m in height) were assigned at the 10 x 10 m scale. Additional environmental data collected included (a) the distance from the plot center to the nearest forest boundary, (b) distance from the plot center to the nearest recreational trail, (c) any evidence of a past conversion to an anthropogenic land-use type, and (d) intensity of hurricane-induced disturbance (low, moderate, or high) (see “Data Collected” in Methods section of Chapter 2).

Random plots which (a) had been previously converted to an anthropogenic land-use type, (b) were <10 m from primary trails, or (c) were subject to severe disturbance were considered unsuitable for the purposes of this study, and were discarded from the dataset. The remaining plots were placed into one of eight distance-from-boundary intervals: 0-10, 10-20, 20-30, 30-40, 40-60, 60-80, 80-100 and >100m. Wider distance classes were used further from the forest boundary in order to compensate for the greater inaccuracy which may be associated with larger distance measurements (those approximately 50 m or greater were more commonly obtained using GIS information rather than measured in the field) and to maximize the number of replicates in the categories (low replication is a reflection of the random sampling strategy used for the survey and the size of the forest remnants available). Random plots 0–10 m and >100 m from boundaries were added to the *boundary* and *reference* plots, respectively (Table 11).

Table 11: Distribution and replication of 10 m x 10 m plots used for estimating the DEI of forest boundaries (includes boundary, reference, and random plots but not trail plots).

Park	Distance (meters from the edge of forest boundaries)								Total
	0-10	10-20	20-30	30-40	40-60	60-80	80-100	>100	
Admiral's Cove	3	1	1	2	1	3	1	10	22
Alder Piper	1	3	0	0	0	0	0	0	4
Bell Lake	3	4	2	2	3	1	1	0	16
Cyril Smith	4	3	0	0	2	1	1	7	18
Glenbourne	2	3	0	0	0	0	0	0	5
Hemlock Ravine	4	1	1	1	1	1	4	10	23
Point Pleasant	4	2	0	1	2	0	0	5	14
Remington Court	1	3	0	0	0	0	0	0	4
Tremount Plateau	3	3	1	1	1	1	0	0	10
Wedgewood	1	4	0	0	2	0	0	0	7
Willett Street	2	3	2	0	0	0	0	0	7
Total	28	30	7	7	12	7	7	32	130

Analyses

Variables Examined

Total exotic cover, exotic dominance, and the numbers of exotic and native species were calculated for the understory and overstory of each plot and/or subplot (only understory measures were calculated for subplots). Total exotic cover was calculated by summing values for all alien taxa. Dominance was calculated as the relative cover of exotics (total exotic cover / total cover of all taxa). Although most studies focus on the former, dominance was included because it is an important measure of invader success and total values may underestimate the relative cover (Lundholm and Larson 2004). Native species richness was examined to provide context for any patterns observed in alien richness. The mean and standard error (SE) of each measure were calculated for all distance intervals at both the 2 m and 10 m scales. Calculations were performed using SPSS 15.0 (SPSS Inc. 2006) and Microsoft Excel (Microsoft Corporation 2002).

Exotic taxa were defined as those whose presence is a result of an accidental or intentional introduction beyond their natural range by human activities (Richardson et al. 2000a). All species introduced to Nova Scotia in following European settlement were considered exotic. In addition, species with native populations in the province were

considered “exotics” if they are restricted to regions and habitats not surveyed in this study (i.e., extralimitals) and introduced genotypes are common elsewhere. This treatment is consistent with other definitions used in a park management context (see Westman 1990). For species with indigenous and non-indigenous populations occupying similar habitats, biogeographical status was assigned based on their relative commonness. References used to identify and determine the biogeographical status of taxa include *Roland’s Flora of Nova Scotia* (Zinck 1998), *Manual of Vascular Plants of Northeastern United States and Adjacent Canada* (Gleason and Cronquist 1991), *Flora of North America: North of Mexico Series* (Flora of North America Editorial Committee 1993+), and the S-ranks of the Atlantic Canada Conservation Data Center (ACCDC 2007).

Edge Interaction

Analyses were performed to test for the simultaneous influence of both edge types (boundaries and trails) on trail vegetation. R version 2.8.0 (R Development Core Team 2008) was used to perform randomization tests on data from the 2x10 m trail subplots. The test statistics used for the analyses were the F-values generated by a series of repeated-measures two-way ANOVAs. Both distance to trail and to the forest boundary were treated as fixed factors. Randomization tests were used because response variables could not be transformed to fit the assumptions of standard parametric models (there were issues with homoscedasticity and normality). To account for the spatial autocorrelation of the subplots, an error term for the “plot” was incorporated into the model (i.e., subplots within a particular plot were treated as the “repeated measures”). The six original trail distance increments were used as factor levels, whereas the 50-100 m and >100 m distance from boundary intervals were combined. The latter was performed in order to increase the power of the randomization test, which may be compromised by low sample sizes and unequal replication (Edgington 1995). By combining the two intervals, replication changed from 17, 12, and 9 plots for the 10-50 m, 50-100 m and >100 m increments, respectively, to 17 and 21 for the 10-50 m and >50 m increments. Both main and interaction effects were tested. The analysis first computed the F-values from the repeated-measures ANOVA and then compared these to distributions of F-values obtained by a randomization of the data set, based on 5000 permutations (i.e. the

probability that $F(\text{observed}) \leq F(\text{randomized data})$ was calculated). In order to account for the lack of independence among 2 m x 10 m subplots, they were only randomized within their respective plots. These larger plots were then randomized amongst the distance-from-boundary intervals.

Distance of Edge Influence

The “Randomization Tests for assessing Edge Influence” (RTEI) program in Visual Basic, Microsoft Excel 97 (Harper and Macdonald 2009), as described in Mascarua-Lopez et al. (2006), was used to estimate the DEI for trails and forest boundaries. This randomization method compares the difference between the mean value of a variable in the interior (reference) forest, and that at a certain distance from the edge, to a distribution of differences created by a randomization of both data sets. For each distance interval the following steps were performed:

- (a) the difference in the means within the interior forest and at a given distance from the edge were calculated;
- (b) x number of values (corresponding to the number of observations within a distance-from-edge interval) were randomly selected from a data set that included values from the interior forest and those at a given distance from the edge;
- (c) the difference between the values selected in step “b” and the unselected ones was calculated;
- (d) steps “b” and “c” were repeated 5000 times to create a distribution of mean differences;
- (e) the actual mean difference, calculated in step “a” was compared to the distribution generated in step “d”.

Because it was hypothesized that measures of exotic taxa would be higher in closer proximity to edges, means that were above the 95 percentile of the randomized distribution were considered significant (i.e., one-sided test at $\alpha = 0.05$). Because it was unknown whether native species richness would be greater or less at the edge compared to interior conditions, significant differences were identified here as values less or greater than the 2.5 and 97.5 percentiles, respectively (two-sided test at $\alpha \leq 0.05$). The DEI was defined as the set of one or more consecutive intervals from the edge

(starting at the 0 m increment) into the forest over which statistically significant values were observed (modified from Harper et al. 2005; Mascarúa López et al. 2006). Forest boundary DEI was investigated at both the 2 m (subplot) and 10 m (plot) scales. RTEI analyses were performed for trails within each of the three distance-from-boundary intervals using data within the 2x10 m subplots.

Results

Forest Boundaries

Gradients in vegetation composition were evident at the 10 m scale. The total cover, dominance and richness of exotic taxa within both the understory and overstory decreased with greater distance from the edge of forest boundaries (Figure 8). The DEI for all three measures of exotic taxa was estimated to be 10-20 m and 40-60 m for the overstory and understory, respectively. Understory native species richness also decreased with successive distance increments and was observed to have a DEI of 20-30 m. No DEI was identified for overstory native species richness, however, as only the 30-40 m and 40-60 m intervals were significantly different from reference conditions.

Of the ten most frequently encountered exotic taxa (excluding those within the 0-10 m interval), six were herbs (*Hieracium flagellare*, *Hieracium floribundum*, *Hieracium pilosella*, *Leontodon autumnalis*, *Ranunculus repens*, and *Taraxacum officinale*) and four were woody or semi-woody species (*Frangula alnus*, *Quercus robur*, *Rosa multiflora*, and *Solanum dulcamara*). Three exotics (*Hieracium lachenalii*, *Hieracium pilosella*, and *Quercus robur*) were found within the understory of four reference plots (three in Point Pleasant, one in Hemlock Ravine). Nevertheless, the infrequency and low abundance of exotics within forest interiors are reflected in mean cover values (total and relative) being less than 0.05% within the reference plots. The patterns of response of individual species varied considerably (Figure 9).

Although not significantly different from reference conditions, relatively high overstory and understory means (accompanied by a high SE) of the exotic measures were observed within the 30-40 m and 80-100 m intervals, respectively. Exotics within the 30-40 m overstory interval can be attributed to a single site within Point Pleasant Park, where

Pinus sylvestris and *Quercus robur* were present at 8% and 0.5% cover, respectively. Exotics within the 80-100 m understory interval reflect a plot within Bell Lake Park where the invasive shrub *Frangula alnus* had a cover value of 16%, with lesser amounts of the herbs *Poa compressa* and *Hieracium flagellare* also present, at 0.2% and 0.05%, respectively.

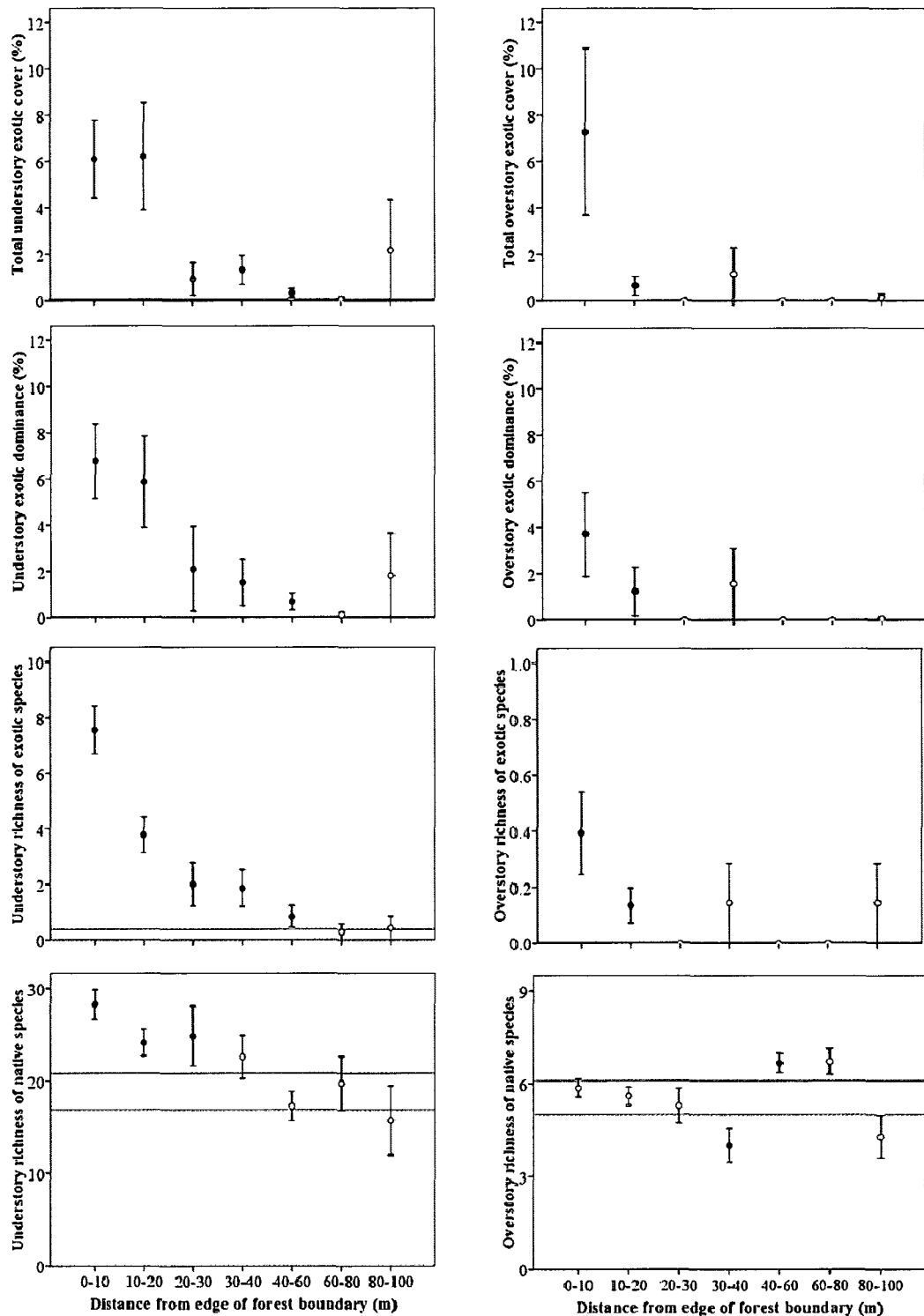


Figure 8: Mean $\pm$ 1 SE for understory and overstory total exotic cover, exotic dominance, and richness of exotic and native taxa as a function of distance from the edge of forest boundaries. The 95% confidence interval of plots >100 m from the boundary is represented by horizontal lines. Filled symbols indicate values that are significantly different from reference conditions. See Table 11 for replication.

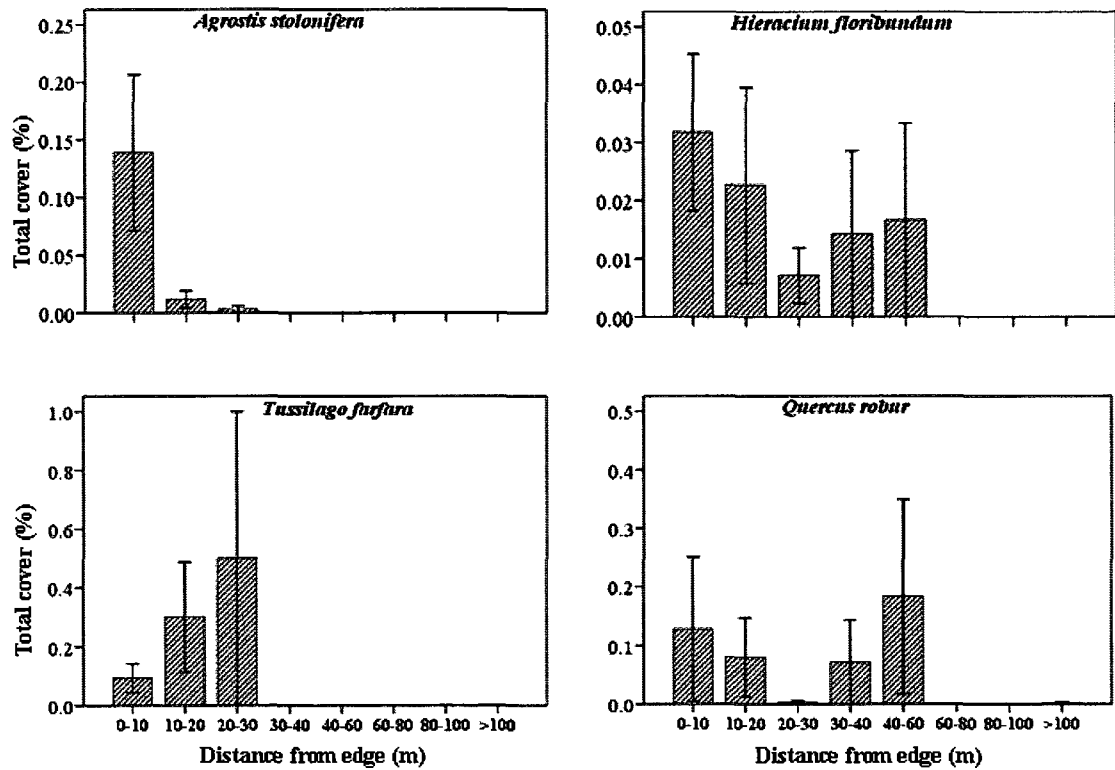


Figure 9: Mean $\pm$ 1 SE of the total cover of four prominent exotic species as a function of distance from the edge of forest boundaries. Examples provided demonstrate variation in individual species responses. *Quercus robur* values are based on combined understory and overstory values whereas others are based on the understory alone. See Table 11 for replication.

Gradients in vegetation were also observed at the 2 m scale. All measures of understory exotic taxa decreased with increasing distance from the edge (Figure 10) and all subplots were significantly different from reference conditions ($\text{DEI} \geq 10$ m). Exotic species accounted for 64 out of 195 taxa identified within plots located at the edge of forest boundaries. Of those exotics encountered in $>20\%$ of the 10 m x 10 m plots (based on an aggregation of 2 m x 10 m subplots, excluding the -2 to 0 m interval), four were graminoids (*Agrostis stolonifera*, *Anthoxanthum odoratum*, *Festuca rubra* ssp. *rubra*, and *Poa pratensis*), eight were forbs (*Hieracium floribundum*, *Hieracium lachenalii*, *Hieracium pilosella*, *Leontodon autumnalis*, *Ranunculus repens*, *Taraxacum officinale*, *Tussilago farfara*, and *Veronica officinalis*) and one was a semi-woody vine (*Solanum dulcamara*). A number of non-native tree species were also commonly encountered,

including *Ulmus glabra*, *Acer platanoides*, and *Pinus sylvestris*, which were found within 18%, 14%, and 11% of the plots respectively. Native species richness was lower within the non-forest habitat (-2-0 m) than at the forest edge (0-2 m), and then declined with increasing distance into the forest. Significant differences in native species richness between increments and reference conditions (mean richness approximately 14 species) were restricted to the 0-2 m and 2-4 m increments (DEI = 4m).

