Searching for climate change solutions in Newfoundland's urban forests

by © Erin Pearson

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Abstract

Urban forests provide municipalities with a suite of benefits and services, and help mitigate the effects of climate change notably, serving to reduce the impact of local warming and urban heat island effects. The gradient of winter climates that exists on the island of Newfoundland (NL) provides a unique natural laboratory to investigate the influence of urban forest canopy cover on both winter and summer temperature changes, and assess the potential for the island to act as a model for future forest conditions throughout mainland Canada. Using this natural laboratory, we characterize the impact of NL urban forests on summer temperatures; however, microclimatic influences on winter appear to be dominated by other factors (e.g., wind exposure). Structural and ecological characteristics of urban forests changed throughout the urban ecosystem, influencing how closely different urban forest regions resemble natural forests, and amplifying the need for alternative indicators of ecological integrity that better reflect their sociocultural influences. This research provides important empirical findings on the relationship between urban forests and urban ecosystems in NL, highlighting the need for more intentional management and planning to ensure the longevity of urban forests and their benefits into the future.

Keywords: urban forest, ecological integrity, microclimate, climate change, urban ecosystems, green infrastructure, temperature mitigation

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I would like to respectfully acknowledge the territory in which this study took place as the ancestral homelands of the Beothuk and the Mi'kmaw.

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Co-Authorship Statement

Dr. Carissa Brown and Dr. Joel Finnis are co-authors on all chapters of this thesis. As the primary author, I have been the primary researcher for the literature review, designing the research proposal, logistical project planning, fieldwork, data analysis, and manuscript preparation. All parts of the project were done in collaboration with Carissa Brown and Joel Finnis, who contributed to project design and conceptualization, data analysis, and manuscript preparations.

Chapter 1: Introduction and thesis overview

1.1 Introduction

1.1.1 Urban forests and climate change

The urban forest is often broadly defined as all trees in a city (Rowntree 1984), though this definition does not incorporate the social and ecological aspects associated with urban forests (FAO 2016; Ordóñez Barona *et al.* 2022). Variations in the definition of urban forests also exist for management and planning purposes between cities (Konijnendijk *et al.* 2006; Ordóñez and Duinker 2014), with some incorporating all trees within their municipal boundary, while others only focus on trees located on public property. A more comprehensive definition for the urban forest would be all tree-dominated systems, both natural or planted, in an urban area along with their associated ecological and social contexts (e.g., relationships and connections to surrounding ecosystems, management and placement of trees, public perception; Rowntree 1998; Konijnendijk *et al.* 2006; FAO 2016; Ordóñez Barona *et al.* 2022). This definition would therefore include all remnants of natural forests, managed stands and parks, street and residential trees, and individual stems throughout the area (Nowak 1993; Ordóñez and Duinker 2014; FAO 2016; Ordóñez Barona *et al.* 2022). As the backbone of green infrastructure (FAO 2016), many cities throughout North America are expanding their urban forests to take advantage of the benefits and services they provide, including the mitigation of climate change impacts (e.g. City of Toronto 2008; City of New York 2011).

Urban forests provide a variety of ecosystem services, defined as goods and services that directly or indirectly contribute to the well-being of human populations (Costanza *et al.* 1992; MEA 2005). The most studied urban forest ecosystem services include: improvements to mental and physical health through reduced air and noise pollution (Nowak *et al.* 2006; FAO 2016;

Mueller *et al.* 2022), promotion of healthy lifestyles, and stress reduction (Kardan *et al.* 2015; FAO 2016); improvement of water quality and stormwater management through reduced runoff, water purification, and increased levels of interception, infiltration, and evapotranspiration of precipitation and stormwater (Ordóñez and Duinker 2012; Kardan *et al.* 2015; FAO 2016; Orta-Ortiz and Geneletti 2022). Other ecosystem services provided by urban forests include economic and aesthetic benefits (Kardan *et al.* 2015), sociocultural connections (e.g., recreation, social cohesion, education opportunities; FAO 2016), lower rates of violent and property crime (Venter *et al.* 2022) and the mitigation of climate change impacts.

It should be noted that urban trees can also create 'disservices', defined as the absence or reduction of services, economic loss, or negative impacts (Conway and Yip 2016; Roman *et al.* 2020a) to residents and the overall ecosystem. Commonly reported disservices resulting from urban forests include increased allergens, pest outbreaks, tree falls, and root damage (Conway and Yip 2016; Roman *et al.* 2020b; Pataki *et al.* 2021). Though disservices are generally outweighed by the positive ecosystem services, they often have a significant influence on public perception and management decisions in urban forest settings. Although not the primary focus of the current study, it is important to note that mismanagement and/or decreasing health of urban forests have the potential to exacerbate key disservices, such as the risk of tree falls.

Urban areas are one of the major contributors to climate change globally (McCarthy *et al.* 2010), producing more than 70% of global carbon dioxide emissions - as well as significant amounts of other greenhouse gasses - while occupying only 2% of the earth's surface (FAO 2016). There is considerable evidence that urban forests help mitigate the current and future impacts of ongoing climate change (Strohbach *et al.* 2012), making urban forests and green infrastructure a main focus of climate change strategies in urban planning and development. In addition to mitigation of stormwater runoff and improving air quality, urban forest ecosystem services assist in climate change mitigation through: acting as large carbon sinks by storing carbon in their vegetation and soil (Strohbach *et al.* 2012; FAO 2016); reducing the impact of extreme weather and floods by providing wind breaks and increased uptake and infiltration of water (Akbari *et al.* 2001; Roy *et al.* 2012); and mitigating the urban heat island effect while decreasing energy consumption by regulating temperatures and humidity (Grey and Deneke 1978; Akbari *et al.* 2001; Roy *et al.* 2012; FAO 2016). Human well-being in urban spaces benefits from the ecosystem services that are provided by urban forests, and these ecosystem services are dependent on increasing and restoring the functionality of urban forests.

1.1.2 Ecological integrity as a management tool

Ecological integrity is defined as the proper functioning and wholeness of an ecosystem in comparison to its natural state (Angermeier and Karr 1994; Parrish *et al.* 2003) so that when it is subject to a disturbance, the system has the capability to recover and return to an end-state that would naturally occur in the ecosystem (Regier 1993). As human populations continue to depend on the extraction of natural resources, implementing principles of sustainability becomes increasingly important in maintaining the features that ecosystems contain, and functions they serve (Woodley *et al.* 1993). Ecological integrity, therefore, is derived from a main goal of human-environment relationships: to understand ecological processes and rectify our ethics of maintaining a proper relationship with the environment. In this way, ecological integrity is widely applied to management practices for both short- and long-term ecosystem management (Munn 1993), including its use in the UNESCO World Heritage Convention (UNESCO 1988) and the Canadian National Parks Act (Parks Canada 2000). In fact, the Canadian National Parks Act identifies that maintenance of ecological integrity is the first priority of National Parks in

Canada, and prioritizes its use a tool for monitoring changes to Parks' ecosystems (Woodley 1993; Parks Canada 2000).

Ecological integrity can be applied to a wide variety of spatial scales and levels of organization in an ecosystem using a number of methods to compare and evaluate indicators of ecological data (e.g., water quality, biomass, species diversity; Steedman and Haider 1993). Though issues can arise in determining which indicators of a system's state are important, and how they relate to other variables (Keddy *et al.* 1993). Assessments of ecological integrity are carried out as either direct numerical comparisons of data within a set standard or guideline, or as empirical comparisons of attributes on a regional scale using a healthy, relatively undisturbed control system (Steedman and Haider 1993). According to Keddy *et al.* (1993) effective indicators of the ecological integrity of ecosystems should be:

- Ecologically meaningful (contribute to the maintenance of environmental processes),
- Macro scale (i.e., community level instead of an individual level),
- General (can be measured on a variety of community types)
- Sensitive (can quickly respond to stresses), and
- Simple (easy to measure).

In natural, intact, complex forest systems, a clear understanding of the dominant ecological processes that dictate the structure and function of the system is required to design methods of assessing its ecological integrity (Rempel *et al.* 2016). Indicators of ecological integrity in forests are often related to aspects of diversity and habitat quality in order to maintain diverse native species populations, genetic diversity, and ecological representativeness (Ordóñez and Duinker 2012; Rempel *et al.* 2016; Steenberg *et al.* 2019). While assessments of ecological integrity often do not provide a simple pass/fail evaluation, and ecosystem management that incorporates ecological integrity cannot fully replicate a natural system, ecological integrity can be used as a tool to improve forest management (Woodley *et al.* 1993). Assessing the ecological integrity of a system can be used to provide insights into where management actions need improvement to meet ecological objectives, identify higher priority areas for management, and identify areas that require more specific research to understand deviations from the system's natural state.

1.1.3 Ecological integrity in urban forests

Using ecological integrity as a comparison to a system's natural state (Regier 1993; Angermeier and Karr 1994; Parrish *et al.* 2003), urban forests can seem completely devoid of integrity as human-nature interactions influence the structure and functions of urban ecosystems (McDonnell *et al.* 2008; Ordóñez and Duinker 2012). In fact, in these urban systems, restoring to pre-urbanized states is often entirely unfeasible and undesirable (Alberti 2010; Steenberg *et al.* 2019). Urban forests are also heavily influenced by legacy effects: since trees are fundamentally long-lived organisms, urban forests are a function of management decisions from decades prior (Steen-Adams *et al.* 2015; Roman *et al.* 2018). Therefore, an appropriate ecological integrity assessment of urban forests must incorporate the complex socio-ecological dynamics and related values (e.g., ecological, social, cultural) of human-nature landscapes (Ordóñez and Duinker 2012) and their time-lagged effects (Steen-Adams *et al.* 2015; Roman *et al.* 2018). Ecological processes that influence the structure and function of more natural ecosystems are often replaced by social processes in urban landscapes (e.g., species preferences, land use, available growing space; Pickett *et al.* 2011; Steenberg *et al.* 2019). Similarly, Differentiated measures of integrity based on an urban-centre to peri-urban forest gradient, therefore, should act as the health indices for urban forests (Ordóñez and Duinker 2012).