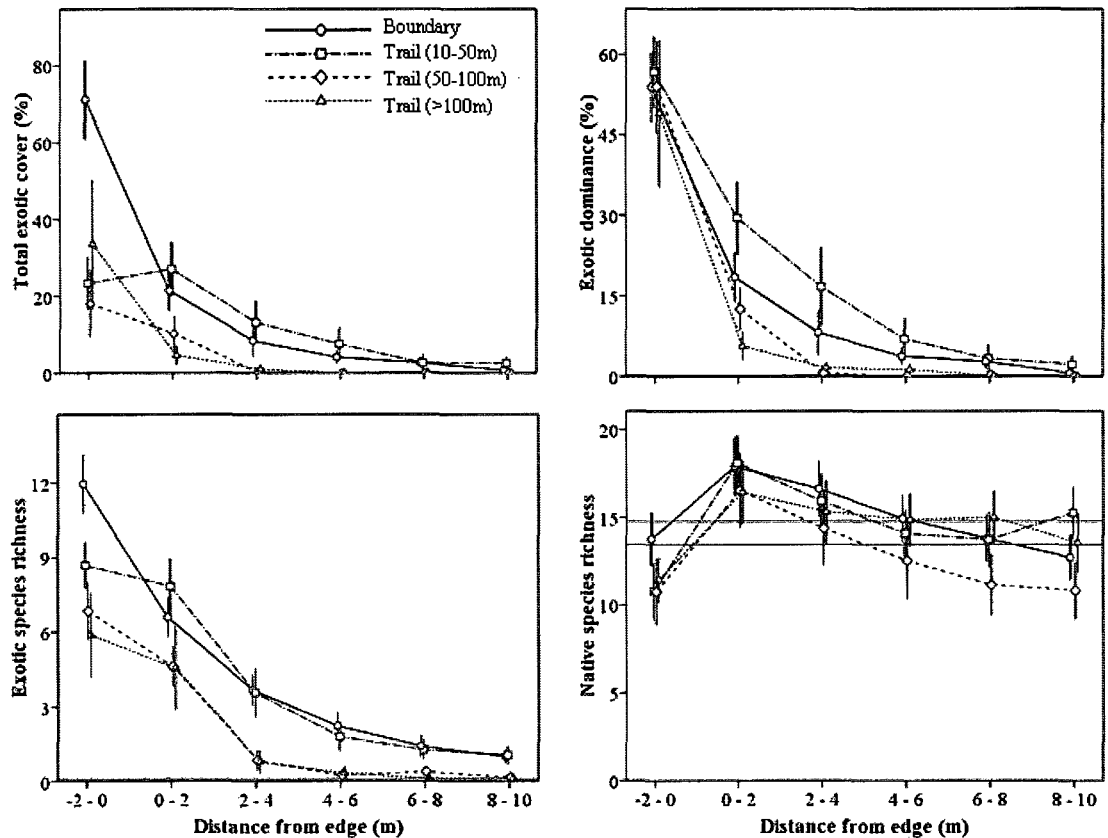


Figure 10: Mean $\pm$ 1 SE of total understory exotic cover, exotic dominance, and richness of exotic and native taxa, as a function of distance from the edge of forest boundaries and trails (for each 2 m increment, $n=22$ for boundaries, 17 for 10-50 m trails, 12 for 50-100 m trails, and 9 for >100 m trails). Horizontal lines represent the 95% confidence interval for the reference ($n = 60$ subplots).

Trails

Results from the randomized two-way repeated-measures ANOVA show that vegetation composition in close proximity to trails is simultaneously influenced by distance to the trail and to the forest boundary edge (Table 12). However, although p-values for trail influence were highly significant for all measures of vegetation, only the dominance and richness of exotics were significantly influenced by distance to the forest boundary. None of the interaction terms were significant, suggesting that forest boundaries and trails influence vegetation in an additive, rather than multiplicative, manner.

Exotic species were prominent within the trail plots, where they accounted for 66 out of 207 taxa encountered. The total cover, dominance, and richness of exotics declined with greater distance from trail edges (Figure 10). Native species richness was also observed to decline with increasing distance into the forest. However, patterns were not monotonic in that values within the 0-2 m increment were higher than those -2 to 0 m from the edge.

Patterns in the cover of individual exotics were highly variable (Figure 11). Of the species encountered in >20% of the 10x10 m trail plots (based on an aggregation of 2x10 m subplots, excluding the -2-0 m interval), five were graminoids (*Agrostis stolonifera*, *Phleum pretense*, *Poa annua*, *Poa compressa*, and *Poa pratensis*), nine were forbs (*Cerastium fontanum ssp. vulgare*, *Hieracium floribundum*, *Hieracium lachenalii*, *Hieracium pilosella*, *Leontodon autumnalis*, *Plantago major*, *Ranunculus repens*, *Taraxacum officinale*, and *Trifolium repens*) and one was a semi-woody vine (*Solanum dulcamara*). *Acer platanoides*, and *Fraxinus excelsior* were the most frequently encountered tree species, being found in 13% and 8% of trail plots respectively. Exotics were only encountered within 2 reference plots, both of which were within Point Pleasant Park. One of these plots contained two exotics (*Hieracium pilosella* and *Quercus robur*), whereas the other had one (*Quercus robur*). As such, mean total cover, dominance, and richness of exotic taxa within reference plots were low, having values of 0.07, 0.06 and 0.05, respectively (Figure 10).

Table 12: MSE, MST, and F-values generated from a two-way repeated-measures ANOVA and P-values obtained by a randomization test of the F-values.

Variable	MSE Plot (df = 36)	MSE Residuals (df = 180)	Boundary (df = 1)			Trail (df = 5)			Trail * Boundary (df = 5)		
			MST	F	P	MST	F	P	MST	F	P
Total exotic cover	910	295	3430	3.77	0.06	3760	12.72	0.00	557	1.89	0.19
Exotic dominance	873	223	4260	4.88	0.01	15900	71.32	0.00	503	2.26	0.15
Exotic species richness	25.6	5.68	221	8.63	0.01	364	64.1	0.00	9	1.6	0.22
Native species richness	183	13.3	67	0.36	0.54	176	13.22	0.00	13	0.95	0.34

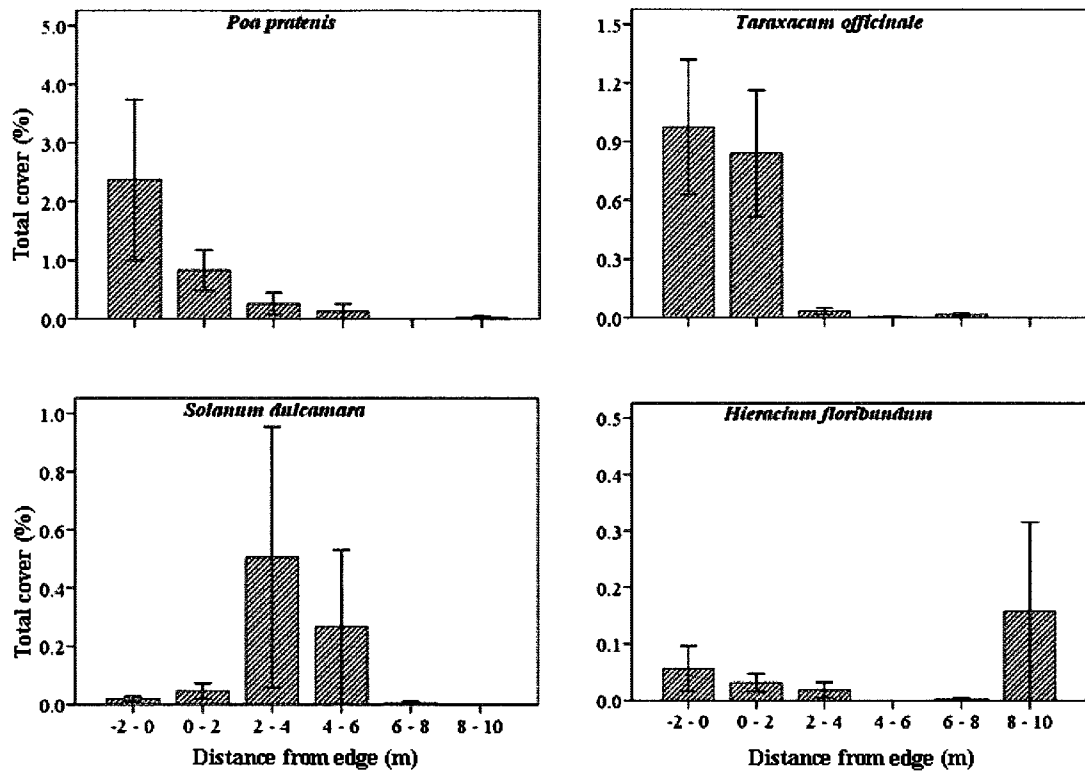


Figure 11: Mean $\pm$ 1 SE of the total cover of four common exotic species as a function of distance from the edge of trails ($n=38$ for each distance increment). Examples provided demonstrate variation in individual species responses.

Differences in trail vegetation amongst the distance-from-boundary intervals are evident. With the exception of the -2 to 0 m increment, all exotic measures were greater in plots 10-50 m from the forest boundary edge than they were for those at distances of 50-100 m or >100 m (Figure 10). When data from subplots were pooled into their respective 10 m x 10 m plots, the cover and richness of alien taxa decreased with successive distance-from-boundary intervals (Table 13). Mean subplot values of exotics declined for all successive 2 m distance increments for trails 10-50 m from the forest boundary. For trails within the 50-100 m and >100 m intervals, mean values leveled off at intermediate distance increments (within the 2-6 m range). Results from the RTEI demonstrate that measures of exotic taxa are significantly different from reference conditions to a depth >10 m for trails in the 10-50 m interval and up to 4 m and 6 m at 50-100 m and >100 m from the forest boundary, respectively (Table 13). The influence of

edges on native species richness was found to extend to a depth of 2 m for trails in both the 10-50 m and 50-100 m intervals (Table 13).

Table 13: Mean $\pm$ SE and distance of edge influence (DEI) for response variables alongside trails 10-50 m, 50-100 m, and >100 m from the forest boundary. Values presented are based on data summarized within the 10 m x 10 m plots. DEI estimates were performed using the 2x10 m subplots and are therefore not available for overstory measures.

	Trail 10-50m (n=17)		Trail 50-100m (n=12)		Trail >100m (n=9)	
	Mean $\pm$ SE	DEI (m)	Mean $\pm$ SE	DEI (m)	Mean $\pm$ SE	DEI (m)
UNDERSTORY						
Total exotic cover	11 $\pm$ 4.0	>10	2.3 $\pm$ 0.98	2	1.1 $\pm$ 0.60	4
Exotic dominance	15 $\pm$ 4.9	>10	5.1 $\pm$ 2.2	4	1.9 $\pm$ 0.88	6
Exotic species richness	9.6 $\pm$ 1.6	>10	5.3 $\pm$ 0.75	4	4.8 $\pm$ 1.8	6
Native species richness	30 $\pm$ 2.4	2	27 $\pm$ 2.9	2	27 $\pm$ 2.3	0
OVERSTORY						
Total exotic cover	5.8 $\pm$ 4.9	na	0.0 $\pm$ 0.0	na	2.8 $\pm$ 2.8	na
Exotic dominance	4.0 $\pm$ 3.2	na	0.0 $\pm$ 0.0	na	2.0 $\pm$ 2.0	na
Exotic species richness	0.12 $\pm$ 0.08	na	0.0 $\pm$ 0.0	na	0.11 $\pm$ 0.11	na
Native species richness	5.4 $\pm$ 0.52	na	5.3 $\pm$ 0.51	na	5.6 $\pm$ 5.6	na

Discussion

Forest Boundaries

The forest boundary DEI of 40-60 m estimated in this study is similar to that identified in other regions. Although they based their estimate on an overall measure of community composition, rather than exotics alone, Hamberg et al. (2008) identified a DEI of 50 m within the city of Helsinki, Finland. A study of agriculturally maintained hardwood forest edges in North Carolina estimated a DEI for the dominance of exotic species to be 60 m from south-facing edges (Fraver 1994). Although Gehlhausen (2002) found exotics to be rare and largely restricted to the edge of mixed forests adjacent to agricultural lands in Illinois, they did observe small numbers of alien plants 40-60 m from the edge at one of their sites which suggests that weak edge influences may have

extended up to this distance. Similar DEIs, based on additional measures of vegetation have been identified in non-urban systems (e.g., Harper and Macdonald 2002).

In contrast to such apparent consistency, other studies have reported different forest boundary DEIs. For example, no significant changes in the cover of exotic plants could be detected beyond 10 m in deciduous forest in Ontario (Burke and Nol 1998) and a study of upland hardwood forest on Prince Edward Island estimated a DEI of more than 120 m based on overall understory species composition (MacQuarrie and Lacroix 2003). Such disparate results are likely a reflection of a multitude of factors relating to the ecological character of the habitats on either side of the edge, the nature of human activities within those habitats, and the different methodologies used to estimate DEI.

Exotic taxa within the overstory were also higher in close proximity to the forest edge, but the DEI for these variables was much less (10-20 m) than for the understory. Although overstory composition is known to vary with distance from the edge of forest boundaries (Wales 1972; Ranney et al. 1981; Whitney and Runkle 1981), there is a lack of studies reporting gradients in the prominence of exotics. Furthermore, there have been few estimates of DEI for overstory composition and none are known for forests in urban protected areas. However, Ramney et al. (1981) found that tree species richness was influenced to a depth of 30 m for west-facing edges and 10 m for others in sugar maple and beech forests in Wisconsin. The greater prominence of exotics in the understory may reflect the higher amounts of introduced herbaceous and shrubby species compared to trees in Nova Scotia, differences in dispersal capabilities among growth forms, the longer growth time required for species to occupy the overstory, climate, and factors related to resource availabilities in the respective strata.

Species richness is well known to decrease with increasing distance from forest boundaries (Fraver 1994; Burke and Nol 1998; Gehlhausen et al. 2000; Marchand and Houle 2006). As evidenced here, and in other studies, such gradients are in part due to a higher richness of alien taxa at the forest edge (Brothers and Spingarn 1992; Gehlhausen et al. 2000; Guirado et al. 2006). However, patterns of native species richness may simultaneously reflect declines in forest species that are sensitive to the environmental conditions promoted by edges (McDonnell and Pickett 1990; Burke and Nol 1998; Harper et al. 2004) as well as greater amounts of some types of natives (Gehlhausen et al. 2000;

Guirado et al. 2006). Although edge-sensitive species have not been identified in this study, others have observed relatively strong responses for forest-interior taxa. For example, Burke and Nol (1998) found that the fern *Dryopteris intermedia* was associated with interior forest and that it was significantly influenced by the edge to a distance of 50 m. In contrast, they found that the maximum DEI for any species promoted by edge proximity was 35 m. However, the occurrence of edge-sensitive species (and the ability to detect them) is likely to vary considerably among geographic areas. For example, although Fraver (1994) found that many natives and exotics were promoted by edges up to a distance of 60m into the forest, no species suspected of being negatively correlated with edge proximity had statistically significant responses in their study.

A number of factors are responsible for variations in vegetation composition across the edge-interior ecotone. Forests closer to anthropogenic edges are more likely to experience high frequentation by people than those farther away (Guirado et al. 2006). This results in trampling and soil compaction, which are well-known to influence vegetation composition through physical damage and modification of habitat (Bagnall 1979; Godefroid and Koedam 2004; Hamberg et al. 2008). Although species richness may be lower in trampled versus non-trampled habitats (Gómez-Limón and de Lucio 1995), such disturbances may facilitate colonization by ruderals (Bhujju and Ohsawa 1998). Several of the exotics frequently encountered at edges in this study, such as *Plantago major*, have been documented to be resistant to trampling effects (Chappell et al. 1971). Higher human frequentation at edges is also expected to influence the prominence of exotics by aiding dispersal (Godefroid and Koedam 2003b). For example, humans are unintentionally able to serve as seed carriers for many plants, including many of the common exotics encountered in this study (Clifford 1956). In addition, forest boundaries are typically characterized by gradients in microclimatic variables - they generally have greater amounts of light, warmer air and soil temperatures, higher wind velocities, and lower relative humidity and soil or litter moisture than do forest interiors (Brothers and Spingarn 1992; Gehlhausen et al. 2000; Harper et al. 2005; Marchand and Houle 2006). Although higher wind velocities may enhance dispersal opportunities, gradients in other microclimatic conditions may also promote exotics by making forest conditions more similar to the non-forest habitats.

The structure of vegetation at the edge may act to inhibit the spread of exotics (Cadenasso and Pickett 2001). Many forest edges are characterized by high vegetation densities and small stem diameters that create a dense wall of vegetation (Wales 1972; Ranney et al. 1981; Brothers and Spingarn 1992; Fraver 1994), a process known as “edge sealing” (Harper et al. 2005). When formed, this wall may act to lessen the intensity of abiotic gradients across the forest ecotone (Williams-Linera 1990; Matlack 1993b). This may discourage invasion, for example, by reducing light availability and wind speeds (Brothers and Spingarn 1992). This type of natural process may be less important ecologically in urban settings due to human intervention. For example, forest edges next to residential areas become more open as a result of recreational use (Bagnall 1979; Moran 1984).

Trails

Plant communities adjacent to trails contained greater amounts of alien taxa when in closer proximity to the forest boundary. This likely reflects both variation in the influence of trails themselves (those closer to the forest boundary are likely to be frequented more often) and the permeating influence of the forest boundary. Such variation in trail influence has been demonstrated in other studies. For example, trail vegetation in Colorado has been found to be simultaneously influenced by distance from the trail edge, level of trail use, and distance from trailheads (Benninger-Truax et al. 1992). Conversely, a study of the association between exotic richness and distance from the entry point of portage trails in Minnesota did not find significant relationships (Dickens et al. 2005).

Because trail vegetation in the 10-50 m increment is also likely affected by the forest boundary and the reference condition used in this study is outside of this zone (and therefore can not be used to partition the different sources of variation), it is difficult to ascertain trail DEI for this interval. However, based on results from outside the zone of forest boundary influence (the 50-100 and >100 m intervals), trails within the study area may be considered to exert a DEI of at least 4-6 m.

A 4-6 m trail DEI is slightly less than some other estimates within forested sites subject to relatively high intensities of recreational use. For example, Hamberg et al.

(2008) estimated that the effects of trails extend approximately 8 m from their edge into urban forests in Finland, and Godefroid and Koedam (2004) found that trail influence extended at least 10 m in a beech forest in central Belgium. Conversely, other studies have estimated trail DEI to be less. Dale and Weaver (1974) suggested that the width of the disturbed vegetation on the side of trails was less than 2 m (Dale and Weaver 1974) and Cole (1987) found that the combined width of the trail and the adjacent disturbed zone usually spans only 3 m (Cole 1987). However, the later studies took place in wilderness areas where trail width is narrower, human frequentation likely less, and forest boundaries less abundant than in urban parks.

Linear travel corridors such as trails promote the invasion of habitats by acting as routes for agents of dispersal, providing suitable habitat, and by providing reservoirs of propagules for future episodes of invasion (Parendes and Jones 2000). Vegetation composition within close proximity to trails may also be influenced by gradients of disturbance (related to trail construction, maintenance, and trampling effects) (Cole 1987), light intensity (Bates 1935; Dale and Weaver 1974; Cole 1978; Hall and Kuss 1989), grazing pressure (Dale and Weaver 1974; Cole 1981), soil density (Bates 1935), soil moisture (Bates 1935; Burden and Randerson 1972), and root competition (Dale and Weaver 1974). These gradients act to promote the amount of edge-influenced forest, thereby making the size of the functional interior smaller than its actual area (Fraver 1994).

Regional Considerations

Exotics did not comprise a major component of the forest community, except for immediately at edges, suggesting that forests within the HRM may be relatively resistant to invasions. For example, whereas only three alien taxa were encountered >100 m from the boundary edge (representing less than 5% of the number of alien taxa identified in the study), a study in southern Ontario found that nearly half of the alien taxa encountered at the edge were also found 100 m into the forest (Burke and Nol 1998). Stapanian et al. (1998) performed surveys of exotics across the continental U.S. and found that compared to most regions, aliens plants were less prominent in northeastern forests. For example, whereas exotics were estimated to comprise 1.5% of the understory forests of the

Northeast, they accounted for 13%, 18%, and 25% in the Southeast, Mid-Atlantic, and in California, respectively. Although the environmental factors influencing a habitat's invasibility are not completely understood (Alpert et al. 2000; Davis et al. 2000), a number of regional features may be important, including the intensity of past and present anthropogenic disturbances, nutrient and light availabilities, climate, and human density (Stapanian et al. 1998; Luken 2003).

The present low prominence of aliens within the study area may not be indicative of future conditions, however. Of the five species identified by Hill and Blaney (2008) as being problematic invasives of natural habitats within the Maritime Atlantic Ecozone, two were commonly encountered in this study: *Frangula alnus* (European alder-buckthorn) and *Pinus sylvestris* (Scotch pine). These were found in 5% and 10% of the random plots, respectively, and were observed to obtain high cover values at sites far from the edge of the forest boundary (up to 40 m and 100 m for the overstory and understory, respectively). In addition, *Poa nemoralis* (wood bluegrass), *Rosa multiflora* (multiflora rose), and *Rosa rugosa* (rugose rose) were relatively common throughout the study area and are identified as potential threats to native biodiversity by Hill and Blaney (2008). Furthermore, other exotics are known to be highly invasive of upland forests within northeastern North America – for example, the herb *Alliaria petiolata* (garlic mustard) (Nuzzo 1999), which has only recently been introduced to Nova Scotia (Hill and Blaney 2008). Because edge influence can extend with time and alter the interior of even large forested areas (Soulé 1986; Nuzzo 1999), the presence of potentially invasive species within the study area is of conservation concern. *Alliaria petiolata*, for example, has been documented to spread throughout forested ecosystems as an advancing “front,” moving at an average of 5 m in a year, with the rate influenced by the occurrence of satellite populations and disturbances (Nuzzo 1999).

Study Limitations and Future Research

In this study, the number of plots within any forest remnant varied with its size and character. The nesting of plots within the remnants represents a potential source of variation that was not accounted for. Unequal plot replication means that the results are influenced more by some remnants than others. However, the implications of this

apparent lack of independence on the estimates of edge influence provided is considered to be reduced by a high degree of variation in the character of forests within a given park and its surrounding urban matrix. Nonetheless, it is apparent that exotics are more prominent in some parks (e.g. Point Pleasant) than others (e.g. Admiral's Cove). Future research directed at understanding the affects of time and intensity of human activities on exotic invasion, or other edge influences, may benefit from focused studies that compare individual remnants. Conversely, studies aimed at describing edge influences at a regional scale would benefit from the use of a study design that treats individual remnants as treatment blocks.

Because of the highly varied character of the study area, this project did not attempt to control for factors such as forest type (tree composition, seral stage, soil properties), character of the adjacent habitat (residential, industrial, transportation, commercial, recreational, etc.), time since edge creation, intensity of human visitation, or edge aspect. However, such factors are known to affect the intensity of edge influences (Moran 1984; Cole 1987; Benninger-Truax et al. 1992; Brothers and Spingarn 1992; Godefroid and Koedam 2004). To better understand the role of these variables on urban edge influences, future research could be targeted at more homogenous systems.

The DEI results obtained here have been compared to those from other studies in an attempt to gain insight into the relative ecological importance of edge influences. However, when comparing results it must be kept in mind that studies have used different analytical methods to identify DEI and this can greatly affect the results (Harper and Macdonald 2008). In addition to randomization techniques, as are employed here and by others (Harper and Macdonald 2001; Mascarúa López et al. 2006), DEIs have been estimated using Helmert procedures (Fraver 1994; Burke and Nol 1998), clustering and ordination (MacQuarrie and Lacroix 2003; Hamberg et al. 2008), visual estimation (Gehlhausen et al. 2000), and the application of a "two-thirds rule" (Chen et al. 1992; MacQuarrie and Lacroix 2003). If researchers and managers are to gain a more comprehensive appreciation of edge dynamics and their implications for conservation, work should be done towards a common analytical method for identifying DEI.

Management Opportunities

The relatively low prominence of exotic taxa within and outside the edge-influenced zone suggests that even small forest remnants can provide important refuge for native plant communities within a highly fragmented urban ecosystem. However, given that urban edge influences may increase with time and that they are likely to affect vegetation composition in other ways (such as by decreasing the abundances of sensitive native species), a precautionary approach is recommended for the design and management of urban forest remnants. If indigenous plant communities are to be a valued component of urban forest remnants then they should be designed to accommodate as much interior habitat as possible. This requires consideration of the size of edge-influenced buffers. For example, if edge influences are considered to extend to a depth of 60 m, a circular patch of forest of 3.1 ha would conserve only 0.5 ha of interior habitat.

The long-term conservation of forest vegetation within an urban environment must also consider the distribution of trails. Due to their ability to help spread invasives that may threaten native woodland flora, Godefroid and Koedam (2003) recommend avoiding the creation of internal edges, such as trails. However, recreational trails do provide some indirect ecological benefits by encouraging human interaction with nature (a service that is especially important in urban areas) and by concentrating recreational use that might otherwise result in a proliferation of unmanaged paths. At a minimum, trail systems should be designed to allow for the existence of habitat outside of their influence and to avoid areas with unique or sensitive vegetation. Whereas the DEI of trails is varied, our results show that their influence extends at least 4-6 m into the forest. Based on such results, the edges of nearby trails would have to be approximately 24 m apart to conserve a strip of unaltered vegetation equal in size to that influenced by a single trail (not including the area that the trail directly occupies). However, because of the ability of edge influences to extend with time, a more precautionary approach than is outlined here may be warranted.

Chapter 4: Discussion

The EI concept has important implications for the design and management of urban areas. However, if it is to be widely adopted as a guiding principle within cities, then a number of social, technical, and philosophical issues would need to be addressed. This chapter provides discussion on some of these issues. More specifically, the role of social perceptions of nature within cities, the concept of a “reference” condition to which current states may be compared, and the use of indicators and indices as quantitative measures of EI are discussed.

Perceptions of Nature

One of the major challenges to improving ecological integrity in urban settings is overcoming human perceptions of the role and importance of “nature” in cities. The detachment of much of the populace from nature and their unwillingness to support meaningful conservation initiatives has been identified as the major challenge to urban wildlife conservation (Noss 2004; Turner et al. 2004; McKinney 2006). For example, there is a low acceptance of indigenous vegetation within urban environments due to perceptions that it is associated with certain risks to society and individuals (health, property damage, crime, etc.) and that cultivation “improves” nature by making it more useful (Breuste 2004). While certain reservations regarding the naturalization of cities are understandable (such as concerns regarding certain wildlife “pests”), others are strongly influenced by malleable aesthetic preferences and inadequate understanding of ecological functions.

Fortunately, the importance of conserving native biodiversity is increasingly being acknowledged by society. This is reflected in many international, national, and regional initiatives, such as the *Convention on Biological Diversity*, the *Canadian Species at Risk Act*, and development of the *2C1Forest (Two Countries, One Forest)* organization. To complement the ecological motives for conserving native biodiversity is an awareness of the more utilitarian services provided by healthy environments. For example, citizens within HRM have indicated that the protection of the natural environment is a key priority for “preserving quality of life, community identity, and opportunities for outdoor

recreation” (HRM 2006a). As such, the municipality aims to “foster the development of an integrated system of natural areas, parks, trails and corridors to maintain ecosystem health and preserve HRM’s quality of life” (HRM 2006a).

Human perceptions of their relationship with nature are strongly influenced by their exposure to and understanding of the natural environment (Sebba 1991; Rohde and Kendle 1994; Chiesura 2004). In this context, it has been suggested that urban national parks, that include anthropogenic habitats as well as natural ones, be established. Besides providing important habitat for native species, such a system would enhance the status of urban nature, thereby encouraging city residents to become more familiar with indigenous biodiversity (Niemelä 1999). Exposure to habitats managed with concepts of naturalization in mind may further encourage a more unified view of the relationship between humans and nature by integrating current aesthetic ideals with the characteristics and species of natural communities.

Reference Conditions

If EI were adopted as a guiding principle for the design and management of urban areas, then it is important to define the state that is to be achieved (i.e. the reference condition). In a North American context, a natural reference condition may be considered to represent the state of the ecosystem and landscape prior to European colonization, and may be regarded as the “best” ecological state that can be attained (Andreasen et al. 2001). However, such definitions ignore the influence of Native Americans and bring forth a number of philosophical and ecological questions regarding appropriate rates of change. Despite such ambiguities, “natural” reference conditions may be helpful in the management of large protected areas. However, it would be unreasonable to attempt to achieve such a state within urban areas (Noss 2004). As such, concepts of “sustainability” rather than “naturalness” may be better employed in defining the desired condition of urban ecosystems. A “sustainable” condition may be defined as one that has been “altered from its pre-settlement, natural state but has responded in a stable manner to the changes” (Andreasen et al. 2001). Within this context, management initiatives could identify specific goals for achieving a state of EI relevant for urban areas. For example, one objective could be to provide adequate habitat for the long-term persistence of indigenous

plant communities within remnants of natural habitats, while also assisting the dispersal of their constituent species through managed ones. In this context, naturalization efforts that promote self-sustaining semblances of natural communities appropriate to site conditions are important.

Multimetric Indices

Following the adoption of EI (sometimes more specifically chemical, physical, and/or biological integrity) as a guiding principle for many initiatives throughout the United States and Canada, were efforts to develop ways in which it may be measured. In this context, ecological indicators are important tools for simplifying assessment and monitoring activities. However, the complexity of ecological systems requires that a large number of indicators be used if current environmental conditions and ecological responses are to be appreciated. As such, indices which aggregate indicator measures are being sought to assist land managers.

Attempts at quantifying EI have focused on developing multimetric indices that relate to the cumulative effects of multiple anthropogenic stressors on ecosystems. They do this by integrating a number of indicators, referred to as “metrics,” that relay information on anthropogenic stressors. The general approach of developing multimetric indices may be summarized in four key steps (Paul 2003): (1) select the metrics, (2) calculate index values for each of them, (3) aggregate the metric index values into an overall index, and (4) interpret the index values. Such indices may be developed at various spatial and organizational scales. For example, within an urban context they could focus on individual parks or be designed for an entire urban landscape or ecological region.

While there are criticisms regarding the use of such indices (see Suter 1993), the main concern is that they are oversimplifications of ecological information. If not accounted for, the loss of important information may result in incorrect interpretations and unwarranted generalizations. This is a serious problem whenever any set of attributes is reduced into a single index. Therefore, although indices designed to reflect the state of EI within a particular context may be useful tools, they should not be depended upon

exclusively. It is also important to have an in-depth examination of particular aspects of ecosystems.

Index of Biological Integrity

Because of their inherent ecological importance and ability to provide insight into multiple environmental stressors and ecological responses, multimetric indices are commonly based on biological measures and are known as *Indices of Biotic Integrity* (IBIs). James Karr has been at the forefront of developing IBIs, which he first used to assess streams based on fish communities (Karr 1981). In his original approach, a composite index was developed using metrics of species richness, relative abundances within species groups, measures of trophic structure, and proportions of individuals having abnormalities associated with pollutants. At each study site, the metrics were assigned values that represented their comparability to reference conditions, and were then summed to acquire a single datum representing biotic integrity. Since Karr's original insight, IBI's have been refined and applied to other taxa such as aquatic invertebrates (Kerans and Karr 1994; Kimberling et al. 2001), birds (O'Connell et al. 2000), and plants (Mack 2001; Jones 2005); and to additional ecosystems including wetlands (Mack 2001), lakes (Minns et al. 1994), sagebrush steppe (Kimberling et al. 2001), and forest (O'Connell et al. 2000). Although no plant IBIs are known to have been developed in the context of urban areas, several studies (e.g. Wang et al. 2001) have successfully designed them based on fish data for city streams.

Since the inception of IBI's, increased attention has been given to objectively selecting metrics. Operationally, metrics have been identified by testing their response across a number of sites that are considered to vary in their EI. In this context, certain anthropogenic stressors, or combinations thereof, may serve as initial EI surrogates for against which candidate metrics are evaluated. The validity of any such surrogate(s) depends on their relative importance in influencing biotic communities, compared to other anthropogenic stressors. Given the importance of management, recreation, and fragmentation-related stressors in cities, they may serve as useful surrogates of EI within an urban protected-area context. However, other factors may also be important, such as intensity of exposure to pollution.

If variation in the intensity of management and edge-related influences were to be used as surrogates for EI within urban parks, then results from this study suggest that a number of stand-level structural and compositional measures of vegetation could serve as useful indicators (see Table 14). Particularly useful metrics include those that are relatively robust against variation in natural processes and may be expected to consistently express gradients in multiple anthropogenic stressors. For example, this study found that exotic taxa (cover and richness) were promoted by management and edge-related influences, and showed little variation amongst intensities of hurricane disturbance. Their consistent response to anthropogenic stress and apparent resilience to variation in natural processes make exotic taxa ideal indicators of EI within the study area. In this context, a number of the PFGs may also serve as valuable indicators of EI. Measures that are only known to respond to specific stressors may also be helpful, and can be particularly valuable in some contexts by helping to diagnose the cause of ecological changes. For example, measures of stand-level structure may be used to convey information on the intensity of management-related stress (although forest structure is known to be influenced by edge influences too, this was not investigated by the present study). Although individual species (native or exotic) may also be used, their occurrence at a particular site is more highly influenced by local environmental conditions. As such, they exhibit higher degrees of variation and can not be as generally applied as structural attributes or groups of taxa which have been identified by some common trait(s).

Table 14: Examples¹ of ecological indicators that are relevant for assessing and monitoring EI within urban ecosystems.

Indicator-type	Example
Landscape-level structure	Area of urban park(s) Area of semi-natural habitat remnant(s) Interior: edge-influenced (boundary) habitat Trail density in semi-natural habitat(s) Isolation of semi-natural habitat(s)
Stand-level structure	CWD basal area Snag density, basal area Tree density, basal area, canopy cover Shrub cover Herbaceous cover Non-vascular cover
Stand-level composition	Exotic cover (relative or total) Exotic species richness Community similarity (e.g., multivariate indices) PFGs (e.g., native pteridophytes) Individual species (e.g., <i>Taraxacum officinale</i>)

¹Examples provided are based on results from Chapters 2 and 3 in addition to the affects of fragmentation as predicted by the Theory of Island Biogeography (for landscape-level measures).

The IBI approach has traditionally been dependent upon ecosystems being classified into relatively homogenous groups, which requires consideration of such enduring site features as edaphic and climactic properties, natural disturbance regime, seral stage, and biological composition. However, in highly heterogenous systems such as the forests of the Acadian Ecoregion, the use of such classifications may be impractical if monitoring efforts wish to describe ecological changes based on information from multiple forest “types”. Dependence on a strict classification scheme for ecological assessment and reporting within such complex ecoregions is complicated by a high degree of variation in natural ecological processes and character, in the contexts of both space (e.g., in soil and moisture regimes) and time (e.g., in disturbance and ensuing seral stages). However, because the intensity of change that is to be detected depends on the strength of the classification used (more refined classifications are able to detect more subtle changes), generalized schemes may be useful for programs that are aimed at describing shifts in highly altered systems. Results from this study demonstrate that

coarse classifications (e.g., “upland forest”) may be useful in urban environments because the intensity of anthropogenic stress is strong enough to allow their influence to be detected using biological measures despite high variation in the natural character of forests.