The urban forest is subject to natural and anthropogenic stressors that can reduce the ecological integrity and lifespan of trees (Westra *et al.* 2012; Ordóñez and Duinker 2012; Nock *et al.* 2013). These stressors that are present in urban areas (e.g., low soil quality, limited water resources, urban heat island effect, air pollution; Sieghardt *et al.* 2005) have a significant impact on the ecological integrity of urban forests, including overall tree losses, reduced lifespans, and structural damage (Ordóñez and Duinker 2012). Human populations also play a significant role in the ecological integrity of urban forests (Alberti 2010). It is intuitive that as the population of an urban area increases in size, anthropogenic stressors that negatively impact ecological integrity also increase. This was confirmed by Pan *et al.* 2021, who found that the ecological integrity of the urban forest is lowered with increasing population size and population density in the Yangtze River Economic Belt in China, which accounts for 21% of the country's population. Similarly, human perceptions of urban forests have a large influence on management decisions, which in turn impact the structure and ecological functions of urban forests (FAO 2016).

While anthropogenic stress is a major source of ecological integrity loss in urban forests, some of the most devastating losses in North America are a result of extreme and abnormal weather events (e.g., hurricanes, extratropical storms, drought, landslides, altered precipitation patterns; Burley *et al.* 2008; Ordóñez and Duinker 2012). Climate change is further exacerbating these issues by increasing the frequency and intensity of extreme weather events. Wind disturbances – in both small-scale 'gap-dynamic' and broad-scale 'catastrophic disturbance' (Oliver 1980; Foster and Boose 1992) forms – cause damage and mortality to trees through breakage of limbs or stems, stem upheaval, and soil damage (Burley *et al.* 2008). While most disturbances from wind in Atlantic Canada are small-scale (Burley *et al.* 2008), the increasing frequency of severe wind events will likely result in more large-scale 'catastrophic disturbances' and increased environmental stress to trees in both natural and urban settings.

Shifts in temperature regimes are an additional source of environmental stress for trees. Chilling (cool temperatures between autumn and late winter) and forcing (warm temperatures between late winter and early spring) have a greater control over spring phenology than photoperiod (Flynn and Wolkovich 2018). This causes changes in spring phenology patterns from shifting winter temperatures (Flynn and Wolkovich 2018; Ettinger *et al.* 2020), which would affect ecosystem services that are reliant on phenology, and increase ecosystem stress. These temperature shifts and asynchrony in phenology can create ideal circumstances for invasive non-native species that outcompete and further compound stress to established trees (Davidson *et al.* 2011; Schlaepfer *et al.* 2011; Ordóñez and Duinker 2015; Esperon-Rodriguez *et al.* 2020). Freeze-thaw cycles also impact spring phenology patterns, cause damage from ice buildup, and increase the risk and occurrence of embolism, limiting the health and growth of trees (Mayr *et al.* 2003; Sperry and Sullivan 2020; Tedla *et al.* 2020).

The impacts to the urban forest from weather events result in higher levels of stress, further compounding the lower levels of ecological integrity of urban forests in anthropogenic environments (Westra *et al.* 2012). The study of ecological integrity in urban forests continues to grow as we realize the importance of understanding how these complex ecosystems will adapt to and survive ongoing climate change.

1.1.4 Urban forest climate amelioration services

Urban areas are subject to the urban heat island phenomenon, where the dark surfaces and reduced vegetation associated with urban spaces increase the absorption of solar energy, thereby increasing temperatures (Oke 1982; IPCC 2001; Akbari *et al.* 2001; Joshi and Joshi

2015). Within urban areas temperatures can be up to 2.5°C higher than surrounding rural areas, and both empirical and anecdotal evidence suggests temperatures within urban spaces are continuing to rise, with many northern hemisphere cities continuing to set record-high summer temperatures (Akbari *et al.* 2001; Ziter *et al.* 2019). The net effect of this phenomenon is that residents living in urban areas are at increasing risk from high temperatures and intense heat waves (Patz *et al.* 2005; Tan *et al.* 2010; Ziter *et al.* 2019).

Through a variety of natural processes, urban forests regulate the microclimate of urban spaces, mitigate the impacts of urban heat islands, and reduce air temperatures (Rowntree 1986; Pataki *et al.* 2021): green spaces increase a city's overall albedo; interception of solar radiation prevents heating of the ground and surrounding buildings; solar radiation is absorbed through photosynthesis; shade trees intercept sunlight; and evapotranspiration cools the air and increases humidity (Akbari *et al.* 2001; Joshi and Joshi 2015). While the foliage required for many of the climate services provided by trees (i.e., evapotranspiration and shade) is significantly reduced in the winter months, urban forests have also been found to assist with regulating winter temperatures and reducing energy use for heating services (Akbari and Taha 1992; Akbari *et al.* 2001). Trees reduce wind speeds, creating an insulating effect under the canopy, and direct or divert prevailing winds (Akbari and Taha 1992; Akbari *et al.* 2001; Joshi and Joshi 2015). The loss of foliage by many trees in the winter months reduces the level of shade provided by the urban forest, which allows more sunlight to heat the ground surface and buildings (Akbari and Taha 1992; Akbari and Konopacki 2004).

Climate mitigation and amelioration services, however, are not linear across the urban landscape. Instead, they are a function of land cover types, level of development, and canopy cover (Akbari *et al.* 2001; Ziter *et al.* 2019; Pataki *et al.* 2021; Jung *et al.* 2021). Areas with dense tree cover (e.g., >40% canopy cover) are more likely to shade higher amounts of impervious surfaces, and therefore have the highest potential for creating a cooler microclimate (Joshi and Joshi 2015; Ziter *et al.* 2019; Pataki *et al.* 2021).

1.1.5 Climate patterns on the island of Newfoundland

The island of Newfoundland (hereafter referred to as Newfoundland) is an ideal natural laboratory in which to examine relationships between urban forests and the urban landscape in the context of climate change. Due to the proximity to both cold (Labrador current) and warm (gulf stream) ocean currents and the North American storm track, Newfoundland experiences highly variable and active weather (e.g. Banfield and Jacobs 1998). Major urban centres on the island exist on a climate gradient (Table 1.1), where St. John's – on the east coast – experiences relatively mild winter temperatures with frequent high wind days (>34 knots), while Corner Brook – on the west coast – experiences cooler winter temperatures with fewer high wind days.

Table 1.1: Winter (December, January, February) climate variables of three urban centres in Newfoundland between 1981 and 2010 (data extracted from the Environment and Climate Change Canada Historical Climate Data website (https://climate.weather.gc.ca/climate_normals_index_e.html) on March 27, 2021).

	Corner Brook	Gander	St. John's
Mean Daily Temperature (°C)	-4.9	-5.5	-3.2
Maximum Daily Temperature (°C)	-1.51	-1.91	-0.02
Minimum Daily Temperature (°C)	-8.23	-9.08	-6.46
Daily Temperature Range (°C)	6.72	7.17	6.44
Number of Days with High Wind (>34 knots)	22	N/A	108

While the effects of climate change in Canada vary, overall the urbanized south of the country is expected to experience warmer winter temperatures, increased freeze-thaw cycles, and prolonged mid-winter thaws (Lemmen *et al.* 2008). Using daily maximum temperatures as a proxy for the frequency of freeze-thaw events, future climate scenarios that predict temperatures to increase by 3-7°C will see urban centres in mainland Canada reflect the freeze-thaw cycles of Newfoundland. For example, Ottawa (current mean daily winter temperature of -8.2°C) could shift towards freeze-thaw conditions typical of present-day St. John's, and Quebec City (current mean daily winter temperature of -10.7°C) could increasingly resemble the typical conditions of present-day Corner Brook (Table 1.2). The thawing cycles of Newfoundland are likely more extreme than mainland Canada will experience, allowing the island to serve as a worse-case scenario for future freeze-thaw conditions.

Table 1.2: Current (1981-2010) and future (2071-2100; based on Coupled Model Intercomparison Project Phase 6 (CMIP6) Shared Socioeconomic Pathways (SSP) 2-4.5 and SSP 5-8.5) mean winter (December, January, February) temperatures for four urban centres in Canada, demonstrating that future winter temperature projections of mainland Canada (Ottawa and Quebec City) under SSP 2-4.5 reflect the current conditions in Newfoundland (St. John's and Corner Brook). Current climate data was extracted from the Environment and Climate Change Canada Historical Climate Data website (https://climate.weather.gc.ca/climate_normals_index_e.html) on March 27, 2021, and future climate data was extracted from the Climate Data for a Resilient Canada website (https://climatedata.ca/) on February 20, 2023.