Index of Ecological Integrity

An index of ecological integrity (IEI) would be the ultimate composite indicator by which to assess and monitor patterns of EI within urban ecosystems. The basic tenets for structuring and developing an IEI are the same as those for IBIs. However, an IEI would aggregate a more comprehensive set of ecological metrics that encompass the temporal, spatial, and organizational diversity of ecosystems. The comprehensiveness of an IEI for urban ecosystems would depend on the scale at which it is developed and applied. For example, an IEI could be developed specifically for remnants of urban forest, for a system of protected areas, or for an entire urban landscape. With any increase in spatial and organizational scale, there is a greater need to integrate a more comprehensive set of metrics that cross ecosystem boundaries.

With increasing spatial extent, landscape-level structural metrics become particularly important. Such measures can provide information on ecological processes and may be quickly derived from remote sensing and geographic information system (GIS) technology. Structural metrics that are applicable to urbanized landscapes include the amount of specific communities within the landscape, ratios of edge-influenced to interior habitats, measures of landscape connectivity, and degrees to which forest remnants are internally fragmented by corridors such as trails (see Table 14). However, information on landscape-level structural features needs to be complemented by ground-level measurements. For example, a measure of the spatial extent of specific habitat types is of little relevance to EI without more detailed information on vegetation structure and composition, including the dominance by alien species. Such information reflects changes in the intensity of fragmentation-related factors and variation in the character of management activities over time.

Although IEIs are currently desired by environmental managers, they are presently viewed as being more of a promising research area rather than a tool ready for

implementation. However, the conceptual foundations for IEI development have been outlined (see Andreasen et al. 2001) and key principles are already being applied to landscapes, such as within the Mid-Atlantic region of the continental United States (Paul 2003).

Conclusion

The concept of EI is relevant to the design and management of urban landscapes, but it has received little attention in this context. However, efforts by a number of conservation-oriented initiatives have been developing methods by which to measure and monitor EI within a variety of ecosystem types and at multiple spatial and organizational scales. As this body of experience continues to develop, managers will be increasingly poised to formally integrate concepts of EI into the design and management of urban landscapes, including protected areas. However, increasing urbanization and its associated ecological consequences call for a more immediate approach to improving EI within cities, and this may be aggressively initiated even though suites of indices may not yet be ready for use. The present study provides information on variation in composition and structure among urban habitats and uses this information to provide insights into important ecological processes acting within urban parks. This information may be used to help direct urban ecosystems towards a state of increased EI by assisting management initiatives in designing more sustainable systems of protected areas. Key aspects of those efforts will be measures to reduce the prominence of alien species and anthropogenic habitats, and to maintain or increase that of native species and natural communities.

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Appendix A: Vascular taxa encountered in study area (includes species in study plots and other observations)

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
ABIBAL	<i>Abies balsamea</i>	balsam-fir	Pinaceae	Tree	Perennial	Native	S5
ACEGIN	<i>Acer ginnala</i>	amur maple	Aceraceae	Tree	Perennial	Exotic	na
ACENEG	<i>Acer negundo</i>	Manitoba maple	Aceraceae	Tree	Perennial	Exotic	SE
ACEPEN	<i>Acer pensylvanicum</i>	striped maple	Aceraceae	Tree	Perennial	Native	S5
ACEPLA	<i>Acer platanoides</i>	Norway maple	Aceraceae	Tree	Perennial	Exotic	SE
ACERUB	<i>Acer rubrum</i>	red maple	Aceraceae	Tree	Perennial	Native	S5
ACESAC	<i>Acer saccharum</i>	sugar maple	Aceraceae	Tree	Perennial	Native	S5
ACESPE	<i>Acer sp.</i>	maple	Aceraceae	Tree	Perennial	Exotic	na
ACESPI	<i>Acer spicatum</i>	mountain maple	Aceraceae	Tree	Perennial	Native	S5
ACHMIL	<i>Achillea millefolium</i>	yarrow	Asteraceae	Forb	Perennial	Native / exotic	S5SE
ACTALB	<i>Actaea pachypoda</i>	white baneberry	Ranunculaceae	Forb	Perennial	Native	S4
ACTRUB	<i>Actaea rubra</i>	red baneberry	Ranunculaceae	Forb	Perennial	Native	S5
ACTSP	<i>Actaea sp.</i>	baneberry	Ranunculaceae	Forb	Perennial	Native	S4S5
AEGPOD	<i>Aegopodium podagraria</i>	goutweed	Apiaceae	Forb	Perennial	Exotic	SE
AESSP	<i>Aesculus sp.</i>	horse-chestnut	Hippocastanaceae	Tree	Perennial	Exotic	SE
AGREUP	<i>Agrimonia eupatoria</i>	European grovebur	Rosaceae	Forb	Perennial	Exotic	SE
AGRCAP	<i>Agrostis capillaris</i>	brown top	Poaceae	Graminoid	Perennial	Exotic	SE
AGRGIG	<i>Agrostis gigantea</i>	red top	Poaceae	Graminoid	Perennial	Exotic	SE
AGRPER	<i>Agrostis perennans</i>	perennial bentgrass	Poaceae	Graminoid	Perennial	Native	S4S5
AGRHYE	<i>Agrostis scabra</i>	tickle-grass	Poaceae	Graminoid	Perennial	Native	S5
AGRSP	<i>Agrostis sp.</i>	bentgrass	Poaceae	Graminoid	Unknown	na	na
AGRSTO	<i>Agrostis stolonifera</i>	creeping bent-grass	Poaceae	Graminoid	Perennial	Exotic / native	S5SE
AJUREP	<i>Ajuga reptans</i>	bugle-weed	Lamiaceae	Forb	Perennial	Exotic	SE
ALCFIL	<i>Alchemilla filicaulis</i>	thin-stem lady's-mantle	Rosaceae	Forb	Perennial	Exotic	SE
ALLSPE	<i>Allium sp.</i>	garlic	Liliaceae	Forb	Perennial	Exotic	na

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
ALNINC	<i>Alnus incana</i>	speckled alder	Betulaceae	Shrub	Perennial	Native	S5
ALNVIR	<i>Alnus viridis</i>	downy alder	Betulaceae	Shrub	Perennial	Native	S5
ALOGEN	<i>Alopecurus geniculatus</i>	water fox-tail	Poaceae	Graminoid	Perennial	Exotic	SE
ALOPRA	<i>Alopecurus pratensis</i>	meadow fox-tail	Poaceae	Graminoid	Perennial	Exotic	SE
AMBART	<i>Ambrosia artemisiifolia</i>	common ragweed	Asteraceae	Forb	Annual	Native	S5
AMELAE	<i>Amelanchier laevis</i>	smooth serviceberry	Rosaceae	Tree	Perennial	Native	S5
AMMSP	<i>Ammelanchier sp.</i>	serviceberries	Rosaceae	Shrub	Perennial	Native	na
ANAMAR	<i>Anaphalis margaritaceae</i>	pearly everlasting	Asteraceae	Forb	Perennial	Native	S5
ANTNEG	<i>Antennaria neglecta</i>	everlasting	Asteraceae	Forb	Perennial	Native	S?
ANTODO	<i>Anthoxanthum odoratum</i>	sweet vernal grass	Poaceae	Graminoid	Perennial	Exotic	SE
APOAND	<i>Apocynum androsaemifolium</i>	spreading dogbane	Apocynaceae	Forb	Perennial	Native	S5
ARAHIS	<i>Aralia hispida</i>	bristly sarsaparilla	Araliaceae	Forb	Perennial	Native	S5
ARANUD	<i>Aralia nudicaulis</i>	wild sarsaparilla	Araliaceae	Forb	Perennial	Native	S5
ARCMIN	<i>Arctium minus</i>	common burdock	Asteraceae	Forb	Biennial	Exotic	SE
ARTSTE	<i>Artemisia stelleriana</i>	beach wormwood	Asteraceae	Forb	Perennial	Exotic	SE
ARTVUL	<i>Artemisia vulgaris</i>	common wormwood	Asteraceae	Forb	Perennial	Exotic	SE
ATHFEL	<i>Athyrium filix-femina</i>	lady fern	Dryopteridaceae	Pteridophyte	Perennial	Native	S5
AVEFAT	<i>Avena fatua</i>	wild oats	Poaceae	Graminoid	Annual	Exotic	SE
BARVUL	<i>Barbarea vulgaris</i>	winter-cress	Brassicaceae	Forb	Biennial	Exotic	SE
BERTHU	<i>Berberis thunbergii</i>	Japanese barberry	Berberidaceae	Shrub	Perennial	Exotic	SE
BETALL	<i>Betula alleghaniensis</i>	yellow birch	Betulaceae	Tree	Perennial	Native	S5
BETPAP	<i>Betula papyrifera</i>	white birch	Betulaceae	Tree	Perennial	Native	S5
BETCOR	<i>Betula papyrifera</i> var. <i>cordifolia</i>	canoe birch	Betulaceae	Tree	Perennial	Native	S5
BETPOP	<i>Betula populifolia</i>	grey birch	Betulaceae	Tree	Perennial	Native	S5
BIDFRO	<i>Bidens frondosa</i>	common beggar's-ticks	Asteraceae	Forb	Annual	Native	S5
BRAERE	<i>Brachyleytrum septentrionale</i>	bearded short-husk	Poaceae	Graminoid	Perennial	Native	S4S5
BROINE	<i>Bromus inermis</i>	smooth Hungarian brome grass	Poaceae	Graminoid	Perennial	Exotic	SE
CALCAN	<i>Calamagrostis canadensis</i>	blue-joint	Poaceae	Graminoid	Perennial	Native	S5

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
CALVUL	<i>Calluna vulgaris</i>	scotch heather	Ericaceae	Shrub	Perennial	Exotic	SE
CALSEP	<i>Calystegia sepium</i>	hedge-bindweed	Convolvulaceae	Forb	Perennial	Exotic / native	S5SE
CAPBUR	<i>Capsella bursa-pastoris</i>	shepherd's purse	Brassicaceae	Forb	Annual	Exotic	SE
CARARC	<i>Carex arctata</i>	black sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARBRU	<i>Carex brunescens</i>	brownish sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARCAN	<i>Carex canescens</i>	hoary sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARCOM	<i>Carex communis</i>	fibrous-root sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARCRA	<i>Carex crawfordii</i>	crawford sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARCRI	<i>Carex crinita</i>	fringed sedge	Cyperaceae	Graminoid	Perennial	Native	S4S5
CARCUM	<i>Carex cumulata</i>	clustered sedge	Cyperaceae	Graminoid	Perennial	Native	S4S5
CARDEB	<i>Carex debilis</i>	white-edge sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARDEF	<i>Carex deflexa</i>	short-stemmed sedge	Cyperaceae	Graminoid	Perennial	Native	S4
CARDEW	<i>Carex deweyana</i>	short-scale sedge	Cyperaceae	Graminoid	Perennial	Native	S4
CARECH	<i>Carex echinata</i>	little prickly sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARFLA	<i>Carex flacca</i>	sedge	Cyperaceae	Graminoid	Perennial	Exotic	SE
CARFOE	<i>Carex foenea</i>	dry-spike sedge	Cyperaceae	Graminoid	Perennial	Native	S3?
CARFOL	<i>Carex folliculata</i>	long sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARGRA	<i>Carex gracillima</i>	graceful sedge	Cyperaceae	Graminoid	Perennial	Native	S4S5
CARINT	<i>Carex intumescens</i>	bladder sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARLUC	<i>Carex lucorum</i>	sedge	Cyperaceae	Graminoid	Perennial	Native	S4
CARLUR	<i>Carex lurida</i>	shallow sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARPAU	<i>Carex magellanica</i>	sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARNIG	<i>Carex nigra</i>	black sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARNOV	<i>Carex novae-angliae</i>	New England sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARPAL	<i>Carex pallescens</i>	pale sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARPRO	<i>Carex projecta</i>	necklace sedge	Cyperaceae	Graminoid	Perennial	Native	S4S5
CARRAD	<i>Carex radiata</i>	stellate sedge	Cyperaceae	Graminoid	Perennial	Native	S4
CARSCO	<i>Carex scoparia</i>	pointed broom sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARSP	<i>Carex sp.</i>	sedge	Cyperaceae	Graminoid	Perennial	Native	na
CARSTI	<i>Carex stipata</i>	stalk-grain sedge	Cyperaceae	Graminoid	Perennial	Native	S5

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
CARSWA	<i>Carex swanii</i>	swan sedge	Cyperaceae	Graminoid	Perennial	Native	S2?
CARTRI	<i>Carex trisperma</i>	three-seed sedge	Cyperaceae	Graminoid	Perennial	Native	S5
CARUMB	<i>Carex umbellata</i>	hidden sedge	Cyperaceae	Graminoid	Perennial	Native	S4
CENMON	<i>Centaurea montana</i>	mountain bluet	Asteraceae	Forb	Perennial	Exotic	na
CENNIG	<i>Centaurea nigra</i>	knapweed	Asteraceae	Forb	Perennial	Exotic	SE
CERVUL	<i>Cerastium fontanum ssp. vulgare</i>	mouse-ear chickweed	Caryophyllaceae	Forb	Perennial	Exotic	SE
CHASPE	<i>Chamaecyparis sp.</i>	false-cypress	Cupressaceae	Tree	Perennial	Exotic	na
CHACAL	<i>Chamaedaphne calyculata</i>	leather-leaf	Ericaceae	Shrub	Perennial	Native	S5
CHEMAJ	<i>Chelidonium majus</i>	celandine	Papaveraceae	Forb	Biennial	Exotic	SE
CHEALB	<i>Chenopodium album</i>	lamb's quarters	Chenopodiaceae	Forb	Annual	Exotic	SE
CICMAC	<i>Cicuta maculata</i>	water-hemlock	Apiaceae	Forb	Perennial	Native	S5
CIRARV	<i>Cirsium arvense</i>	Canada thistle	Asteraceae	Forb	Perennial	Exotic	SE
CIRSPE	<i>Cirsium sp.</i>	thistle	Asteraceae	Forb	Perennial	Exotic	na
CIRVUL	<i>Cirsium vulgare</i>	bull thistle	Asteraceae	Forb	Biennial	Exotic	SE
CLIBOR	<i>Clintonia borealis</i>	blue-bead lilly	Liliaceae	Forb	Perennial	Native	S5
COMPER	<i>Comptonia peregrina</i>	sweetfern	Myricaceae	Shrub	Perennial	Native	S5
CONARV	<i>Convolvulus arvensis</i>	field-bindweed	Convolvulaceae	Forb	Perennial	Exotic	SE
CONYCAN	<i>Coryza canadensis</i>	horseweed	Asteraceae	Forb	Annual	Native	S5
COPTRI	<i>Coptis trifolia</i>	goldenthread	Ranunculaceae	Forb	Perennial	Native	S5
CORALT	<i>Cornus alternifolia</i>	alternate-leaved dogwood	Cornaceae	Shrub	Perennial	Native	S5
CORCAN	<i>Cornus canadensis</i>	bunchberry	Cornaceae	Forb	Perennial	Native	S5
CORSER	<i>Cornus sericea</i>	red osier dogwood	Cornaceae	Shrub	Perennial	Native	S5
CORSPE	<i>Cornus sp.</i>	dogwood	Cornaceae	Shrub	Perennial	Exotic	na
CORDID	<i>Coronopus didymus</i>	swine-cress	Brassicaceae	Forb	Biennial, annual	Exotic	SE
CORCOR	<i>Corylus cornuta</i>	hazelnut	Betulaceae	Shrub	Perennial	Native	S5
COTSPE	<i>Cotoneaster sp.</i>	cotoneaster	Rosaceae	Shrub	Perennial	Exotic	na
CRAMON	<i>Crataegus monogyna</i>	English hawthorn	Rosaceae	Tree	Perennial	Exotic	SE
CRASP	<i>Crataegus sp.</i>	hawthorn	Rosaceae	Shrub	Perennial	na	na
CYPACA	<i>Cypripedium acaule</i>	pink lady's-slipper	Orchidaceae	Forb	Perennial	Native	S5

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
CYSFRA	<i>Cystopteris fragilis</i>	fragile fern	Dryopteridaceae	Pteridophyte	Perennial	Native	S4
CYSSP	<i>Cystopteris sp.</i>	fragile fern	Dryopteridaceae	Pteridophyte	Perennial	Native	na
DACGLO	<i>Dactylis glomerata</i>	orchard grass	Poaceae	Graminoid	Perennial	Exotic	SE
DANCOM	<i>Danthonia compressa</i>	flattened oatgrass	Poaceae	Graminoid	Perennial	Native	S4
DANSPI	<i>Danthonia spicata</i>	wire / poverty grass	Poaceae	Graminoid	Perennial	Native	S5
POTFRU	<i>Dasiphora fruticosa</i>	shrubby cinquefoil	Rosaceae	Shrub	Perennial	Exotic / native	S4
DAUCAR	<i>Daucus carota</i>	wild carrot	Asteraceae	Forb	Biennial	Exotic	SE
DENPUN	<i>Dennstaedtia punctilobula</i>	hay-scented fern	Dennstaedtiaceae	Pteridophyte	Perennial	Native	S5
DESCAEP	<i>Deschampsia cespitosa ssp. parviflora</i>	tufted hair grass	Poaceae	Graminoid	Perennial	Exotic	na
DEFLE	<i>Deschampsia flexuosa</i>	common hair grass	Poaceae	Graminoid	Perennial	Native	S5
DIARM	<i>Dianthus armeria</i>	deptford pink	Caryophyllaceae	Forb	Biennial, annual	Exotic	SE
PANLAN	<i>Dichanthelium acuminatum</i> var. <i>fasciculatum</i>	western witchgrass	Poaceae	Graminoid	Perennial	Native	S5
PANBOR	<i>Dichanthelium boreale</i>	northern witchgrass	Poaceae	Graminoid	Perennial	Native	S5
DIELON	<i>Diervilla lonicera</i>	bush-honeysuckle	Caprifoliaceae	Shrub	Perennial	Native	S5
DIGISC	<i>Digitaria ischaemum</i>	small crab grass	Poaceae	Graminoid	Annual	Exotic	SE
ASTUMB	<i>Doellingeria umbellata</i>	tall white aster	Asteraceae	Forb	Perennial	Native	S5
DROROT	<i>Drosera rotundifolia</i>	round-leaved sundew	Droseraceae	Forb	Perennial	Native	S5
DRYCAM	<i>Dryopteris campyloptera</i>	eastern spreading wood fern	Dryopteridaceae	Pteridophyte	Perennial	Native	S5
DRYCAR	<i>Dryopteris carthusiana</i>	spinulose wood fern	Dryopteridaceae	Pteridophyte	Perennial	Native	S5
DRYCRI	<i>Dryopteris cristata</i>	crested shield fern	Dryopteridaceae	Pteridophyte	Perennial	Native	S5
DRYINT	<i>Dryopteris intermedia</i>	evergreen wood fern	Dryopteridaceae	Pteridophyte	Perennial	Native	S5
ELYREP	<i>Elymus repens</i>	witch-grass	Poaceae	Graminoid	Perennial	Exotic	SE
EPIREP	<i>Epigaea repens</i>	mayflower	Ericaceae	Shrub	Perennial	Native	S5
EPIANG	<i>Epilobium angustifolium</i>	fireweed	Onagraceae	Forb	Perennial	Native	S5
EPICIL	<i>Epilobium ciliatum</i>	willow-herb	Onagraceae	Forb	Perennial	Native	S5
EPIHEL	<i>Epipactis helleborine</i>	helleborine	Orchidaceae	Forb	Perennial	Exotic	SE

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
EQUARV	<i>Equisetum arvense</i>	field-horsetail	Equisetaceae	Pteridophyte	Perennial	Native	S5
EQUSP	<i>Equisetum sp.</i>	scouring rush	Equisetaceae	Forb	Perennial	Native	na
EQUSYL	<i>Equisetum sylvaticum</i>	wood-horsetail	Equisetaceae	Pteridophyte	Perennial	Native	S5
EREHIE	<i>Erechtites hieraciifolia</i>	fireweed	Asteraceae	Forb	Annual	Native	S5
ERIANN	<i>Erigeron annuus</i>	white-top fleabane	Asteraceae	Forb	Annual	Native	S4S5
ERISTR	<i>Erigeron strigosus</i>	daisy fleabane	Asteraceae	Forb	Annual	Native	S5
ERUGAL	<i>Erucastrum gallicum</i>	dog-mustard	Brassicaceae	Forb	Biennial, annual	Exotic	SE
ERYCHE	<i>Erysimum cheiranthoides</i>	wormseed mustard	Brassicaceae	Forb	Annual	Exotic / native	S5SE
ERYHIE	<i>Erysimum hieracifolium</i>	European wallflower	Brassicaceae	Forb	Biennial, perennial	Exotic	SE
EUOSPE	<i>Euonymus sp.</i>	spindle-tree	Celastraceae	Shrub	Perennial	Exotic	na
ASTRAD	<i>Eurybia radula</i>	rough aster	Asteraceae	Forb	Perennial	Native	S5
EUTGRA	<i>Euthamia graminifolia</i>	narrow-leaved goldenrod	Asteraceae	Forb	Perennial	Native	S5
FAGGRA	<i>Fagus grandifolia</i>	American beech	Fagaceae	Tree	Perennial	Native	S5
FAGSYL	<i>Fagus sylvatica</i>	European beech	Fagaceae	Tree	Perennial	Exotic	na
FESFIL	<i>Festuca filiformis</i>	hair fescue	Poaceae	Graminoid	Perennial	Exotic	SE
FESHET	<i>Festuca heteromalla</i>	spreading fescue	Poaceae	Graminoid	Perennial	Exotic	SE
FESRUBR	<i>Festuca rubra ssp. rubra</i>	red fescue	Poaceae	Graminoid	Perennial	Exotic / native	S5
FESSP	<i>Festuca sp.</i>	fescue	Poaceae	Graminoid	Perennial	na	na
FESQVI	<i>Festuca trachyphylla</i>	sheep's fescue	Poaceae	Graminoid	Perennial	Exotic	SE
FRAANA	<i>Fragaria ananassa</i>	strawberry	Rosaceae	Forb	Perennial	Exotic	SE
FRASP	<i>Fragaria sp.</i>	strawberry	Rosaceae	Forb	Perennial	Native	S4S5
FRAVES	<i>Fragaria vesca</i>	woodland strawberry	Rosaceae	Forb	Perennial	Native	S4
FRAVIR	<i>Fragaria virginiana</i>	strawberry	Rosaceae	Forb	Perennial	Native	S5
RHAFRA	<i>Frangula alnus</i>	European alder-buckthorn	Rhamnaceae	Shrub	Perennial	Exotic	SE
FRAAME	<i>Fraxinus americana</i>	white ash	Oleaceae	Tree	Perennial	Native	S5
FRAEXC	<i>Fraxinus excelsior</i>	European ash	Oleaceae	Tree	Perennial	Exotic	SE
FRAPENS	<i>Fraxinus pennsylvanica</i> var. <i>subintegrifolia</i>	red ash	Oleaceae	Tree	Perennial	Exotic	na

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
GALTET	<i>Galeopsis tetrahit</i>	hemp-nettle	Lamiaceae	Forb	Annual	Exotic	SE
GALQUA	<i>Galinsoga quadriradiata</i>	common quickweed	Asteraceae	Forb	Annual	Exotic	SE
GALPAL	<i>Galium palustre</i>	common bedstraw	Rosaceae	Forb	Perennial	Native	S5
GALSP	<i>Galium sp.</i>	bedstraw	Rubiaceae	Forb	Unknown	na	na
GAUHIS	<i>Gaultheria hispidula</i>	creeping snowberry	Ericaceae	Shrub	Perennial	Native	S5
GAUPRO	<i>Gaultheria procumbens</i>	teaberry	Ericaceae	Shrub	Perennial	Native	S5
GAYBAC	<i>Gaylussacia baccata</i>	huckleberry	Ericaceae	Shrub	Perennial	Native	S5
GERSP	<i>Geranium sp.</i>	geranium (introduced)	Geraniaceae	Forb	Unknown	Exotic	na
GLEHED	<i>Glechoma hederacea</i>	ground-ivy	Lamiaceae	Forb	Perennial	Exotic	SE
GLYCAN	<i>Glyceria canadensis</i>	rattlesnake grass	Poaceae	Graminoid	Perennial	Native	S5
GLYGRA	<i>Glyceria grandis</i>	reed manna-grass	Poaceae	Graminoid	Perennial	Native	S4S5
GLYSTR	<i>Glyceria striata</i>	fowl manna-grass	Poaceae	Graminoid	Perennial	Native	S5
GLYLAX	<i>Glyceria X laxa</i>	northern mannagrass	Poaceae	Graminoid	Perennial	Native	S4?
GNASP	<i>Gnaphalium sp.</i>	cutweed	Asteraceae	Forb	Unknown	na	na
GNAULI	<i>Gnaphalium uliginosum</i>	low cutweed	Asteraceae	Forb	Annual	Exotic	SE
HAMVIR	<i>Hamamelis virginiana</i>	witch-hazel	Hamamelidaceae	Shrub	Perennial	Native	S5
HEDHEL	<i>Hedera helix</i>	English ivy	Apiaceae	Climber (~woody)	Perennial	Exotic	SE
HEMSP	<i>Hemerocallis sp.</i>	day-lily	Liliaceae	Forb	Perennial	Exotic	SE
HESMAT	<i>Hesperis matronalis</i>	dame's rocket	Brassicaceae	Forb	Biennial, perennial	Exotic	SE
HIEAUR	<i>Hieracium aurantiacum</i>	orange hawkweed	Asteraceae	Forb	Perennial	Exotic	SE
HIECAN	<i>Hieracium canadense</i>	Canada hawkweed	Asteraceae	Forb	Perennial	Native	S4S5
HIEFLA	<i>Hieracium flagellare</i>	whiplash hawkweed	Asteraceae	Forb	Perennial	Exotic	SE
HIEFLO	<i>Hieracium floribundum</i>	king devil	Asteraceae	Forb	Perennial	Exotic	SE
HIELAC	<i>Hieracium lachenalii</i>	common hawkweed	Asteraceae	Forb	Perennial	Exotic	SE
HIEPAN	<i>Hieracium paniculatum</i>	panicked hawkweed	Asteraceae	Forb	Perennial	Native	S3
HIEPIL	<i>Hieracium pilosella</i>	mouse-eared hawkweed	Asteraceae	Forb	Perennial	Exotic	SE
HIESCA	<i>Hieracium scabrum</i>	rough hawkweed	Asteraceae	Forb	Perennial	Native	S5
HIESP	<i>Hieracium sp.</i>	hawkweed	Asteraceae	Forb	Perennial	na	na
HOLLAN	<i>Holcus lanatus</i>	velvet-grass	Poaceae	Graminoid	Perennial	Exotic	SE
HOSSP	<i>Hosta sp.</i>	hosta	Liliaceae	Forb	Perennial	Exotic	na

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
HEDCAE	<i>Houstonia caerulea</i>	bluets	Rubiaceae	Forb	Perennial	Native	S5
HYDPANG	<i>Hydrangea paniculata</i> <i>var. grandiflora</i>	peegee hydrangea	Saxifragaceae	Shrub	Perennial	Exotic	na
HYPPER	<i>Hypericum perforatum</i>	common St. John's-wort	Clusiaceae	Forb	Perennial	Exotic	SE
HYPRAD	<i>Hypochaeris radicata</i>	cat's-ear	Asteraceae	Forb	Perennial	Exotic	SE
ILEGLA	<i>Ilex glabra</i>	inkberry	Aquifoliaceae	Shrub	Perennial	Native	S5
ILEVER	<i>Ilex verticillata</i>	Canada holly	Aquifoliaceae	Shrub	Perennial	Native	S5
IMPCAP	<i>Impatiens capensis</i>	jewel-weed	Balsaminaceae	Forb	Annual	Native	S5
IMPGLA	<i>Impatiens glandulifera</i>	policeman's helmet	Balsaminaceae	Forb	Annual	Exotic	SE
IRISP	<i>Iris sp.</i>	iris	Iridaceae	Forb	Perennial	na	na
IRISPE	<i>Iris sp.</i>	iris	Iridaceae	Forb	Perennial	Exotic	na
JUNBRE	<i>Juncus brevicaudatus</i>	narrow-panicked rush	Juncaceae	Graminoid	Perennial	Native	S5
JUNBUF	<i>Juncus bufonius</i>	toad rush	Juncaceae	Graminoid	Annual	Native	S5
JUNEFF	<i>Juncus effusus</i>	soft rush	Juncaceae	Graminoid	Perennial	Native	S5
JUNSPN	<i>Juncus sp.</i>	rush	Juncaceae	Graminoid	Perennial	Native	na
JUNTEN	<i>Juncus tenuis</i>	slender rush	Juncaceae	Graminoid	Perennial	Native	S5
JUNCOM	<i>Juniperus communis</i>	common juniper	Cupressaceae	Shrub	Perennial	Native	S5
JUNSP	<i>Juniperus sp.</i>	juniper (introduced)	Cupressaceae	Shrub	Perennial	Exotic	na
KALANG	<i>Kalmia angustifolia</i>	sheep laurel	Ericaceae	Shrub	Perennial	Native	S5
LACCAN	<i>Lactuca canadensis</i>	wild lettuce	Asteraceae	Forb	Biennial, annual	Native	S5
LAPCOM	<i>Lapsana communis</i>	nipplewort	Asteraceae	Forb	Annual	Exotic	SE
LARLAR	<i>Larix laricina</i>	eastern larch	Pinaceae	Tree	Perennial	Native	S5
LEDGRO	<i>Ledum groenlandicum</i>	Labrador-tea	Ericaceae	Shrub	Perennial	Native	S5
LEOAUT	<i>Leontodon autumnalis</i>	fall dandelion	Asteraceae	Forb	Perennial	Exotic	SE
LEPVIR	<i>Lepidium virginicum</i>	peppergrass	Brassicaceae	Forb	Biennial, annual	Exotic	SE
CHRLEU	<i>Leucanthemum vulgare</i>	ox-eye daisy	Asteraceae	Forb	Perennial	Exotic	SE
LINVUL	<i>Linaria vulgaris</i>	butter-and-eggs	Scrophulariaceae	Forb	Perennial	Exotic	SE
LINBOR	<i>Linnaea borealis</i>	twinflower	Caprifoliaceae	Forb	Perennial	Native	S5
LOLPER	<i>Lolium perenne</i>	perennial rye-grass	Poaceae	Graminoid	Perennial	Exotic	SE
FESPRE	<i>Lolium pratense</i>	meadow fescue	Poaceae	Graminoid	Perennial	Exotic	SE
LONCAN	<i>Lonicera canadensis</i>	fly-honeysuckle	Caprifoliaceae	Shrub	Perennial	Native	S5

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
LONSP	<i>Lonicera sp.</i>	honeysuckle	Caprifoliaceae	Shrub	Perennial	Exotic	na
LOTOR	<i>Lotus corniculatus</i>	birdsfoot-trefoil	Fabaceae	Forb	Perennial	Exotic	SE
LUNANN	<i>Lunaria annua</i>	honesty	Brassicaceae	Forb	Annual	Exotic	SE
LUPPOL	<i>Lupinus polyphyllus</i>	garden lupine	Fabaceae	Forb	Perennial	Exotic	SE
LUZLUZ	<i>Luzula luzuloides</i>	forest wood-rush	Juncaceae	Graminoid	Perennial	Exotic	SE
LUZMUL	<i>Luzula multiflora</i>	common woodrush	Juncaceae	Graminoid	Perennial	Native	S5
LUZSP	<i>Luzula sp.</i>	woodrush	Juncaceae	Graminoid	Perennial	na	na
LYCCLA	<i>Lycopodium clavatum</i>	running pine	Lycopodiaceae	Pteridophyte	Perennial	Native	S5
LYCOBS	<i>Lycopodium obscurum</i>	ground pine	Lycopodiaceae	Pteridophyte	Perennial	Native	S5
DIPSP	<i>Lycopodium sp.</i>	club-moss	Lycopodiaceae	Pteridophyte	Perennial	Native	na
DIPTRI	<i>Lycopodium tristachyum</i>	ground-cedar	Lycopodiaceae	Pteridophyte	Perennial	Native	S4
LYCUNI	<i>Lycopus uniflorus</i>	bugle-weed	Lamiaceae	Forb	Perennial	Native	S5
LYSNUM	<i>Lysimachia nummularia</i>	moneywort	Primulaceae	Forb	Perennial	Exotic	SE
LYSTER	<i>Lysimachia terrestris</i>	loosestrife	Primulaceae	Forb	Perennial	Native	S5
LYTSAL	<i>Lythrum salicaria</i>	purple loosestrife	Lythraceae	Forb	Perennial	Exotic	SE
MAICAN	<i>Maianthemum canadense</i>	false wild-lily-of-the-valley	Liliaceae	Forb	Perennial	Native	S5
SMIRAC	<i>Maianthemum racemosum</i>	false solomon's seal	Liliaceae	Forb	Perennial	Native	S4S5
PYRMAL	<i>Malus sp.</i>	apple	Rosaceae	Tree	Perennial	Exotic	SE
MALMOS	<i>Malva moschata</i>	musk-mallow	Malvaceae	Forb	Perennial	Exotic	SE
MATMAT	<i>Matricaria discoidea</i>	pineapple-weed	Asteraceae	Forb	Annual	Exotic	SE
MATSTR	<i>Matteuccia struthiopteris</i>	ostrich fern	Dryopteridaceae	Pteridophyte	Perennial	Native	S5
MEDVIR	<i>Medeola virginiana</i>	indian cucumber-root	Liliaceae	Forb	Perennial	Native	S5
MEDLUP	<i>Medicago lupulina</i>	black medick	Fabaceae	Forb	Biennial, annual	Exotic	SE
MEDSAT	<i>Medicago sativa</i>	alfalfa	Fabaceae	Forb	Perennial	Exotic	SE
MELLIN	<i>Melampyrum lineare</i>	cow-wheat	Scrophulariaceae	Forb	Annual	Native	S5
MELOFF	<i>Melilotus officinalis</i>	white / yellow sweet clover	Fabaceae	Forb	Biennial, annual, perennial	Exotic	SE
MELSPE	<i>Melilotus sp.</i>	sweet clover (introduced)	Fabaceae	Forb	Biennial, annual, perennial	Exotic	na