City	Current Mean Winter Temperature (°C)	Future Mean Winter Temperature Projections (°C)	
		SSP 2 - 4.5	SSP 5 – 8.5
St. John's	-3.2		
Corner Brook	-4.9		
Ottawa	-8.2	-4.1	-1.0
Quebec City	-10.7	-4.7	-2.2

Similarly, the frequency of high winds in mainland Canada is expected to increase due to stronger convective storms (Gillner *et al.* 2015), shifting storm tracks (Woollings and Blackburn 2012), and greater frequency of particularly powerful extratropical cyclones ("weather bombs"; e.g., Seiler and Zwiers 2015). This shift is reflective of the wind patterns in Newfoundland (Table 1.1), with the west coast offering an analogue for current mainland conditions (low wind climate), and the east coast offering an extreme proxy for a windier future. The occasional exposure to remnants of tropical storms (post-tropical transitions) and frequent extreme winter weather events in Newfoundland (Allen *et al.* 2010; Reale *et al.* 2019) additionally mean that Newfoundland experiences more extreme peak winds than many other Canadian cities. This again allows Newfoundland to serve as a worse-case scenario of future wind trends in Canadian cities. Newfoundland's winter climate gradient therefore provides ideal conditions for assessing how urban forests in Canada will withstand the transition from the current colder state with fewer high wind days to the future warmer temperatures with more frequent high winds.

Newfoundland has additionally experienced five distinct climate 'epochs' since the late 1800s, defined by periods of high, low, or stationary mean daily temperatures (Banfield and Jacobs 1998). Notably, a cold climate regime dominated the region from the early 1970s until the late 1990s, preceded and followed by periods of warmer climates (Finnis *et al.* 2015). Winds follow a similar trend, with a period of less frequent high winds between the 1980s and mid 2000s, followed by a gradual increase. Newfoundland is currently in an unusually warm epoch, providing a regional amplification of broader global trends; this further allows this epoch (Finnis and Bell 2015; Finnis *et al.* 2015) to serve as a model for potential impacts of rapid warming on existing tree species.

While there is considerable evidence on the impact to health and productivity of trees from warming temperatures during the growing season (Way and Montgomery 2015; Valladares 2017), less attention has been paid to the potential effects of winter warming. Given that warming is expected to be more extreme in the winter season, and winter temperatures are a major limiting factor for tree species (Ramage *et al.* 2013; Jenerette *et al.* 2016; Kendal *et al.* 2018), these potential effects may be more telling of the future impacts to urban forests. The climate gradient in Newfoundland described above make the island an ideal location for examinations of the relationships between urban forests and climate change.

1.2 Study areas

This study took place within four urban centres in Newfoundland, Canada: Corner Brook, Gander, Conception Bay South (CBS), and St. John's (Figure 1.1). For the purposes of this study, an urban centre was defined using an adaptation of the definition of population centre by Statistics Canada (previously referred to as urban area prior to 2011; Statistics Canada 2011) as areas with a population of at least 1,000 persons (Statistics Canada 2021) and high population density compared to other Newfoundland municipalities. While most urban centres included in this study do not meet the population density requirement set by Statistics Canada (2011, 2021) of 400 persons/km^2, they were still considered urban centres as this definition would filter out most municipalities in Newfoundland, including the capital city of St. John's.

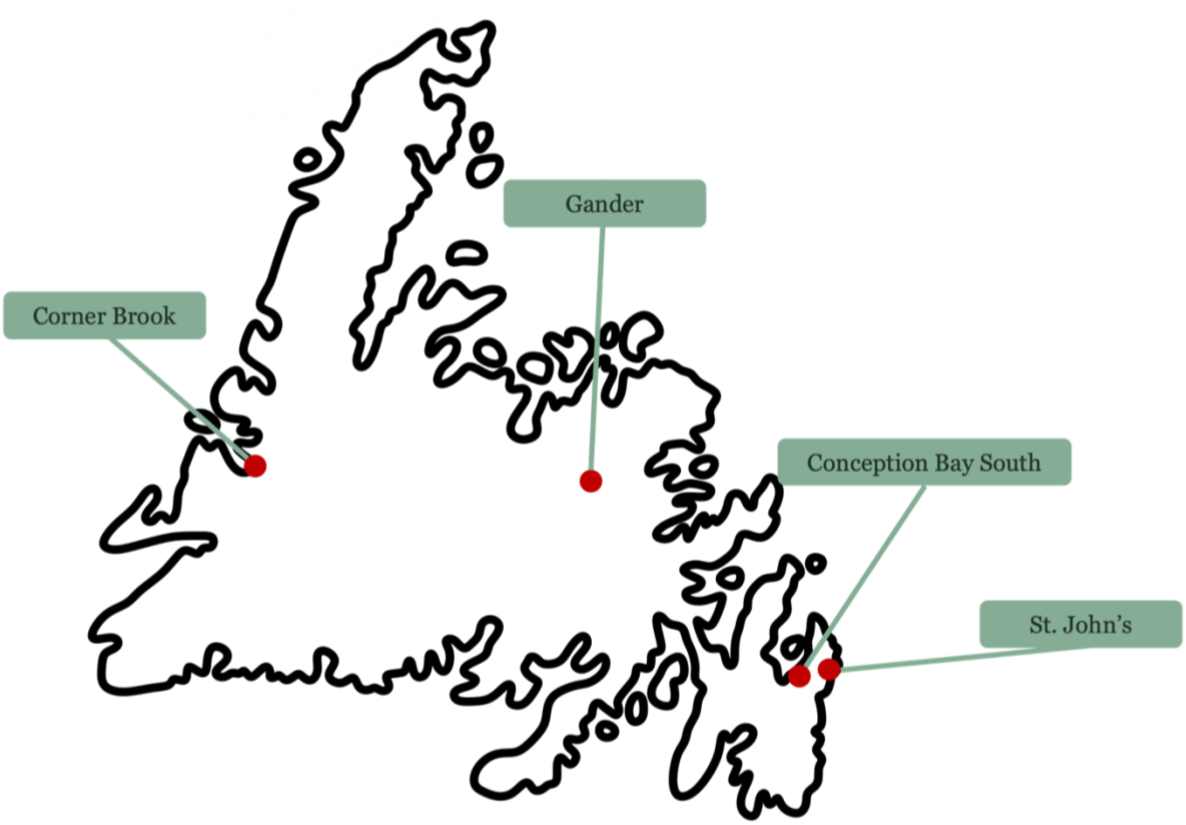


Figure 1.1: Four urban centres in Newfoundland chosen as field locations for this study.

1.2.1 Corner Brook

Corner Brook, located on the west coast of Newfoundland, was incorporated as the province of Newfoundland and Labrador's second city in 1955 when the then town amalgamated with adjacent municipalities. With a population of 19,333 (130.7 persons/km^2) (Statistics Canada 2022a), it is the largest urban centre in Newfoundland outside of the Avalon peninsula.

Part of the west coast climatic zone (Banfield 1983) and Western Newfoundland Ecoregion (Damman 1983; Figure 1.2, Figure 1.3), Corner Brook experiences cold and snowy winters with relatively persistent snow cover in winter months. The region's relatively early spring and moderately warm and sunny summer seasons mean this forest region experiences the most favorable growing conditions on the island (Banfield 1983; Damman 1983). The Western Newfoundland Ecoregion contains fertile soils heavily forested by predominantly balsam fir

(*Abies balsamea*) (Damman 1983). Corner Brook is further categorized into the Corner Brook subregion of the Western Newfoundland Ecoregion, which is characterized by dense forests, rugged topography, and productive, nutrient-rich soils (Damman 1983). Prolonged dry spells are rare in this region, with annual precipitation ranging between 1000mm and 1200mm, making forest fires uncommon (Banfield 1983; Damman 1983).

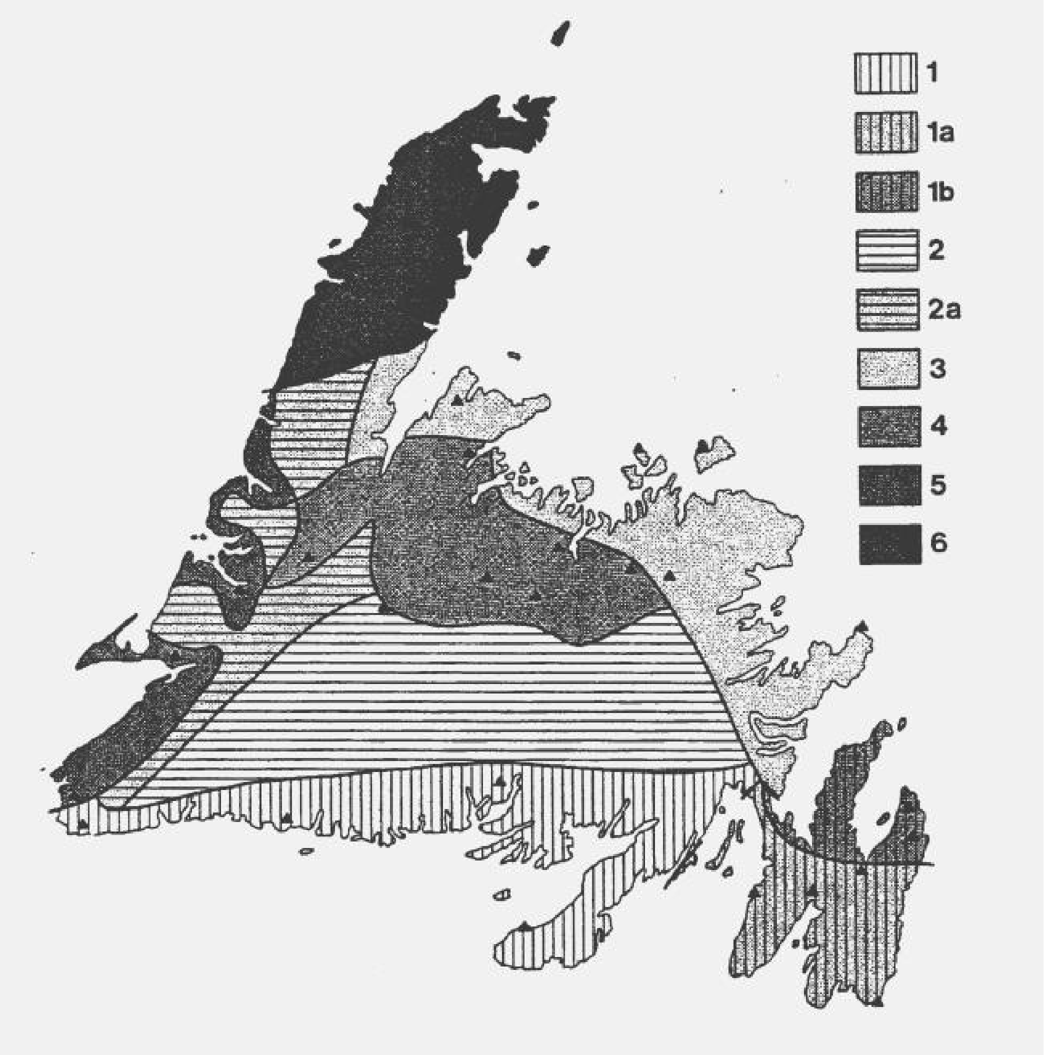
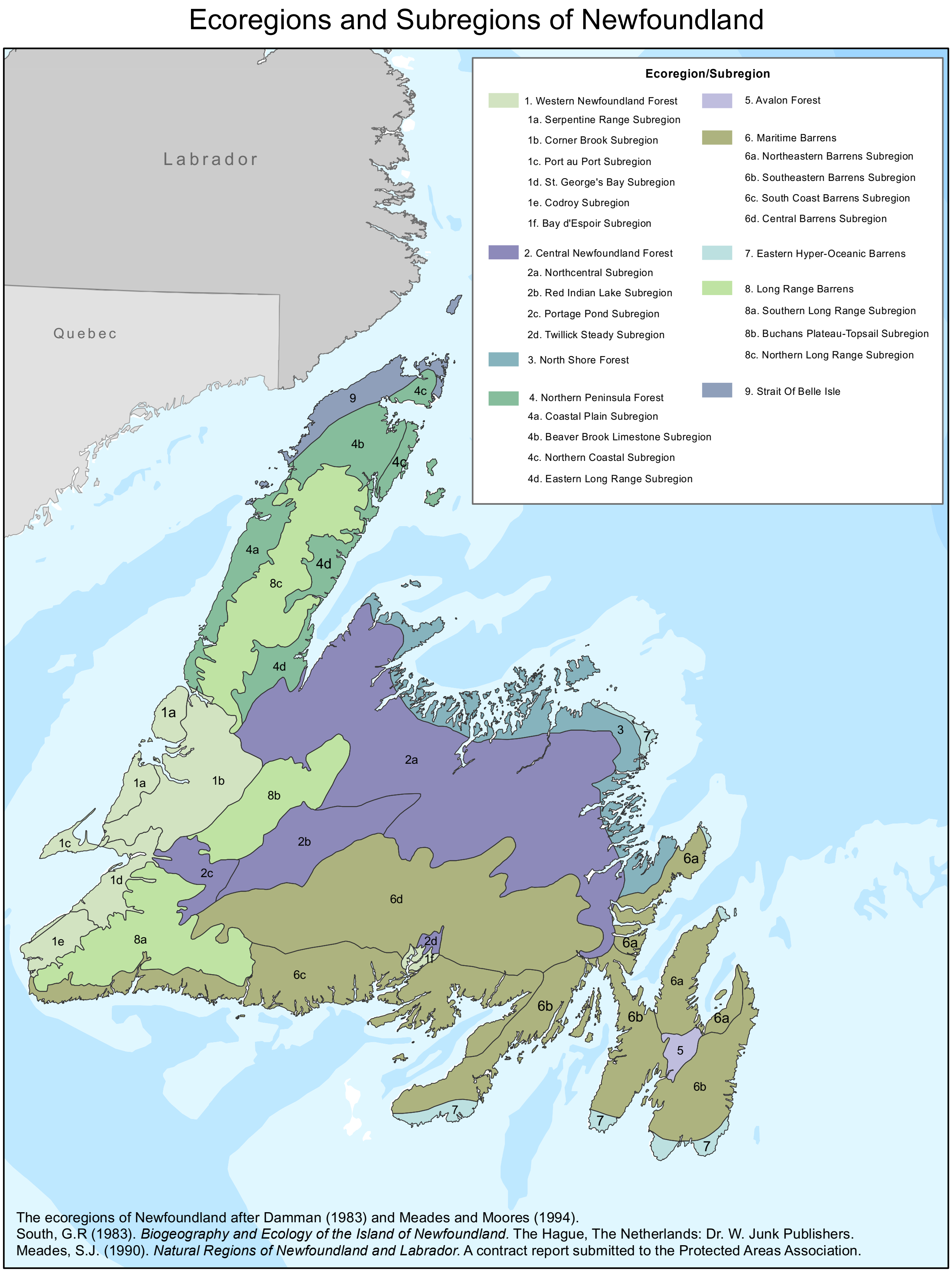


Figure 1.2: Climatic zones map for Newfoundland, Canada. 1, 1a, 1b: South and south-east coasts and immediate hinterlands (subsections defined by winter climates), 2: Central uplands, 2a: western hills and mountains, 3: East coast and hinterlands, 4: Central lowlands, 5: West coast (Bonne Bay and south), 6: Northern Peninsula (source: Banfield 1983).


Ecoregions and Subregions of Newfoundland

The ecoregions of Newfoundland after Damman (1983) and Meades and Moores (1994).
South, G.R (1983). *Biogeography and Ecology of the Island of Newfoundland.* The Hague, The Netherlands: Dr. W. Junk Publishers.
Meades, S.J. (1990). *Natural Regions of Newfoundland and Labrador.* A contract report submitted to the Protected Areas Association.

Figure 1.3: Ecoregions map for the island of Newfoundland, Canada (source: Department of Fisheries, Forestry, and Agriculture, Government of Newfoundland and Labrador)

1.2.2 Gander

The town of Gander, located in the central region of Newfoundland, was first incorporated in 1958, and serves as one of the main hubs for the central portion of Newfoundland, with a population of 11,880 (113.7 persons/km^2) (Statistics Canada 2022b).

Gander is located in the central lowlands climatic region and Central Newfoundland Ecoregion (Figure 1.2, Figure 1.3), which contains the most continentally influenced climate (Banfield 1983; Damman 1983). This part of the island experiences cold and dry winters with very severe frost events (minimum temperatures near -25 to -30) and warm and moderately sunny summer seasons (Banfield 1983). As one of the driest parts of the island – annual precipitation ranging between 900mm and 1000mm (Banfield 1983) – forest fires are an important part of forest dynamics in this heavily forested region (Damman 1983). Following fires, stands of black spruce (*Picea mariana*) and white birch (*Betula papyrifera*) cover extensive areas of the Central Newfoundland Ecoregion, with dwarf shrub kalmia (*Kalmia agustifolia*) dominating areas with poor regeneration or poor nutrient availability (Damman 1983). *Hylocomium*-balsam fir forest type dominates areas in the region not disturbed by fire, and black spruce-lichen forests are exclusively present in this ecoregion (Damman 1983). The town of Gander is located in the Northcentral subregion of the Central Newfoundland Ecoregion, and is characterized by high summer temperatures, lower rainfalls and prolonged dry spells, leading to high forest fire frequency and extensive black spruce stands (Damman 1983).

1.2.3 Conception Bay South

The town of Conception Bay South, located on the Avalon peninsula on the east coast of Newfoundland, was incorporated in 1973 following the amalgamation of nine different communities along the Conception Bay coastline. With a population of 27,168 (454.9

persons/km^2) (Statistics Canada 2022c), it is one of the largest municipalities in the St. John's metro region.

As part of the south and south-east coast and immediate hinterlands climatic zone and Maritime Barrens Ecoregion (Figure 1.2, Figure 1.3), Conception Bay South experiences significant maritime influence on climate, and extensive barren areas (Banfield 1983; Damman 1983). Winters are relatively mild with freezing rain frequent in late winter months, though the northern part of the Avalon Peninsula tends to experience less mild winters than the remainder of this climatic zone (Banfield 1983). Summers are cool with high frequency of sea fog, and annual precipitation ranges between 1200mm and 1700mm (Banfield 1983; Damman 1983).

Historically, the Maritime Barrens Ecoregion contained denser forests than are present throughout the region now; regular forest fires and subsequent settlement of the Avalon Peninsula replaced forests with a more barren landscape (Damman 1983). Presently, forest cover is primarily restricted to valleys, hill tops, and slopes in this ecoregion, with barrens composed of dwarf shrub, heaths, bogs, and shallow fens dominating the region (Damman 1983). Remaining forests in the region are generally dense and poor quality: while diameter growth is good, height growth remains poor (Wilton 1956; Damman 1983). Balsam fir is the most important tree in these remaining forests (Damman 1983). Conception Bay South can be further categorized into the Northeastern Barrens subregion of the Maritime Barrens Ecoregion, which is characterized by warmer summers, lower fog frequency, and more extensive forests.

1.2.4 St. John's

St. John's, also located on the Avalon peninsula, was the province of Newfoundland and Labrador's first city – incorporated in 1921 – and has since become the capital city. As the capital city, it houses almost 25% of the province's 510,500 inhabitants (Statistics Canada 2022e), with a population in the city of 110,525 (247.8 persons/km^2) (Statistics Canada 2022d).

Like Conception Bay South, St. John's is part of the south and south-east coast and immediate hinterlands climatic zone and Maritime Barrens Ecoregion (Figure 1.2, Figure 1.3), and therefore these two urban areas share climatic and vegetative characteristics. St. John's, however, is in the Southeastern Barrens subregion of the Maritime Barrens Ecoregion, which is defined by extensive barrens of *Kalmia agustifolia* with small pockets of forests (Damman 1983).