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MENARV	<i>Mentha arvensis</i>	field mint	Lamiaceae	Forb	Perennial	Exotic / native	S5
MITREP	<i>Mitchella repens</i>	partridge-berry	Rubiaceae	Forb	Perennial	Native	S5
MONHYP	<i>Monotropa hypopithys</i>	pine-sap	Monotropaceae	Forb	Perennial	Native	S4
MONUNI	<i>Monotropa uniflora</i>	indian pipe	Monotropaceae	Forb	Perennial	Native	S5
MYOARV	<i>Myosotis arvensis</i>	rough forget-me-not	Boraginaceae	Forb	Biennial, annual	Exotic	SE
MYOSYL	<i>Myosotis sylvatica</i>	garden forget-me-not	Boraginaceae	Forb	Perennial	Exotic	SE
MYRPEN	<i>Myrica pensylvanica</i>	bayberry	Myricaceae	Shrub	Perennial	Native	S5
MYRODO	<i>Myrrhis odorata</i>	sweet cicely	Apiaceae	Forb	Perennial	Exotic	na
NEMMUC	<i>Nemopanthus mucronata</i>	false holly	Aquifoliaceae	Shrub	Perennial	Native	S5
ASTACU	<i>Oclemena acuminata</i>	wood aster	Asteraceae	Forb	Perennial	Native	S5
OENBIE	<i>Oenothera biennis</i>	evening-primrose	Onagraceae	Forb	Biennial, perennial	Native	S5
OENPER	<i>Oenothera perennis</i>	sundrops	Onagraceae	Forb	Perennial	Native	S5
ONOUSEN	<i>Onoclea sensibilis</i>	sensitive fern	Dryopteridaceae	Pteridophyte	Perennial	Native	S5
ORYASP	<i>Oryzopsis asperifolia</i>	rice-grass	Poaceae	Graminoid	Perennial	Native	S4
OSMCIN	<i>Osmunda cinnamomea</i>	cinnamon fern	Osmundaceae	Pteridophyte	Perennial	Native	S5
OSMCLA	<i>Osmunda claytoniana</i>	interrupted fern	Osmundaceae	Pteridophyte	Perennial	Native	S5
OSMREG	<i>Osmunda regalis</i>	royal fern	Osmundaceae	Pteridophyte	Perennial	Native	S5
OXAAACE	<i>Oxalis montana</i>	wood-sorrel	Oxalidaceae	Forb	Perennial	Native	S5
OXASTR	<i>Oxalis stricta</i>	yellow wood-sorrel	Oxalidaceae	Forb	Perennial	Native	S5
PANSP	<i>Panicum sp.</i>	panic-grass	Poaceae	Graminoid	Unknown	na	na
PARQUI	<i>Parthenocissus quinquefolia</i>	Virginia creeper	Vitaceae	Climber (~woody)	Perennial	Exotic	SE
PHAARU	<i>Phalaris arundinacea</i>	reed canary-grass	Poaceae	Graminoid	Perennial	Exotic / native	S5
PHAARUV	<i>Phalaris arundinacea forma. variegata</i>	ribbon grass	Poaceae	Graminoid	Perennial	Exotic	na
PHECON	<i>Phegopteris connectilis</i>	northern beech fern	Thelypteridaceae	Pteridophyte	Perennial	Native	S5
PHLPRA	<i>Phleum pratense</i>	timothy	Poaceae	Graminoid	Perennial	Exotic	SE
PHLPAN	<i>Phlox paniculata</i>	garden phlox	Polemoniaceae	Forb	Perennial	Exotic	SE
PHLSUB	<i>Phlox subulata</i>	moss-pink	Polemoniaceae	Forb	Perennial	Exotic	SE
AROMEL	<i>Photinia melanocarpa</i>	black chokeberry	Rosaceae	Shrub	Perennial	Native	S5

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
AROARB	<i>Photinia pyrifolia</i>	red chokeberry	Rosaceae	Shrub	Perennial	Native	S4S5
AROSP	<i>Photinia sp.</i>	chokeberry	Rosaceae	Shrub	Perennial	Native	S5
PICEXC	<i>Picea abies</i>	norway spruce	Pinaceae	Tree	Perennial	Exotic	SE
PICGLA	<i>Picea glauca</i>	white spruce	Pinaceae	Tree	Perennial	Native	S5
PICMAR	<i>Picea mariana</i>	black spruce	Pinaceae	Tree	Perennial	Native	S5
PICPUN	<i>Picea pungens</i>	Colorado / blue spruce	Pinaceae	Tree	Perennial	Exotic	na
PICRUB	<i>Picea rubens</i>	red spruce	Pinaceae	Tree	Perennial	Native	S5
PICSPN	<i>Picea sp.</i>	red or black spruce	Pinaceae	Tree	Perennial	Native	na
PINNIG	<i>Pinus nigra</i>	black pine	Pinaceae	Tree	Perennial	Exotic	na
PINRES	<i>Pinus resinosa</i>	red pine	Pinaceae	Tree	Perennial	Native	S4S5
PINSP	<i>Pinus sp.</i>	pine	Pinaceae	Tree	Perennial	na	na
PINSTR	<i>Pinus strobus</i>	white pine	Pinaceae	Tree	Perennial	Native	S5
PINSYL	<i>Pinus sylvestris</i>	Scotch pine	Pinaceae	Tree	Perennial	Exotic	SE
PLALAN	<i>Plantago lanceolata</i>	narrow-leaved plantain	Plantaginaceae	Forb	Perennial	Exotic	SE
PLAMAJ	<i>Plantago major</i>	common plantain	Plantaginaceae	Forb	Perennial	Exotic / native	SE
POAANN	<i>Poa annua</i>	annual spear-grass	Poaceae	Graminoid	Annual	Exotic	SE
POACOM	<i>Poa compressa</i>	canada bluegrass	Poaceae	Graminoid	Perennial	Exotic	SE
POANEM	<i>Poa nemoralis</i>	wood bluegrass	Poaceae	Graminoid	Perennial	Exotic	SE
POAPAL	<i>Poa palustris</i>	fowl meadow grass	Poaceae	Graminoid	Perennial	Native	S5
POAPRA	<i>Poa pratensis</i>	Kentucky bluegrass	Poaceae	Graminoid	Perennial	Exotic / native	S5SE
POASP	<i>Poa sp</i>	bluegrass	Poaceae	Graminoid	Perennial	na	na
POATRI	<i>Poa trivialis</i>	rough-stalked bluegrass	Poaceae	Graminoid	Perennial	Exotic	SE
POLBIF	<i>Polygonatum biflorum</i>	smooth solomons seal	Liliaceae	Forb	Perennial	Exotic	SE
POLARE	<i>Polygonum arenastrum</i>	oval-leaf knotweed	Polygonaceae	Forb	Annual	Exotic / native	S5SE
POLAVI	<i>Polygonum aviculare</i>	knotweed	Polygonaceae	Forb	Annual	Exotic / native	S5SE
POLCIL	<i>Polygonum cilinode</i>	fringed black bindweed	Polygonaceae	Forb	Perennial	Native	S5
POLCON	<i>Polygonum convolvulus</i>	wild buckwheat	Polygonaceae	Forb	Perennial	Exotic	SE
POLCUS	<i>Polygonum cuspidatum</i>	Japanese knotweed	Polygonaceae	Forb	Perennial	Exotic	SE

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
POLPER	<i>Polygonum persicaria</i>	lady's thumb	Polygonaceae	Forb	Annual	Exotic	SE
POLSP	<i>Polygonum sp.</i>	bindweed	Polygonaceae	Forb	Unknown	na	na
POLVIR	<i>Polypodium virginianum</i>	rock polypody	Polypodiaceae	Pteridophyte	Perennial	Native	S5
POPALB	<i>Populus alba</i>	white poplar	Salicaceae	Tree	Perennial	Exotic	SE
POPBAL	<i>Populus balsamifera</i>	balsam poplar	Salicaceae	Tree	Perennial	Exotic / native	S4
POPGRA	<i>Populus grandidentata</i>	large-toothed aspen	Salicaceae	Tree	Perennial	Native	S5
POPTRE	<i>Populus tremuloides</i>	trembling aspen	Salicaceae	Tree	Perennial	Native	S5
POTANG	<i>Potentilla anglica</i>	English cinquefoil	Rosaceae	Forb	Perennial	Exotic	SE
POTARG	<i>Potentilla argentea</i>	silvery cinquefoil	Rosaceae	Forb	Perennial	Exotic	SE
POTNOR	<i>Potentilla norvegica</i>	rough cinquefoil	Rosaceae	Forb	Biennial, annual, perennial	Exotic / native	S5
POTSIM	<i>Potentilla simplex</i>	cinquefoil	Rosaceae	Forb	Perennial	Native	S5
POTSP	<i>Potentilla sp.</i>	cinquefoil	Rosaceae	Forb	Perennial	na	na
PREALT	<i>Prenanthes altissima</i>	tall rattlesnake-root	Asteraceae	Forb	Perennial	Native	S4S5
PRETRI	<i>Prenanthes trifoliata</i>	lion's-paw	Asteraceae	Forb	Perennial	Native	S5
PRUVUL	<i>Prunella vulgaris</i>	heal-all	Lamiaceae	Forb	Perennial	Exotic / native	S5SE
PRECER	<i>Prunus cerasus</i>	sour red cherry	Rosaceae	Tree	Perennial	Exotic	SE
PRUPEN	<i>Prunus pensylvanica</i>	pin-cherry	Rosaceae	Tree	Perennial	Native	S5
PRUSP	<i>Prunus sp.</i>	cherry	Rosaceae	Tree	Perennial	na	na
PRUSPE	<i>Prunus sp.</i>	cherry	Rosaceae	Tree	Perennial	Exotic	na
PRUVIR	<i>Prunus virginiana</i>	choke-cherry	Rosaceae	Tree	Perennial	Native	S5
PTEAQU	<i>Pteridium aquilinum</i>	bracken	Dennstaedtiaceae	Pteridophyte	Perennial	Native	S5
PYRELL	<i>Pyrola elliptica</i>	shinleaf	Pyrolaceae	Forb	Perennial	Native	S5
PYRSP	<i>Pyrola sp.</i>	pyrola sp.	Pyrolaceae	Forb	Perennial	Native	na
QUEROBU	<i>Quercus robur</i>	English oak	Fagaceae	Tree	Perennial	Exotic	SE
QUEROB	<i>Quercus rubra</i>	red oak	Fagaceae	Tree	Perennial	Native	S5
RANACR	<i>Ranunculus acris</i>	tall buttercup	Ranunculaceae	Forb	Perennial	Exotic	SE
RANREP	<i>Ranunculus repens</i>	creeping buttercup	Ranunculaceae	Forb	Perennial	Exotic	SE
RAPRAP	<i>Raphanus raphanistrum</i>	wild radish	Brassicaceae	Forb	Annual	Exotic	SE
RHACAT	<i>Rhamnus cathartica</i>	common buckthorn	Rhamnaceae	Shrub	Perennial	Exotic	SE

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
RHESPE	<i>Rheum sp.</i>	rhubarb	Polygonaceae	Forb	Perennial	Exotic	na
RHOCAN	<i>Rhododendron canadense</i>	rhodora	Ericaceae	Shrub	Perennial	Native	S5
RHOSPE	<i>Rhododendron sp.</i>	rhododendron	Ericaceae	Shrub	Perennial	Exotic	na
RHOYED	<i>Rhododendron yedoense</i>	rhododendron	Ericaceae	Shrub	Perennial	Exotic	na
RIBSAT	<i>Ribes rubrum</i>	red currant	Grossulariaceae	Shrub	Perennial	Exotic	SE
RIBSP	<i>Ribes sp.</i>	ribes	Grossulariaceae	Shrub	Perennial	na	na
RORPAL	<i>Rorippa palustris ssp. fernaldiana</i>	marsh cress	Brassicaceae	Forb	Biennial, annual, perennial	na	S?
ROSCAR	<i>Rosa carolina</i>	wild rose	Rosaceae	Subshrub	Perennial	Native	S4S5
ROSMUL	<i>Rosa multiflora</i>	multiflora rose	Rosaceae	Climber (~woody)	Perennial	Exotic	SE
ROSNIT	<i>Rosa nitida</i>	swamp-rose	Rosaceae	Subshrub	Perennial	Native	S4
ROSRUG	<i>Rosa rugosa</i>	rugose rose	Rosaceae	Subshrub	Perennial	Exotic	SE
ROSSP	<i>Rosa sp.</i>	rose (unknown)	Rosaceae	Subshrub	Perennial	na	na
ROSVIR	<i>Rosa virginiana</i>	common wild rose	Rosaceae	Subshrub	Perennial	Native	S5
RUBALL	<i>Rubus allegheniensis</i>	common blackberry	Rosaceae	Subshrub	Perennial	Native	S5
RUBCAN	<i>Rubus canadensis</i>	smooth blackberry	Rosaceae	Subshrub	Perennial	Native	S5
RUBHIS	<i>Rubus hispida</i>	bristly dewberry	Rosaceae	Subshrub	Perennial	Native	S5
RUBIDA	<i>Rubus idaeus ssp. strigosus</i>	red raspberry	Rosaceae	Subshrub	Perennial	Native	S5
RUBODO	<i>Rubus odoratus</i>	flowering raspberry	Rosaceae	Subshrub	Perennial	Exotic	SE
RUBREC	<i>Rubus recurvicaulis</i>	Blanchard's dewberry	Rosaceae	Subshrub	Perennial	Native	S?
RUBSET	<i>Rubus setosus</i>	small bristleberry	Rosaceae	Subshrub	Perennial	Native	S4?
RUBSP	<i>Rubus sp.</i>	blackberry	Rosaceae	Subshrub	Perennial	Native	na
RUDHIR	<i>Rudbeckia hirta</i>	black-eyed susan	Asteraceae	Forb	Biennial, perennial	Exotic	SE
RUMACET	<i>Rumex acetosa</i>	garden-sorrel	Polygonaceae	Forb	Perennial	Exotic	SE
RUMACE	<i>Rumex acetosella</i>	sheep-sorrel	Polygonaceae	Forb	Perennial	Exotic	SE
RUMCRI	<i>Rumex crispus</i>	curled dock	Polygonaceae	Forb	Perennial	Exotic	SE
RUMSP	<i>Rumex sp.</i>	dock	Polygonaceae	Forb	Unknown	na	na
SAGPRO	<i>Sagina procumbens</i>	pearlwort	Caryophyllaceae	Forb	Perennial	Exotic / native	S5SE
SALSP	<i>Salix sp.</i>	willow	Salicaceae	Tree	Perennial	na	na

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
SALSP	<i>Salix sp.</i>	willow	Salicaceae	Tree	Perennial	Exotic	na
SAMRAC	<i>Sambucus racemosa</i>	red-berried elder	Caprifoliaceae	Shrub	Perennial	Native	S5
SAMSP	<i>Sambucus sp.</i>	elder sp.	Caprifoliaceae	Shrub	Perennial	Native	S5
SARPUR	<i>Sarracenia purpurea</i>	pitcher-plant	Sarraceniaceae	Forb	Perennial	Native	S5
SCICYP	<i>Scirpus cyperinus</i>	cottongrass bulrush	Cyperaceae	Graminoid	Perennial	Native	S5
SCIMC	<i>Scirpus microcarpus</i>	small-fruit bulrush	Cyperaceae	Graminoid	Perennial	Native	S5
SCISP	<i>Scirpus sp.</i>	bulrush	Cyperaceae	Graminoid	Perennial	Native	na
SEDSPU	<i>Sedum spurium</i>	garden stonecrop	Crassulaceae	Forb	Perennial	Exotic	SE
SENJAC	<i>Senecio jacobaea</i>	tansy ragwort	Asteraceae	Forb	Biennial, perennial	Exotic	SE
SENSP	<i>Senecio sp.</i>	groundsel	Asteraceae	Forb	Unknown	na	na
SENSYL	<i>Senecio sylvaticus</i>	woodland groundsel	Asteraceae	Forb	Annual	Exotic	SE
SENVIS	<i>Senecio viscosus</i>	clammy groundsel	Asteraceae	Forb	Annual	Exotic	SE
SENVUL	<i>Senecio vulgaris</i>	common groundsel	Asteraceae	Forb	Annual	Exotic	SE
SETGLA	<i>Setaria glauca</i>	yellow foxtail	Poaceae	Graminoid	Annual	Exotic	SE
POTTRI	<i>Sibbaldiopsis tridentata</i>	three-toothed cinquefoil	Rosaceae	Forb	Perennial	Native	S5
SISOFF	<i>Sisymbrium officinale</i>	hedge mustard	Brassicaceae	Forb	Annual	Exotic	SE
SISMON	<i>Sisyrinchium montanum</i>	blue-eyed grass	Iridaceae	Forb	Perennial	Native	S5
SOLDUL	<i>Solanum dulcamara</i>	bittersweet	Solanaceae	Climber (~woody)	Perennial	Exotic	SE
SOLCAN	<i>Solidago canadensis</i>	Canada goldenrod	Asteraceae	Forb	Perennial	Native	S5
SOLJUN	<i>Solidago juncea</i>	early goldenrod	Asteraceae	Forb	Perennial	Native	S5
SOLPUB	<i>Solidago puberula</i>	rough goldenrod	Asteraceae	Forb	Perennial	Native	S5
SOLRUG	<i>Solidago rugosa</i>	rough goldenrod	Asteraceae	Forb	Perennial	Native	S5
SOLSPN	<i>Solidago sp.</i>	goldenrod	Asteraceae	Forb	Perennial	Native	S5
SONARV	<i>Sonchus arvensis</i>	perennial sow-thistle	Asteraceae	Forb	Perennial	Exotic	SE
SONASP	<i>Sonchus asper</i>	spiny sow-thistle	Asteraceae	Forb	Annual	Exotic	SE
SONOLE	<i>Sonchus oleraceus</i>	annual sow-thistle	Asteraceae	Forb	Annual	Exotic	SE
SONSPE	<i>Sonchus sp.</i>	sow-thistle	Asteraceae	Forb	Unknown	Exotic	SE
SORSOR	<i>Sorbaria sorbifolia</i>	false spiraea	Rosaceae	Shrub	Perennial	Exotic	SE
SORAME	<i>Sorbus americana</i>	mountain-ash	Rosaceae	Tree	Perennial	Native	S5
SORAUC	<i>Sorbus aucuparia</i>	European mountain-ash	Rosaceae	Tree	Perennial	Exotic	SE