1.3 Thesis overview and objectives

The purpose of this research was to gain a better understanding of urban forests on the island of Newfoundland, and assess the potential for the island to act as a model of future urban forest conditions that will be found throughout mainland Canada, and potentially other regions of North America. Here, I wanted to better understand how the urban forest benefits urban landscapes, and how urban landscapes impact the status of urban forests by incorporating 2 key urban forest research priorities highlighted by Bush *et al.* (2018): 1) development of ecological indicators and baseline data, and 2) continued studies on the effect of climate change parameters on tree growth and functionality. In this chapter, I provided an extensive introduction and context for this research. I discussed the importance of ecological integrity in urban forests to maintain the extensive services they provide. I also discussed how urban forests influence the microclimate of urban spaces, and how the climate patterns in Newfoundland replicate future changes in Canada's climate. What follows in Chapter 2 is a description of our study design, and our findings, including characteristics found in different parts of the urban forest landscape in

Newfoundland, and their influence on local microclimates. Specifically, this project addressed the following research objectives:

1. **Compare how characteristics of urban forests on the island of Newfoundland change throughout the urban forest landscape**

    H1: Areas within the urban landscape with increased development do not resemble the characteristics of more natural intact boreal forest stands of the region. This means that **P1**: stand and site characteristics and species compositions of peri-urban sites will not be reflective of those in treed urban and untreed urban sites.

2. **Empirically assess urban tree mitigation of temperature-related climate change impacts**

    H2: Temperature related impacts of climate change – specifically daily temperatures and weather events – are mitigated by urban forests. This means that **P2**: daily temperature ranges as well as the frequency and duration of weather events in treed urban and peri-urban sites will not be as extreme as those in untreed urban sites; that is, notable temperature extremes (heatwaves; hard frosts) are expected to occur less often, and/or show shorter durations.

Finally, in Chapter 3 I outline suggestions for further urban forest research both on the island and beyond, and discuss implications of our research for municipalities and urban forest managers, including the shifting focus of non-native species in urban spaces. Incorporating knowledge of local and future climates, site conditions, environmental characteristics, and desired services into managing current and future urban forests remains one of the greatest challenges in urban forest management. A canopy dominated by trees that are not suited to future climate and environmental conditions can lower ecological integrity, and hinder the ability of urban forests to adapt to climate change. Similarly, changes to the urban landscape, including altered climates, will impact the ecological integrity of urban forests. Integrating these findings of ecological integrity and climate mitigation into current and future urban forest management practices is necessary to ensure the longevity of urban forests and their benefits into the future.

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Chapter 2: Searching for climate change solutions in Newfoundland's urban forests

2.1 Introduction

Trees within an urban centre are an important component of healthy, sustainable cities, and provide ecological, aesthetic, and health benefits to urban areas and residents (Kardan *et al.* 2015). These urban forests help mitigate the effects of climate change in urban centres (Strohbach *et al.* 2012): they improve air quality by filtering air pollutants and enriching oxygen (Nowak *et al.* 2006); reduce the impact of extreme weather events by serving as a wind break (Akbari *et al.* 2001) and limiting flood levels (Roy *et al.* 2012); and mitigate the urban heat island effects by regulating temperature and humidity (Roy *et al.* 2012). Urban forests also act as a carbon sink in cities (Strohbach *et al.* 2012), which have the some of the highest carbon emissions globally (McCarthy *et al.* 2010). However, the effects of climate change may hinder the climate mitigation services provided by urban forests if trees are not able to adapt to novel climate conditions (Ordóñez and Duinker 2015).

The ability to adapt to new and changing conditions, or adaptive capacity, varies both within and between species (Melillo *et al.* 1996; Aitken *et al.* 2008; Roloff *et al.* 2009; Esperon-Rodriguez *et al.* 2020), and stand-alone trees in an urban environment respond differently than an assemblage of trees in a forested setting (Kendal *et al.* 2018). Species also vary in their ability to provide benefits and services to the urban landscape (Manes *et al.* 2012; Conway *et al.* 2019; Amini Parsa *et al.* 2020; Schlaepfer *et al.* 2020), and low adaptive capacity of some species may further reduce urban forest ecosystem services.

The functioning and wholeness of an ecosystem in comparison to its natural habitat can be described as its ecological integrity (Angermeier and Karr 1994; Parrish *et al.* 2003). Through this lens, urban forests can seem to lack integrity (McDonnell *et al.* 2008) as human-nature interactions influence their structure and functions (Ordóñez and Duinker 2012). Incorporating the complex socio-ecological dynamics of urban areas into ecological assessments would provide a more accurate representation of the integrity of these human-nature landscapes (Ordóñez and Duinker 2012). The similarity between adaptive capacity and resilience means low adaptive capacity lowers the ecological integrity of the ecosystem (Holling 1973; Ordóñez and Duinker 2012). Climate change exacerbates the risk of stress, damage, and mortality in urban forests with low adaptive capacity, and has the potential to lower the adaptive capacity of trees (Esperon-Rodriguez *et al.* 2020), making ecological integrity essential in the maintenance of the urban forest, and mitigation of future climate change impacts.

Some of the most devastating losses of ecological integrity in North America are a result of extreme and abnormal weather events, which push trees to the limits of their adaptive capacity (Burley *et al.* 2008; Ordóñez and Duinker 2012). Wind causes breakage of limbs or stems, stem upheaval, and soil damage (Oliver 1980; Foster and Boose 1992; Burley *et al.* 2008); shifts in temperature regimes cause changes in spring phenology patterns (Flynn and Wolkovich 2018; Ettinger *et al.* 2020); freeze-thaw cycles alter spring phenology patterns, cause damage from ice buildup, and increase the risk and occurrence of embolism (Mayr *et al.* 2003; Sperry and Sullivan 2020; Tedla *et al.* 2020). The impacts of weather on urban forests are further compounded by additional anthropogenic stresses (e.g., low soil quality, limited water resources, urban heat island effect, air pollution; Sieghardt *et al.* 2005; Westra *et al.* 2012). Climate change is further exacerbating these issues by increasing the occurrences of extreme weather events and causing shifts in temperature patterns. Adaptation and resistance to change requires an urban forest with diverse structures (Ordóñez and Duinker 2015), composed of species that are tolerant to changes in climate (Yang 2009; Roloff *et al.* 2009) and other damaging agents (Poland and McCullough 2006).

While there is considerable evidence on the impact to health and productivity from warming temperatures during the growing season (Way and Montgomery 2015; Valladares 2017), less attention has been paid to the potential effects of winter warming. The predicted effects of climate change in Canada vary throughout the country; however, the urbanized south is expected to experience warmer winter temperatures, increased freeze-thaw cycles, prolonged mid-winter thaws, and increased frequency of high winds (Lemmen *et al.* 2008; Woollings and Blackburn 2012; Gillner *et al.* 2015; Seiler and Zwiers 2015).

Newfoundland is an ideal natural laboratory in which to examine relationships between many anticipated climate shifts and urban forest health and microclimatic effects, particularly during winter conditions. Due to the proximity to both cold (Labrador current) and warm (gulf stream) ocean currents as well as North Atlantic storm track, Newfoundland experiences highly variable and unusually active weather; resulting winters are characterized by pronounced temperature shifts; relatively high winds; rapid freeze-thaw cycling; and frequent icing events (Banfield and Jacobs 1998). Major urban centres on the island exist on a climate gradient, with the east coast experiencing all of these winter hazards more frequently than the relatively cooler and calmer west coast (Environment and Climate Change Canada 2011). Newfoundland has additionally experienced five distinct climate ‘epochs’ since the late 1800s, the most recent of which has created enhanced warming trends (Finnis *et al.* 2015) and frequent extreme wind events in the past 20 years. While likely more extreme than mainland Canada will experience (Lemmen *et al.* 2008, Allen *et al.* 2010b; Reale *et al.* 2019), these compounded aspects of Newfoundland climate provide opportunities to study i) the influence extreme winds, icing, and freeze-thaw conditions on urban forest health and microclimatic influence, and ii) the impacts of rapid winter warming on existing urban trees. However, such studies require initial evaluation of composition, health, and microclimates of urban forests across the province.

Here, we aim to better understand the relationship between the urban landscape and urban forests on the island of Newfoundland by assessing i) how characteristics of the urban forest change throughout the urban landscape and ii) how local climate change impacts are mitigated by urban trees. We hypothesized urban forests located in more developed parts of the landscape have few shared characteristics with the native boreal forest stands of the region, and that temperature related climate change impacts – including daily temperature fluctuations and weather events – are mitigated by urban trees. Integrating these findings of ecological integrity and climate mitigation into current and future urban forest management practices is necessary to ensure the longevity of urban forests and their benefits into the future.

2.2 Methods

Our study took place in four urban centres on the island of Newfoundland, covering the western, central, and eastern regions of the island: Corner Brook, Gander, Conception Bay South (CBS), and St. John's (Figure 1.1). Given the geographic extent of our study sites, they represent three distinct ecoregions and three separate climatic zones. Corner Brook is part of the west coast climatic zone, defined by cold winters with reliable snow cover, and relatively warm sunny summer seasons (Banfield 1983). As part of the Western Newfoundland Ecoregion, it contains the most favorable growing conditions on the island, with heavily forested woodlands dominated by balsam fir (*Abies balsamea*; Damman 1983). Moving further east, Gander is part of the central lowlands climatic zone, which is the most continentally influenced climatic zone in Newfoundland, with cold, dry winters and warm, dry summers (Banfield 1983). The dry climate associated with this part of the Central Newfoundland Ecoregion makes it the most distinctly boreal ecoregion on the island, where forest fires are common, leading to vast forest cover of black spruce (*Picea mariana*) and white birch (*Betula papyrifera*; Damman 1983). CBS and St. John's, located on the eastern Avalon Peninsula, are both part of the south and south-east coast and immediate hinterlands climatic zone. The climate in this region has a heavy maritime influence, with mild winters and cool summers, and dense sea fog – though these trends are more pronounced in the region surrounding St. John's (Banfield 1983). Both urban centres are also part of the Maritime Barrens Ecoregion, which contains extensive barrens, while forests – primarily composed of balsam fir – are restricted to valleys, hill tops, and slopes (Damman 1983).

2.2.2 Study design

We used two gradient analyses that extended from the west coast to the east coast of Newfoundland, which allowed us to compare results along a scale of environmental conditions (Ter Braak and Prentice 1988; McDonnell and Hahs 2008). Gradients are a well-established analysis tool to assist in the understanding of the ecology and distribution of plants in response to changes in the environment (Ter Braak and Prentice 1988), including changes in climate (Pardos *et al.* 2021). This research project used two gradients: 1) crossing three distinct climate zones and associated ecoregions, and 2) urban forest canopy cover, where the urban forest canopy cover gradient is nested within the gradient of winter climates. These gradients influenced decisions regarding the choice of four urban study areas in Newfoundland: Corner Brook, Gander, CBS, and St. John's – which were systematically chosen to represent a gradient of local winter climate regimes. Here, the west coast of the island tends to experience cooler temperatures with low winds, and the east coast tends to experience milder temperatures with more frequent high winds.

These two gradients allowed us to compare our results along a scale of local climates, and assess how these climates influence the ecological integrity of urban forests (Heckmann *et al.* 2008). Nested within each urban centre, we established the urban forest canopy cover gradient: untreed urban (UU), treed urban (TU), and peri-urban (PU) areas. These three site types were chosen to assess how the ecological integrity of urban forests compare to more natural continuous closed-canopy forest and to open-canopy urban environments.

PU sites were defined as large tracts of continuous canopy forest within or nearby the urban centre. Here, we chose sites that would be representative of a native forest stand for each region of the island (i.e., unmanaged or unmanicured parks and recreation areas), but remained accessible to residents (e.g., through walking trails; Figure 2.1). Residential properties that contained at least 1 tree were used as TU sites, with permission from property owners. TU sites were contained within one neighbourhood of each urban centre to reduce variation in our results based on site conditions, such as elevation and proximity to coastlines. Business and institutional regions of the urban centre with little to no tree cover were chosen as our UU sites. Corner Brook was the exception to this, given the more extensive urban forest that exists throughout this urban centre, UU sites in downtown Corner Brook contained more tree cover than other study locations. However, an effort was made to still place sites in regions of the downtown area that had minimal canopy cover.

Figure 2.1: Examples of a) peri-urban (Manuels River, Conception Bay South), b) treed urban (Townsite Neighbourhood, Corner Brook,) and c) untreed urban (Memorial University campus, St. John's) site types representing the urban forest canopy cover gradient.

Urban centres were first assessed using Google Earth to identify potential regions that could represent each site type. This was followed up with an initial reconnaissance visit to ensure the requirements of each site type were met, and finally permissions to conduct research were obtained from municipal representatives, charitable organizations, and residents prior to finalizing field sites in each urban centre (Table 2.1, Figure 2.2, Figure 2.3, Figure 2.4).

Table 2.1: Locations chosen as untreed urban, treed urban, and peri-urban sites in each of four urban centres in Newfoundland.

	Untreed Urban	**Treed Urban**	**Peri-Urban**
Corner Brook	Downtown Corner Brook	Townsite Neighbourhood	Margaret Bowater Park
Gander	Airport Boulevard	Elizabeth Drive-Magee Road-Cooper Boulevard	Airport Nordic Ski Club
Conception Bay South	Business Area of CBS Highway	Talcville & Chamberlains Neighbourhoods	Manuels River
St. John’s	Memorial University Campus	Georgestown & Rabbit Town Neighbourhoods	MUN Botanical Garden

Figure 2.2: Map of peri-urban (Margaret Bowater Park; green), treed urban (Townsite Neighbourhood; pink), and untreed urban (Downtown Corner Brook, gold) site type locations in Corner Brook. Each site type had 10 individual plots.

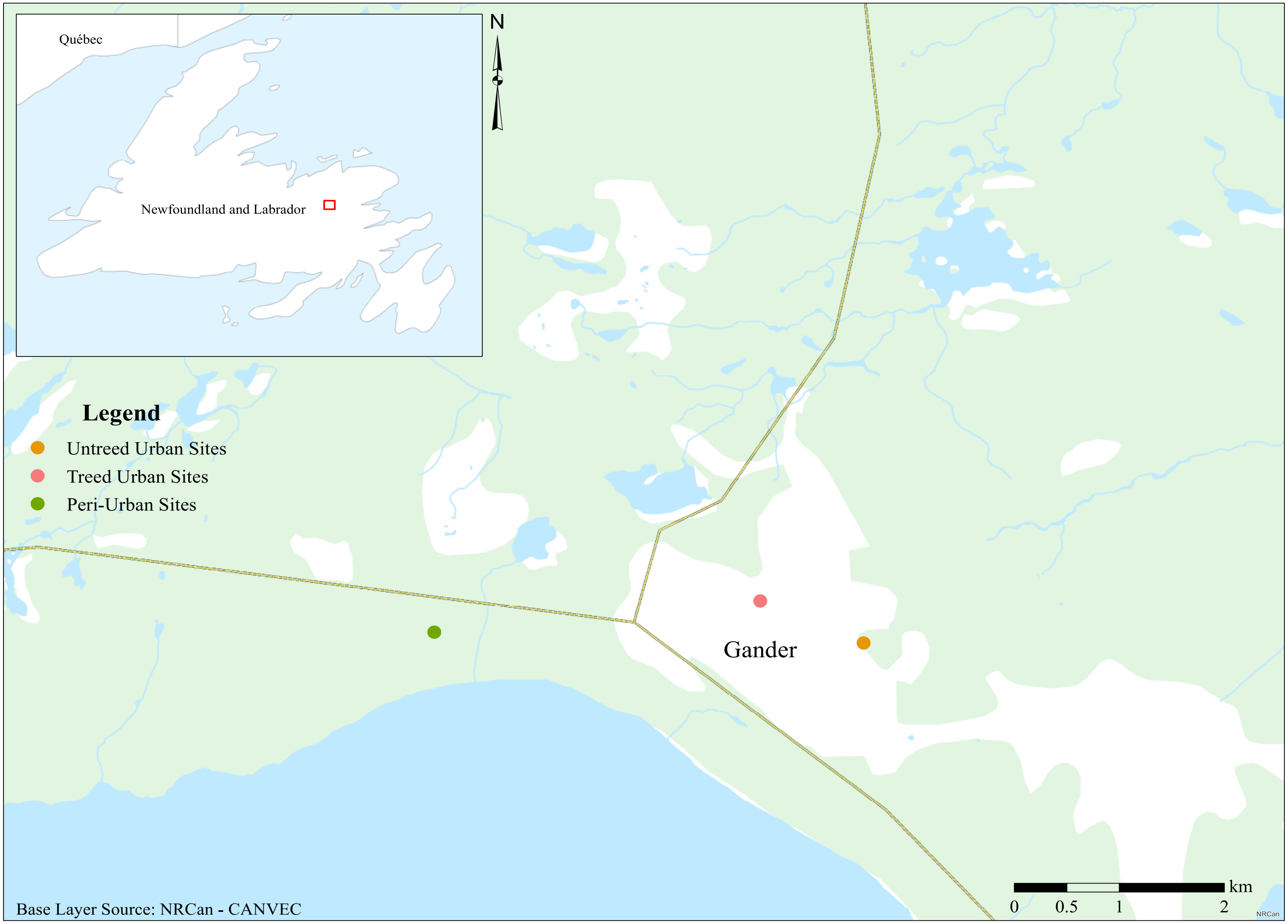

Figure 2.3: Map of peri-urban (Airport Nordic Ski Club; green), treed urban (Elizabeth Drive-Magee Road-Cooper Boulevard; pink), and untreed urban (Airport Boulevard, gold) site type locations in Gander. Each site type had 10 individual plots.