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
SPERUB	<i>Spergularia rubra</i>	purple sandspurry	Caryophyllaceae	Forb	Annual and perennial	Exotic	SE
SPIALB	<i>Spiraea alba</i>	meadow-sweet	Rosaceae	Shrub	Perennial	Native	S5
SPISPE	<i>Spiraea sp.</i>	spirea	Rosaceae	Shrub	Perennial	Exotic	na
SPITOM	<i>Spiraea tomentosa</i>	steeplebush	Rosaceae	Shrub	Perennial	Native	S5
SPIALC	<i>Spiranthes lacera</i>	northern slender ladies'-tresses	Orchidaceae	Forb	Perennial	Native	S5
STEGRA	<i>Stellaria graminea</i>	grass-leaved stitchwort	Caryophyllaceae	Forb	Perennial	Exotic	SE
STEMED	<i>Stellaria media</i>	common starwort	Caryophyllaceae	Forb	Annual	Exotic	SE
STRROS	<i>Streptopus lanceolatus</i>	rosy twisted stalk	Liliaceae	Forb	Perennial	Native	S5
SYMALBL	<i>Symphoricarpos albus</i> var. <i>laevigatus</i>	snowberry	Caprifoliaceae	Shrub	Perennial	Exotic	SE
ASTLAN	<i>Symphyotrichum lanceolatum</i>	white-panickled aster	Asteraceae	Forb	Perennial	Native	S4S5
ASTLAT	<i>Symphyotrichum lateriflorum</i>	aster	Asteraceae	Forb	Perennial	Native	S5
ASTNOV	<i>Symphyotrichum novibelgii</i>	New York aster	Asteraceae	Forb	Perennial	Native	S5
ASTPUN	<i>Symphyotrichum puniceum</i>	swamp aster	Asteraceae	Forb	Perennial	Native	S5
SYRSP	<i>Syringa vulgaris</i>	common lilac	Oleaceae	Shrub	Perennial	Exotic	SE
TANVUL	<i>Tanacetum vulgare</i>	tansy	Asteraceae	Forb	Perennial	Exotic	SE
TAROFF	<i>Taraxacum officinale</i>	dandelion	Asteraceae	Forb	Perennial	Exotic	SE
TAXSPE	<i>Taxus sp.</i>	yew	Taxaceae	Shrub	Perennial	Exotic	na
THENOV	<i>Thelypteris noveboracensis</i>	New York fern	Thelypteridaceae	Pteridophyte	Perennial	Native	S5
THAARV	<i>Thlaspi arvense</i>	penny-cress	Brassicaceae	Forb	Annual	Exotic	SE
THUOCCE	<i>Thuja occidentalis</i>	northern white cedar	Cupressaceae	Tree	Perennial	Exotic / native	S1S2
TILEUR	<i>Tilia x vulgaris</i>	basswood	Tiliaceae	Tree	Perennial	Exotic	SE
TRAPRA	<i>Tragopogon pratensis</i>	goat's-beard	Asteraceae	Forb	Biennial	Exotic	SE
TRIVIR	<i>Triadenum virginicum</i>	marsh St. John's-wort	Clusiaceae	Forb	Perennial	Native	S4S5
TRIBOR	<i>Trientalis borealis</i>	starflower	Primulaceae	Forb	Perennial	Native	S5
TRIARV	<i>Trifolium arvense</i>	rabbitfoot clover	Fabaceae	Forb	Annual	Exotic	SE

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
TRIAUR	<i>Trifolium aureum</i>	yellow clover	Fabaceae	Forb	Biennial, annual	Exotic	SE
TRICAM	<i>Trifolium campestre</i>	low hop clover	Fabaceae	Forb	Biennial, annual	Exotic	SE
TRIHVB	<i>Trifolium hybridum</i>	alsike clover	Fabaceae	Forb	Perennial / annual	Exotic	SE
TRIPRA	<i>Trifolium pratense</i>	red clover	Fabaceae	Forb	Biennial, perennial	Exotic	SE
TRIREF	<i>Trifolium repens</i>	white clover	Fabaceae	Forb	Perennial	Exotic	SE
TRIUND	<i>Trillium undulatum</i>	painted trillium	Liliaceae	Forb	Perennial	Native	S5
MATMAR	<i>Tripleurospermum maritima</i>	mayweed / scentless chamomile	Asteraceae	Forb	Annual	Exotic	SE
TSUCAN	<i>Tsuga canadensis</i>	eastern hemlock	Pinaceae	Tree	Perennial	Native	S4S5
TUSFAR	<i>Tussilago farfara</i>	coltsfoot	Asteraceae	Forb	Perennial	Exotic	SE
TYPLAT	<i>Typha latifolia</i>	broad-leaved cat-tail	Typhaceae	Graminoid	Perennial	Native	S5
ULMAME	<i>Ulmus americana</i>	American elm	Ulmaceae	Tree	Perennial	Exotic / native	S4
ULMGLA	<i>Ulmus glabra</i>	Scotch elm	Ulmaceae	Tree	Perennial	Exotic	SE
ULMPRO	<i>Ulmus procera</i>	English elm	Ulmaceae	Tree	Perennial	Exotic	SE
VACANG	<i>Vaccinium angustifolium</i>	lowbush blueberry	Ericaceae	Shrub	Perennial	Native	S5
VACMYR	<i>Vaccinium myrtilloides</i>	velvet-leaf blueberry	Ericaceae	Shrub	Perennial	Native	S5
VACVIT	<i>Vaccinium vitis-idaea</i>	mountain cranberry	Ericaceae	Shrub	Perennial	Native	S5
VERARV	<i>Veronica arvensis</i>	field-speedwell	Scrophulariaceae	Forb	Annual	Exotic	SE
VERCHA	<i>Veronica chamaedrys</i>	birds-eye	Scrophulariaceae	Forb	Perennial	Exotic	SE
VERLON	<i>Veronica longifolia</i>	garden speedwell	Scrophulariaceae	Forb	Perennial	Exotic	SE
VEROFF	<i>Veronica officinalis</i>	common speedwell	Scrophulariaceae	Forb	Perennial	Exotic / native	S5SE
VERPER	<i>Veronica persica</i>	bird's-eye speedwell	Scrophulariaceae	Forb	Annual	Exotic	SE
VERSERS	<i>Veronica serpyllifolia</i> ssp. <i>serpyllifolia</i>	thyme-leaved speedwell	Scrophulariaceae	Forb	Perennial	Exotic	SE
VIBALN	<i>Viburnum lantanoides</i>	hobble-bush	Caprifoliaceae	Shrub	Perennial	Native	S5
VIBNUD	<i>Viburnum nudum</i>	wild raisin	Caprifoliaceae	Shrub	Perennial	Native	S5
VICCRA	<i>Vicia cracca</i>	tufted vetch	Fabaceae	Forb	Perennial	Exotic	SE
VIOLAN	<i>Viola lanceolata</i>	lance-leaved violet	Violaceae	Forb	Perennial	Native	S5
VIOMAC	<i>Viola macloskeyi</i>	small white violet	Violaceae	Forb	Perennial	Native	S5
VIOSAG	<i>Viola sagittata</i>	arrow-leaved violet	Violaceae	Forb	Perennial	Native	S3S4

Sp. code	Scientific name	Common name	Family	Growth form	Life history	Status	CDC S-rank
VIO SP	<i>Viola sp.</i>	violet	Violaceae	Forb	Unknown	na	na
VIO TRI	<i>Viola tricolor</i>	pansy	Violaceae	Forb	Annual	Exotic	SE
WEIFLO	<i>Weigela florida</i>	weigela	Caprifoliaceae	Shrub	Perennial	Exotic	na

Appendix B: Non-vascular taxa encountered in study area (includes species encountered in study plots and other observations of the ground vegetation)

Sp. code	Scientific name	Common name	Family
Bryophytes			
ATRSP	<i>Atrichum sp.</i>	Atrichum moss	Polytrichaceae
AULPAL	<i>Aulacomnium palustre</i>	Aulacomnium moss	Aulacomniaceae
AULSP	<i>Aulacomnium sp.</i>	Aulacomnium moss	Aulacomniaceae
BAZTRI	<i>Bazzania trilobata</i>	three-lobed bazzania	Lepidoziaceae
BRASP	<i>Brachythecium sp.</i>	Brachythecium moss	Brachytheciaceae
BRYSP	<i>Bryum sp.</i>	Bryum moss	Bryaceae
CALLHAL	<i>Callicladium haldanianum</i>	Callicladium moss	Hypnaceae
CALLSP	<i>Callicladium sp.</i>	Callicladium moss	Hypnaceae
CALSP	<i>Calypogeia sp.</i>	Calypogeia liverwort	Calypogeiaceae
CERPUR	<i>Ceratodon purpureus</i>	Ceratodon moss	Ditrichaceae
CERSP	<i>Ceratodon sp.</i>	Ceratodon moss	Ditrichaceae
CLIDEN	<i>Climacium dendroides</i>	Climacium moss	Climaciaceae
CLISP	<i>Climacium sp.</i>	Climacium moss	Climaciaceae
DICRSP	<i>Dicranella sp.</i>	Dicranella moss	Dicranaceae
DICFLA	<i>Dicranum flagellare</i>	Dicranum moss	Dicranaceae
DICFUS	<i>Dicranum fuscens</i>	Dicranum moss	Dicranaceae
DICMAJ	<i>Dicranum majus</i>	Dicranum moss	Dicranaceae
DICMON	<i>Dicranum montanum</i>	Dicranum moss	Dicranaceae
DICPOL	<i>Dicranum polysetum</i>	Dicranum moss	Dicranaceae
DICSCO	<i>Dicranum scoparium</i>	broom moss	Dicranaceae
DICSP	<i>Dicranum sp.</i>	Dicranum moss	Dicranaceae
DRESP	<i>Drepanocladus sp.</i>	Drepanocladus moss	Amblystegiaceae
DREUNC	<i>Drepanocladus uncinatus</i>	Drepanocladus moss	Amblystegiaceae
HEDCIL	<i>Hedwigia ciliata</i>	Hedwigia moss	Hedwigiaceae
HEDSP	<i>Hedwigia sp.</i>	Hedwigia moss	Hedwigiaceae
HERSP	<i>Herzogiella sp.</i>	Herzogiella moss	Hypnaceae
HYLSP	<i>Hylocomium sp.</i>	stair-step moss	Hylocomiaceae
HYLSPL	<i>Hylocomium splendens</i>	stair-step moss	Hylocomiaceae
HYPIMP	<i>Hypnum imponens</i>	Hypnum moss	Hypnaceae
HYPSP	<i>Hypnum sp.</i>	Hypnum moss	Hypnaceae
LEPREP	<i>Lepidozia repens</i>	Lepidozia liverwort	Lepidoziaceae
LEPSP	<i>Lepidozia sp.</i>	Lepidozia liverwort	Lepidoziaceae
LUEGLA	<i>Leucobryum glaucum</i>	pin-cushion moss	Leucobryaceae
MNISP	<i>Mnium sp.</i>	Mnium moss	Mniaceae
ODODEN	<i>Odontoschisma demudatum</i>	Odontoschisma liverwort	Calypogeiaceae
ODOSP	<i>Odontoschisma sp.</i>	Odontoschisma liverwort	Adelanthaceae
PLASP	<i>Plagiomnium sp.</i>	Plagiomnium moss	Mniaceae
PLESCH	<i>Pleurozium schreberi</i>	schreber's moss	Entodontaceae
POHNUT	<i>Pohlia nutans</i>	Pohlia moss	Bryaceae
POHSP	<i>Pohlia sp.</i>	Pohlia moss	Bryaceae
POLCOM	<i>Polytrichum commune var perigoniale</i>	hair-cap moss	Polytrichaceae
POLJUN	<i>Polytrichum juniperinum</i>	hair-cap moss	Polytrichaceae
POLOHI	<i>Polytrichum ohioense or formosum</i>	hair-cap moss	Polytrichaceae

Sp. code	Scientific name	Common name	Family
POLYSP	<i>Polytrichum sp.</i>	hair-cap moss	Polytrichaceae
PTISP	<i>Ptilidium sp.</i>	Ptilidium liverwort	Ptilidiaceae
RHYSP	<i>Rhytidiadelphus sp.</i>	Rhytidiadelphus moss	Rhytidiaceae
SPHGIR	<i>Sphagnum girgensohnii</i>	peatmoss	Sphagnaceae
SPHSP	<i>Sphagnum sp.</i>	peatmoss	Sphagnaceae
TETPEL	<i>Tetraphis pellucida</i>	Tetraphis moss	Tetraphidaceae
TETANG	<i>Tetraplodon angustatus</i>	Tetraplodon moss	Splachnaceae
THUDEL	<i>Thuidium delicatulum</i>	Thuidium moss	Thuidiaceae
THUSP	<i>Thuidium sp.</i>	Thuidium moss	Thuidiaceae
ULOCRI	<i>Ulota crispa</i>	Ulota moss	Orthotrichaceae
Lichens			
BRYNIT	<i>Bryoria nitidula</i>	tundra horsehair lichen	Parmeliaceae
CLAARB	<i>Cladonia arbuscula</i>	reindeer lichen	Cladoniaceae
CLAMIT	<i>Cladonia mitis</i>	green reindeer lichen	Cladoniaceae
CLARAN	<i>Cladonia rangiferina</i>	grey reindeer lichen	Cladoniaceae
CLASP	<i>Cladonia sp.</i>	reindeer lichen	Cladoniaceae
CLASTE	<i>Cladonia stellaris</i>	star-tipped reindeer lichen	Cladoniaceae
CLADCAE	<i>Cladonia caespiticia</i>	stubby-stalked cladonia	Cladoniaceae
CLADCAR	<i>Cladonia carneola</i>	crowned pixie-cup	Cladoniaceae
CLADCEN	<i>Cladonia cenotea</i>	powdered funnel lichen	Cladoniaceae
CLADCHL	<i>Cladonia chlorophaea</i>	mealy pixie-cup	Cladoniaceae
CLADCON	<i>Cladonia coniocraea</i>	common powderhorn	Cladoniaceae
CLADCRI	<i>Cladonia crispata</i>	organ-pipe lichen	Cladoniaceae
CLADCRI	<i>Cladonia cristatella</i>	british soldiers	Cladoniaceae
CLADDIG	<i>Cladonia digitata</i>	finger pixie-cup	Cladoniaceae
CLADFIM	<i>Cladonia fimbriata</i>	trumpet lichen	Cladoniaceae
CLADFUR	<i>Cladonia furcata</i>	many-forked cladonia	Cladoniaceae
CLADGRA	<i>Cladonia gracilis ssp. Gracilis</i>	smooth cladonia	Cladoniaceae
CLADMAC	<i>Cladonia macilenta</i>	lipstick powderhorn	Cladoniaceae
CLADMAX	<i>Cladonia maxima</i>	giant cladonia	Cladoniaceae
CLADOCH	<i>Cladonia ochrochlora</i>	smooth-footed powderhorn	Cladoniaceae
CLADPHY	<i>Cladonia phyllophora</i>	felt cladonia	Cladoniaceae
CLADPLE	<i>Cladonia pleurota</i>	red-fruited pixie-cup	Cladoniaceae
CLADPYX	<i>Cladonia pyxidata</i>	pebbled pixie-cup	Cladoniaceae
CLADSCA	<i>Cladonia scabriuscula</i>	mealy forked cladonia	Cladoniaceae
CLADSP	<i>Cladonia sp.</i>	lichen	Cladoniaceae
CLADSQU	<i>Cladonia squamosa</i>	dragon cladonia	Cladoniaceae
CLADUNC	<i>Cladonia uncialis</i>	thorn cladonia	Cladoniaceae

Appendix C: Plot information

Plot #	Park	Easting*	Northing*	Plot type
0	Admiral's Cove	448089	4951697	Undisturbed forest
1	Admiral's Cove	448271	4951453	Moderately-disturbed forest
2	Admiral's Cove	448282	4952215	Moderately-disturbed forest
3	Admiral's Cove	448318	4951950	Undisturbed forest
4	Admiral's Cove	448315	4951741	Undisturbed forest
5	Admiral's Cove	448213	4951703	Undisturbed forest
6	Admiral's Cove	448406	4951677	Undisturbed forest
7	Admiral's Cove	448626	4952107	Undisturbed forest
8	Admiral's Cove	448295	4952057	Moderately-disturbed forest
9	Admiral's Cove	448462	4951996	Moderately-disturbed forest
10	Admiral's Cove	448589	4952165	Moderately-disturbed forest
11	Admiral's Cove	448037	4951597	Undisturbed forest
12	Admiral's Cove	448217	4952208	Undisturbed forest
13	Admiral's Cove	448037	4952280	Undisturbed forest
14	Admiral's Cove	448361	4952042	Moderately-disturbed forest
15	Admiral's Cove	448397	4951797	Undisturbed forest
16	Alder Piper	459164	4947364	Undisturbed forest
17	Alder Piper	459225	4947400	Undisturbed forest
18	Alder Piper	459134	4947356	Undisturbed forest
19	Alder Piper	459192	4947357	Undisturbed forest
20	Alder Piper	459172	4947399	Undisturbed forest
21	Alder Piper	459255	4947428	Undisturbed forest
22	Alder Piper	459106	4947377	Derelict
23	Alder Piper	459210	4947389	Derelict
25	Alder Piper	459038	4947364	Derelict
26	Arnold Whitworth	456544	4945072	Undisturbed forest
28	Arnold Whitworth	456528	4945149	Horticultural
29	Arnold Whitworth	456471	4945222	Undisturbed forest
30	Barrington St.	454814	4942708	Horticultural
31	Cogswell	453036	4944024	Horticultural
32	Cogswell	452995	4944065	Horticultural
33	Cogswell	453026	4944054	Horticultural
34	Cogswell	453058	4944010	Horticultural
35	Conrose	452401	4942925	Horticultural
36	Conrose	452358	4942972	Intensive recreation
39	Conrose	452382	4943020	Horticultural
40	Conrose	452276	4942989	Horticultural