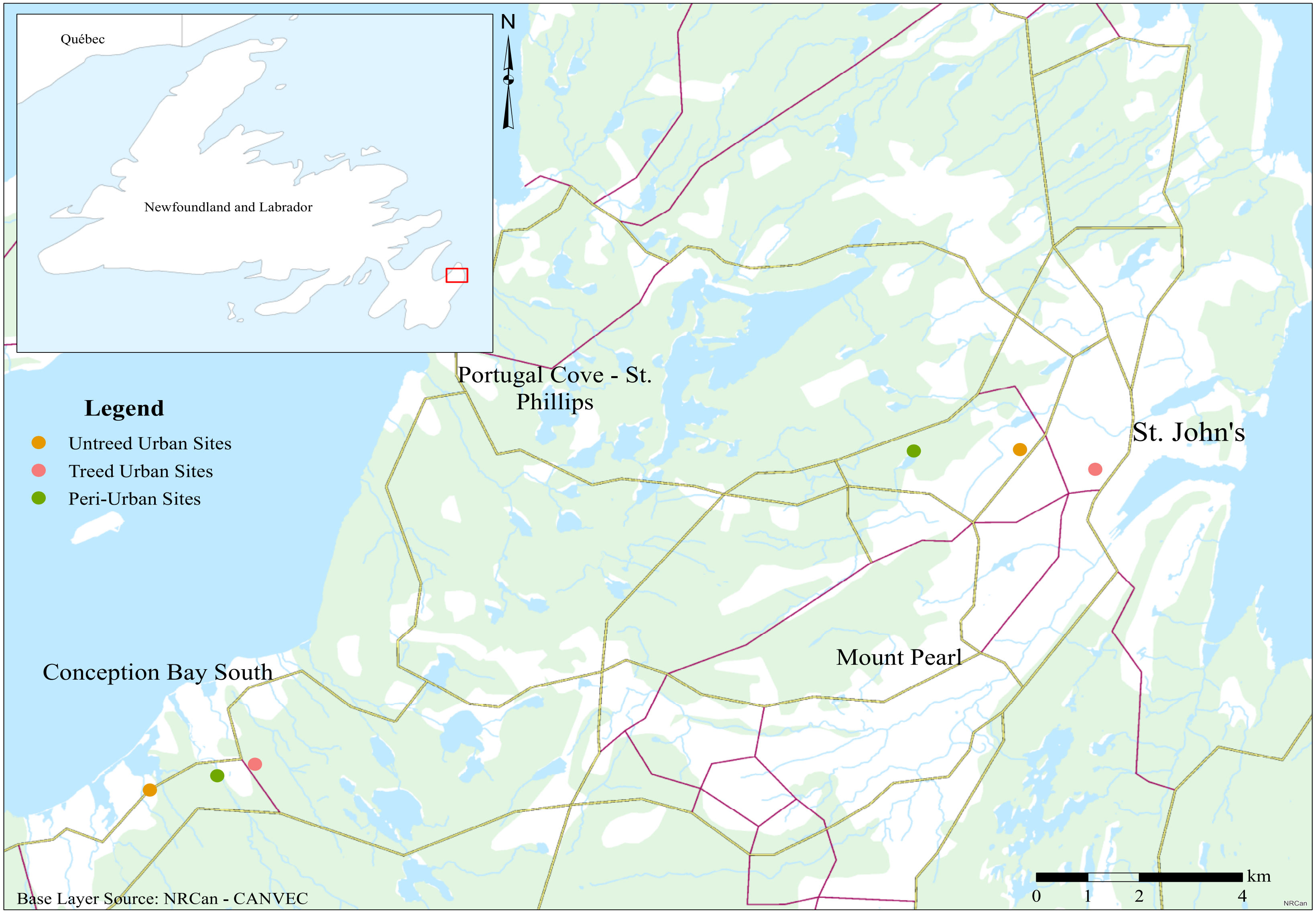

Figure 2.4: Map of site type locations in Conception Bay South and St. John's. Site types in Conception Bay South are Manuels River (peri-urban; green), Chamberlains & Talcville Neighbourhoods (treed urban; pink) and CBS Highway Business Area (untreed urban; gold). Site types in St. John's are MUN Botanical Garden (peri-urban), Georgestown and Rabbittown Neighbourhoods (treed urban) and Memorial University Campus (untreed urban). Each site type had 10 individual plots.

2.2.3 Ecological integrity sampling methods

2.2.3.1 Field methods

There are many ways to measure the ecological integrity of an ecosystem (Steedman and Haider 1993). For the purposes of this study, we focused on factors that influence the substrate, community composition, and structure. Within these factors, there are a variety of measures that contribute to ecological integrity in an urban forest, including species richness, microclimate regulation, nutrient availability, tree density and structure, and canopy cover (Heckmann *et al.* 2008; Ordóñez and Duinker 2012). At each site (e.g., PU in CBS), we established 10 plots, resulting in a total of 40 replicates in each level of the canopy cover gradient (i.e., 10 PU plots in each of the four urban centres). Plot locations in both UU and PU sites were systematically chosen to best represent the natural conditions of the surrounding area. In TU sites, plot locations were dictated by availability of trees on properties and permissions from property owners. Plots were located a minimum of 10m from trails in PU sites to avoid the main edge effects of adjacent land-use (i.e., light and wind infiltration from edges; Matlack 1994). However, given sites are located within urban centres we recognized it would not be possible nor representative to fully avoid edge effects as they are distinctly present in urban forests (Guerra *et al.* 2017), and an important characteristic to capture within all parts of the urban landscape. To assess our different measures of ecological integrity, two field assessments were conducted at each plot, where the centre of the plot was a tree or utility pole: 1) a smaller 1m x 1m plot, and 2) a larger 5m radius plot (Figure 2.5).

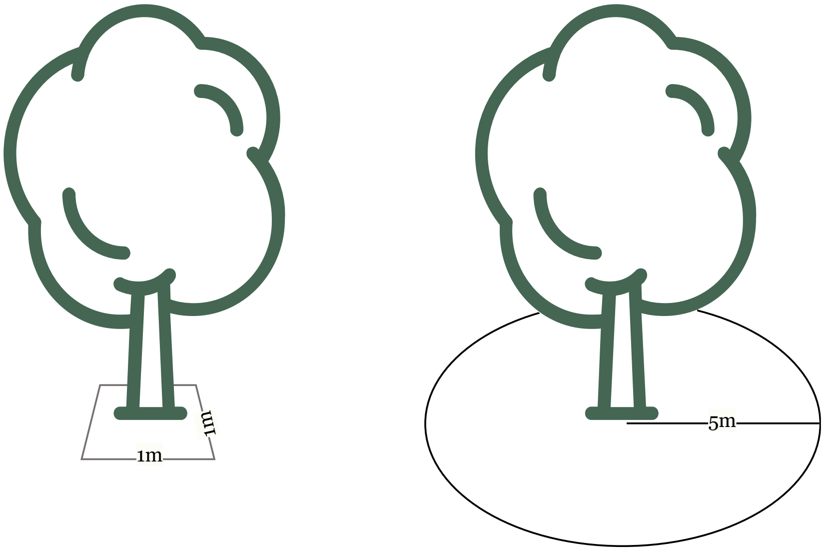

Figure 2.5: Two scales of field assessments used to measure ecological integrity: the first (left) being a smaller 1m x 1m plot and the second (right) being a larger 5m radius plot. The central point of each plot was a tree (peri-urban and treed urban) or utility pole (untreed urban).

Small plots were laid out using pre-measured PVC piping with the tree/utility pole in the centre of the plot and corners aligned with the cardinal directions (Figure 2.6a). Within each small plot, we conducted a ground vegetation survey whereby all woody and herbaceous vegetation was identified and counted and – in TU and PU sites – identified the plot centre species, both of which contribute to a measure of species richness and evenness, influencing the community composition. Factors that influence the substrate were measured in small plots by recording the percentage of ground cover types within the plot and soil organic matter depth at each of the four corners using a Japanese soil sword (Nisaku). Ground coverage was measured by the same individual (Pearson) at each plot to ensure consistency in percentage estimations. We also assessed measures of structure in the small plot, including canopy cover, canopy height, and tree height. Total tree height and canopy height of the centre tree were measured using a Nikon Forestry Pro II Laser Rangefinger/Hypsometer. The level of canopy cover (including cover from buildings and other urban structures) was measured at the northern edge of the smaller plot using a Model-A spherical densiometer.

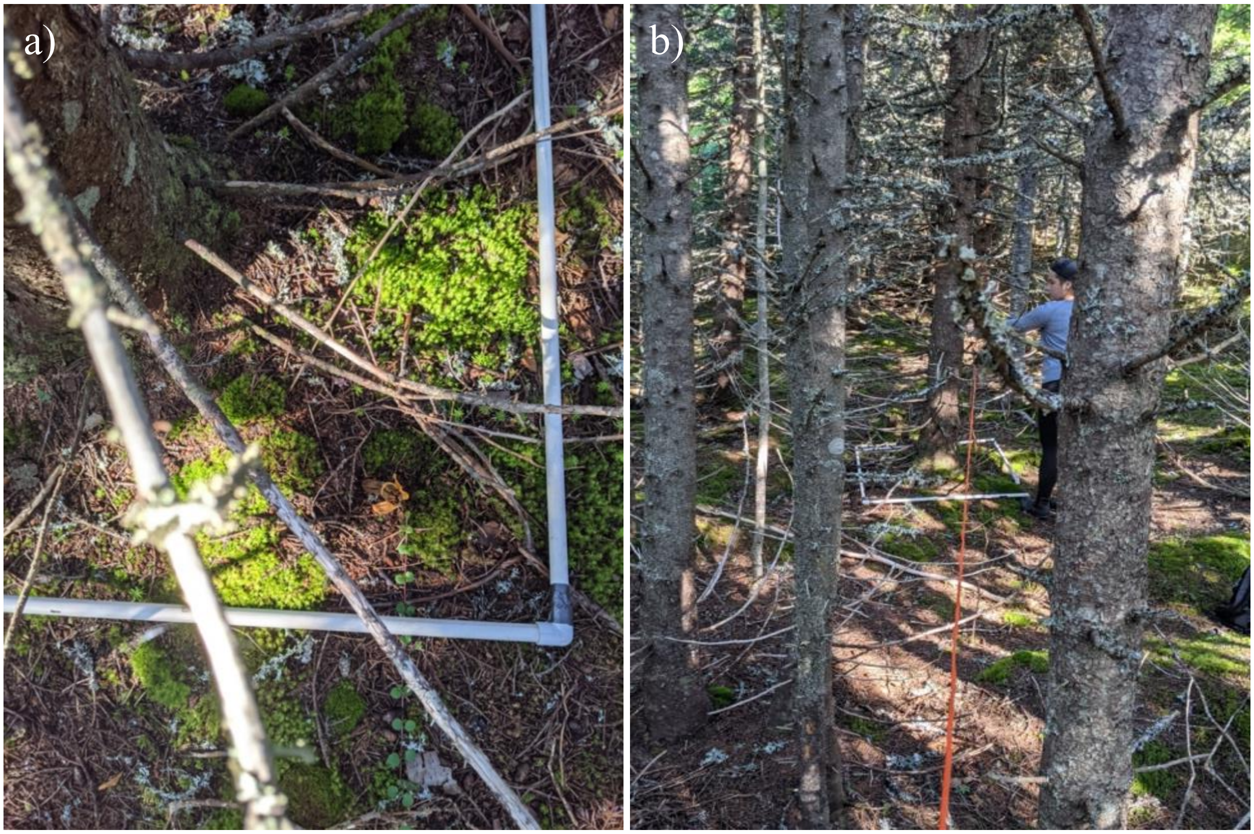

Figure 2.6: Layout of a) small 1m x 1m plot and b) large 5m radius plot. Small plots were established with PVC pipe surrounding a tree that acted as the centre of the plot where the four corners were placed in the four cardinal directions. Large plots were established using a 5m plot chord with a tree representing the centre of the plot.