Plot #	Park	Easting*	Northing*	Plot type
41	Cyril Smith	454759	4948935	Undisturbed forest
42	Cyril Smith	454576	4948786	Undisturbed forest
43	Cyril Smith	454402	4948964	Undisturbed forest
44	Cyril Smith	454475	4948738	Undisturbed forest
45	Cyril Smith	454341	4948984	Undisturbed forest
46	Cyril Smith	455043	4948915	Tertiary forest
47	Cyril Smith	454554	4948529	Undisturbed forest
48	Cyril Smith	454702	4948989	Tertiary forest
49	Cyril Smith	454500	4948698	Undisturbed forest
50	Cyril Smith	454321	4948813	Undisturbed forest
51	Cyril Smith	454398	4948895	Undisturbed forest
52	Cyril Smith	454342	4949070	Undisturbed forest
53	Cyril Smith	454633	4948994	Tertiary forest
54	Cyril Smith	454881	4948970	Undisturbed forest
55	Cyril Smith	454610	4948827	Undisturbed forest
56	Cyril Smith	454852	4948928	Undisturbed forest
57	Fort Needham	452353	4946088	Tertiary forest
58	Fort Needham	452412	4945797	Horticultural
59	Fort Needham	452506	4945733	Horticultural
60	Fort Needham	452313	4945988	Tertiary forest
61	Fort Needham	452369	4946057	Tertiary forest
62	Fort Needham	452447	4945756	Horticultural
63	Fort Needham	452272	4945981	Tertiary forest
64	Fort Needham	452463	4945730	Horticultural
65	Fort Needham	452349	4945974	Horticultural
67	Fort Needham	452282	4945914	Horticultural
68	Fort Needham	452428	4945878	Intensive recreation
69	Fort Needham	452435	4945937	Horticultural
71	Fort Needham	452345	4945939	Horticultural
72	Fuller Terrace	452817	4945111	Horticultural
74	Glenbourne	446476	4947231	Undisturbed forest
75	Glenbourne	446442	4947223	Undisturbed forest
76	Glenbourne	446336	4947271	Moderately-disturbed forest
77	Glenbourne	446367	4947231	Moderately-disturbed forest
78	Glenbourne	446404	4947203	Undisturbed forest
79	Glenbourne	446291	4947283	Highly-disturbed forest
80	Glenbourne	446325	4947247	Moderately-disturbed forest
81	Glenbourne	446292	4947413	Moderately-disturbed forest
82	Glenbourne	446399	4947250	Intensive recreation

Plot #	Park	Easting*	Northing*	Plot type
83	Glenbourne	446348	4947409	Intensive recreation
84	Glenbourne	446407	4947272	Intensive recreation
86	Glenbourne	446427	4947267	Derelict
87	Glenbourne	446303	4947335	Derelict
88	Hemlock Ravine	447260	4948633	Undisturbed forest
89	Hemlock Ravine	447191	4948350	Moderately-disturbed forest
90	Hemlock Ravine	447254	4948727	Highly-disturbed forest
91	Hemlock Ravine	447011	4949132	Undisturbed forest
92	Hemlock Ravine	446690	4948627	Undisturbed forest
93	Hemlock Ravine	446645	4948922	Undisturbed forest
94	Hemlock Ravine	446429	4948474	Undisturbed forest
95	Hemlock Ravine	446885	4948692	Undisturbed forest
96	Hemlock Ravine	447248	4949333	Undisturbed forest
97	Hemlock Ravine	447015	4948711	Undisturbed forest
98	Hemlock Ravine	446683	4948452	Undisturbed forest
99	Hemlock Ravine	447420	4948825	Undisturbed forest
100	Hemlock Ravine	446497	4948674	Undisturbed forest
101	Hemlock Ravine	446510	4948450	Undisturbed forest
102	Hemlock Ravine	446778	4948317	Undisturbed forest
103	Hemlock Ravine	446924	4948372	Undisturbed forest
104	Hemlock Ravine	446828	4948961	Highly-disturbed forest
105	Hemlock Ravine	447052	4949184	Undisturbed forest
106	Hemlock Ravine	447053	4949185	Undisturbed forest
107	Hemlock Ravine	446461	4948481	Undisturbed forest
108	Lincoln Cross	448017	4945898	Undisturbed forest
109	Lincoln Cross	448024	4945878	Undisturbed forest
110	Lincoln Cross	448043	4945909	Horticultural
111	Point Pleasant	454957	4941737	Undisturbed forest
112	Point Pleasant	455089	4941629	Undisturbed forest
113	Point Pleasant	454479	4940963	Undisturbed forest
114	Point Pleasant	455093	4941443	Undisturbed forest
115	Point Pleasant	454581	4941523	Undisturbed forest
116	Point Pleasant	454818	4940918	Highly-disturbed forest
117	Point Pleasant	454697	4941135	Highly-disturbed forest
118	Point Pleasant	454742	4941155	Moderately-disturbed forest
119	Point Pleasant	454987	4941692	Highly-disturbed Forest
120	Point Pleasant	454624	4940974	Highly-disturbed forest
121	Point Pleasant	454646	4941305	Highly-disturbed forest
122	Point Pleasant	455098	4941212	Highly-disturbed Forest

Plot #	Park	Easting*	Northing*	Plot type
123	Point Pleasant	455176	4940989	Highly-disturbed Forest
124	Point Pleasant	454672	4941452	Highly-disturbed forest
125	Point Pleasant	455019	4941724	Undisturbed forest
126	Point Pleasant	455052	4940973	Highly-disturbed Forest
127	Point Pleasant	454744	4941274	Moderately-disturbed forest
128	Point Pleasant	454714	4941437	Highly-disturbed Forest
129	Point Pleasant	455167	4941546	Moderately-disturbed forest
130	Point Pleasant	454882	4941566	Moderately-disturbed forest
132	Point Pleasant	455003	4940663	Horticultural
133	Point Pleasant	455321	4941140	Horticultural
134	Point Pleasant	454764	4941123	Horticultural
135	Point Pleasant	455246	4940855	Horticultural
136	Point Pleasant	455144	4940803	Horticultural
139	Randall Avenue	449344	4945236	Undisturbed forest
140	Randall Avenue	449290	4945231	Undisturbed forest
141	Randall Avenue	449322	4945218	Horticultural
142	Randall Avenue	449311	4945246	Horticultural
143	Remington Court	446922	4947056	Moderately-disturbed forest
144	Remington Court	446922	4947018	Undisturbed forest
145	Remington Court	446898	4947043	Undisturbed forest
146	Remington Court	446848	4947013	Undisturbed forest
147	Remington Court	446910	4947095	Undisturbed forest
148	Remington Court	446892	4947003	Moderately-disturbed forest
150	Remington Court	446857	4946994	Horticultural
151	Seaview	451113	4947089	Horticultural
152	Seaview	450894	4946895	Derelict
154	Seaview	450994	4947054	Horticultural
155	Seaview	451004	4946973	Horticultural
158	Seaview	450920	4946999	Horticultural
159	Titus Smith	449609	4945503	Horticultural
160	Titus Smith	449671	4345511	Horticultural
162	Titus Smith	449641	4945516	Horticultural
163	Titus Smith	449642	4945466	Horticultural
164	Tremount Plateau	448203	4946970	Undisturbed forest
165	Tremount Plateau	447959	4946789	Undisturbed forest
166	Tremount Plateau	448146	4946913	Undisturbed forest
167	Tremount Plateau	447959	4946789	Moderately-disturbed forest
168	Tremount Plateau	447868	4946822	Undisturbed forest
169	Tremount Plateau	448086	4947057	Undisturbed forest

Plot #	Park	Easting*	Northing*	Plot type
170	Tremount Plateau	448159	4947001	Undisturbed forest
171	Tremount Plateau	447861	4946885	Undisturbed forest
172	Tremount Plateau	448014	4947058	Undisturbed forest
173	Tremount Plateau	447844	4946916	Undisturbed forest
174	Tremount Plateau	448089	4947002	Moderately-disturbed forest
175	Tremount Plateau	448191	4946926	Undisturbed forest
176	Tremount Plateau	447913	4946843	Horticultural
177	Tremount Plateau	448058	4947005	Derelict
180	Tremount Plateau	447984	4946859	Horticultural
181	Tremount Plateau	448009	4946936	Intensive recreation
182	Uplands	441390	4952638	Horticultural
183	Uplands	441471	4952630	Intensive recreation
184	Uplands	441465	4952686	Horticultural
187	Willett St.	447939	4945237	Derelict
188	Willett St.	447959	4945180	Undisturbed forest
189	Willett St.	447937	4945137	Highly-disturbed forest
190	Willett St.	447989	4945216	Undisturbed forest
191	Willett St.	447846	4945213	Moderately-disturbed forest
192	Willett St.	447915	4945183	Undisturbed forest
193	Willett St.	447881	4945069	Undisturbed forest
194	Willett St.	447840	4945135	Undisturbed forest
195	Willett St.	447847	4945163	Undisturbed forest
196	Willett St.	448024	4945245	Moderately-disturbed forest
197	Willett St.	447887	4945158	Undisturbed forest
198	Young / Kaye	452534	4945605	Horticultural
199	Young / Kaye	452515	4945583	Horticultural
200	Bell Lake	459871	4946615	Undisturbed forest
201	Wedgewood	446812	4947564	Horticultural
202	Bell Lake	459672	4946715	Undisturbed forest
203	Bell Lake	459530	4947229	Undisturbed forest
204	Bell Lake	447989	4945216	Undisturbed forest
205	Bell Lake	459487	4947280	Undisturbed forest
206	Bell Lake	459527	4947265	Undisturbed forest
207	Bell Lake	459650	4946692	Undisturbed forest
208	Bell Lake	459459	4947307	Undisturbed forest
209	Bell Lake	459843	4946591	Undisturbed forest
210	Bell Lake	459578	4947173	Undisturbed forest
211	Bell Lake	459376	4947231	Highly-disturbed forest
212	Montebello	457062	4949949	Intensive recreation

Plot #	Park	Easting*	Northing*	Plot type
213	Montebello	457002	4949880	Horticultural
214	Montebello	457015	4949986	Intensive recreation
215	Wedgewood	446916	4947517	Moderately-disturbed forest
216	Wedgewood	446828	4947625	Undisturbed forest
217	Wedgewood	446951	4947566	Undisturbed forest
218	Wedgewood	446881	4947610	Moderately-disturbed forest
219	Wedgewood	446831	4947658	Highly-disturbed forest
220	Wedgewood	446788	4947679	Undisturbed forest
221	Wedgewood	446928	4947552	Undisturbed forest
222	Bell Lake	459311	4947057	Undisturbed forest
223	Bell Lake	459429	4947033	Derelict
224	Bell Lake	459936	4946540	Undisturbed forest
225	Bell Lake	459860	4946550	Undisturbed forest
226	Bell Lake	459521	4946818	Highly-disturbed forest
227	Montebello	456994	4950018	Horticultural
228	Wedgewood	446865	4947529	Undisturbed forest
229	Point Pleasant	455090	4941334	Horticultural
E1	Hemlock Ravine	447254	4948238	Boundary
E10	Alder Piper	459179	4947360	Boundary
E11	Bell Lake	459754	4946618	Boundary
E12	Wedgewood	446853	4947592	Boundary
E13	Point Pleasant	454717	4941681	Boundary
E15	Point Pleasant	454362	4941535	Boundary
E16	Glenbourne	446428	4947208	Boundary
E17	Point Pleasant	454517	4941390	Boundary
E18	Cyril Smith	454887	4949060	Boundary
E19	Bell Lake	459441	4947056	Boundary
E2	Hemlock Ravine	447014	4948336	Boundary
E20	Bell Lake	459446	4947349	Boundary
E21	Willett St.	448006	4945188	Boundary
E22	Willett St.	447833	4945228	Boundary
E23	Hemlock Ravine	446745	4948380	Boundary
E3	Admiral's Cove	448397	4952294	Boundary
E4	Admiral's Cove	448058	4952316	Boundary
E5	Admiral's Cove	448198	4951921	Boundary
E6	Cyril Smith	454626	4948454	Boundary
E7	Cyril Smith	454547	4948315	Boundary
E8	Tremount Plateau	448035	4946868	Boundary
E9	Tremount Plateau	448103	4946948	Boundary

Plot #	Park	Easting*	Northing*	Plot type
H1	Hemlock Ravine	447342	4948486	Tertiary forest
H2	Point Pleasant	454926	4940963	Tertiary forest
H3	Point Pleasant	454930	4940892	Tertiary forest
H5	Point Pleasant	455150	4941314	Tertiary forest
R1	Hemlock Ravine	446806	4948581	Reference
R10	Point Pleasant	455081	4940957	Reference
R11	Hemlock Ravine	446765	4948678	Reference
R12	Hemlock Ravine	447082	4948911	Reference
R2	Admiral's Cove	448370	4952134	Reference
R3	Admiral's Cove	448376	4951909	Reference
R4	Admiral's Cove	448165	4951644	Reference
R5	Cyril Smith	454726	4948935	Reference
R6	Cyril Smith	454809	4948969	Reference
R7	Cyril Smith	454385	4948741	Reference
R8	Point Pleasant	455122	4940990	Reference
R9	Point Pleasant	454590	4941051	Reference
T1	Hemlock Ravine	447349	4948476	Trail 50-100m
T10	Tremount Plateau	448189	4946958	Trail 10-50m
T11	Tremount Plateau	448166	4946945	Trail 50-100m
T12	Tremount Plateau	448021	4946828	Trail 10-50m
T13	Alder Piper	459175	4947379	Trail 10-50m
T14	Wedgewood	446872	4947513	Trail 10-50m
T15	Point Pleasant	454709	4941581	Trail 50-100m
T16	Point Pleasant	454512	4941443	Trail 10-50m
T17	Remington Court	446883	4947073	Trail 10-50m
T18	Glenbourne	446341	4947258	Trail 10-50m
T19	Point Pleasant	454506	4941549	Trail 10-50m
T2	Hemlock Ravine	447375	4948448	Trail 10-50m
T20	Point Pleasant	455104	4941375	Trail 10-50m
T21	Cyril Smith	454494	4948662	Trail 10-50m
T22	Cyril Smith	454430	4948657	Trail 50-100m
T23	Cyril Smith	454420	4948686	Trail >100m
T24	Cyril Smith	454461	4948567	Trail 50-100m
T25	Cyril Smith	454417	4948769	Trail >100m
T26	Cyril Smith	454365	4948675	Trail >100m
T27	Cyril Smith	454418	4948656	Trail 50-100m
T28	Point Pleasant	455139	4940926	Trail 50-100m
T29	Point Pleasant	454908	4941647	Trail >100m
T3	Hemlock Ravine	447213	4948160	Trail 10-50m

Plot #	Park	Easting*	Northing*	Plot type
T30	Point Pleasant	454575	4941159	Trail 50-100m
T31	Point Pleasant	454614	4941019	Trail >100m
T32	Point Pleasant	454647	4941127	Trail >100m
T33	Willett St.	447970	4945228	Trail 10-50m
T34	Willett St.	447885	4945186	Trail 50-100m
T35	Willett St.	447868	4945175	Trail 50-100m
T36	Hemlock Ravine	447232	4948377	Trail 50-100m
T37	Hemlock Ravine	447145	4948326	Trail 50-100m
T38	Hemlock Ravine	446832	4948507	Trail >100m
T39	Hemlock Ravine	446964	4948689	Trail >100m
T4	Hemlock Ravine	447080	4948310	Trail 10-50m
T40	Hemlock Ravine	447154	4948881	Trail >100m
T41	Willett St.	447910	4945212	Trail 10-50m
T5	Hemlock Ravine	447103	4948327	Trail 10-50m
T6	Hemlock Ravine	447018	4948393	Trail 10-50m
T7	Hemlock Ravine	447026	4948440	Trail 50-100m
T8	Cyril Smith	454606	4948444	Trail 10-50m
T9	Cyril Smith	454430	4948270	Trail 10-50m

*NAD 83, Zone 20

Appendix D: Plot information available on compact disc

Information Available on CD

Location

- Photos
- GPS coordinates
- Park maps showing plot locations

Management and fragmentation

- Habitat type
- Distance and aspect to closest trail and forest boundary
- Trail width
- Adjacent boundary habitat
- Evidence of past land use

Environmental

- Site slope and aspect
- Rock outcrop class
- Surface stoniness class
- Microtopography class
- Slope position
- Seepage class
- % cover of bare soil, needles, leaves, duff

Vegetation composition and structure

- Tree composition and structure (species, DBH, basal area, and canopy cover)
- Understory composition and structure (species, strata, percent cover)
- Species info (nomenclature, family, CDC S-rank, life history, growth form, biogeographical status, PFG)
- Community diversity (species richness, species diversity)
- Coarse woody debris and snags (diameter, basal area, species, decay class)
- Percent cover of strata (non-vascular, herb, shrub, canopy)
- Exotic cover, dominance, and richness
- Vegetation type (from NSDNR's Forest Ecosystem Classification)