Large plots were established using a 5m plot cord which was extended in all directions from the plot centre (Figure 2.6b). Here, we recorded the species identification of each tree that fell within the plot – including the centre tree in PU and TU sites – and measured their DBH using a diameter-tape to contribute to our measures of both community composition and structure. All overstory (large plot) and understory (small plot) species were identified as native or non-native using the guides listed in Appendix I and contributed to the measure of species richness.

2.2.3.2 Statistical analyses

The stand and site features measured in the ecological integrity assessment each contribute to one of three ecological integrity measures, and together help inform the overall ecological integrity for each plot (Figure 2.7). The factors influencing ecological integrity were then compared along the both the winter climate and urban forest canopy cover gradients.

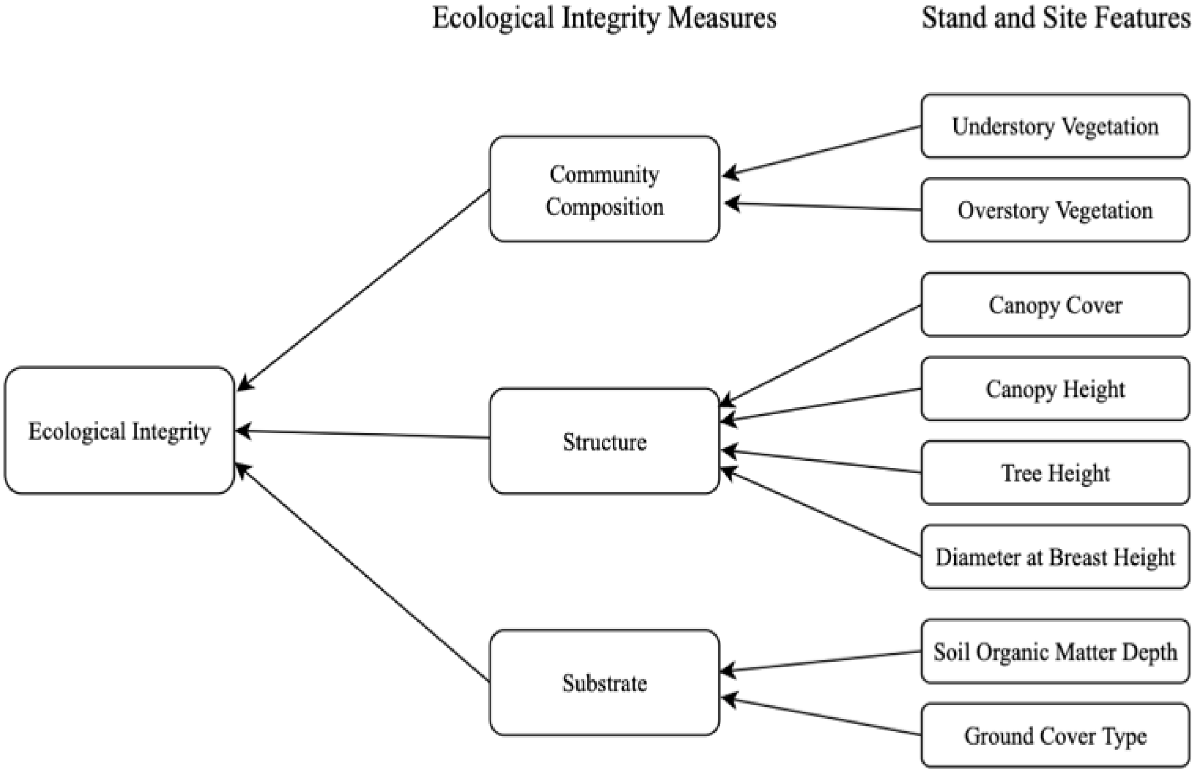

Figure 2.7: Flow chart representing how stand and site features measured at each plot relate to three measures of ecological integrity.

All statistical analyses were performed using R version 4.0.3 (R Core Team 2021) via RStudio version 1.3.1093 (R Studio Inc. 2020). To test our hypothesized differences in our stand and site-level ecological integrity measures between urban centres and site types, we used generalized linear models with urban centre, site type, and their two-way interaction as predictor variables. Canopy cover, diameter, and height (i.e., continuous data) were modelled with the Gaussian distribution, and species richness (i.e., discrete data) was modelled with the Poisson distribution (Zuur 2009) using the “lme4” package (version 1.1-30; Bates *et al.* 2022). Models met assumptions of homogeneity, normality, and independence using residual vs. fitted values, QQ-plots, and residuals vs. explanatory variables, respectively (Appendix II; Zuur 2009).

2.2.3.3 Multivariate analysis

To characterize urban forests in Newfoundland, we performed non-metric multidimensional scaling (NMDS) using the "vegan" package (version 2.6-2; Oksanen *et al.* 2022). NMDS is a non-parametric ordination technique that places ecological data along a chosen number of axes based on the similarity of observed data. As we did not have a substantial enough presence of many overstory and understory species to act as the basis for NMDS, we based the ordination on the percent cover of ground cover types (Appendix III). Our NMDS graphs used Bray-Curtis coefficients as measures of dissimilarity (Oksanen *et al.* 2022), commonly used with ecological data, and the best results were reached under two dimensions.

Our initial NMDS analyses led us to perform subsequent comparisons. Using ninety-five percent confidence interval ellipses around the centroid of each site type on our NMDS containing data from all urban centres and all site types, we observed that ground cover types were strongly influenced by site type. Similarly, vegetation groups (i.e., coniferous vs. deciduous, native vs. non-native), and environmental characteristics appeared to be strongly associated with site types. Based on those initial findings, we then ran NMDS analyses for each urban centre to assess if the influence of site types on ground cover and vegetation in both the overstory and understory was consistent at this smaller scale. Finally, we ran NMDS analysis for PU and TU site types with ninety-five percent confidence interval ellipses around the centroid of each urban centre to observe any differences in ground cover types, vegetation communities, and environmental characteristics influenced by the study locations.

2.2.4 Temperature monitoring methods

2.2.4.1 Field methods

Above ground and below ground temperature loggers were installed at each plot to monitor air and soil temperature, respectively, every 4 hours using iButton Thermocron dataloggers (Maxim Integrated, iButton Thermochron, DS1921G-F5#). Networks of stationary temperature sensors are very effective at characterizing broad temporal patterns and dynamics (Ziter *et al.* 2019), and the small size of the iButton Thermocron dataloggers reduce the likelihood of tampering, theft, or damage in high-profile areas (Malevich and Klink 2011). All dataloggers were wrapped in peri-film prior to their deployment to reduce the potential for damage from precipitation, ground water, or pooling water. Above ground dataloggers were installed on utility poles and signposts in UU sites, and on tree trunks in both TU and PU sites (Figure 2.8). They were located between 1.5m and 2m above ground level on the north-facing side to avoid deep snowpack and the effects of direct sunlight (Malevich and Klink 2011). Below ground dataloggers were installed between 5cm and 10cm under the soil, directly below the above ground logger, approximately 1 foot from the base of the tree to avoid roots. Metadata was collected at each site, guided by Oke (2006), including ground cover types, level of canopy cover, canopy height, species composition, and surrounding urban infrastructure.

Figure 2.8: iButton Thermochron datalogger attached to the trunk of a tree in a peri-urban site. Datalogger is approximately 1.8m above ground, on the north side of the tree. Photo is taken facing south.

Most dataloggers were installed during the fall of 2020; however, due to delays in gaining permissions from residential property owners, UU and TU sites in CBS and St. John's were installed in January 2021. Temperature data was downloaded twice throughout their deployment, once in spring 2021 and a second time in late fall 2021/early winter 2022. iButton dataloggers were collected a final time in summer 2022.

2.2.5.3 Data analyses

To determine the influence of urban forests on local temperatures, hourly data from temperature loggers were first aggregated into daily temperature data (minimum, maximum, and mean), averaged over all ten temperature loggers in a given site. Resulting above- and below-ground daily timeseries were then used to calculate an array of climate indices for each site, related to plant health, stresses, or potential microclimatic influence (Table 2.2). Initial analyses focused on seasonal temperature averages, including daily minima (Tmin), maxima (Tmax), and means (Tavg) calculated for winter (December through February) and summer (June through August); microclimatic influence was further assessed by considering the number of relatively hot days (Tmax > 20ºC; > 25ºC). Plant growth potential was examined in terms of growing season duration, measured as the maximum duration between subsequent frosts (last spring frost to first fall/winter frost), and growing degree days (calculated using a base of 5ºC and 10ºC). Winter stresses and microclimate variations were considered in terms of various cold events. These included the frequency frost days (above ground Tmin < 0ºC), more extreme freeze days (above ground Tmin < -2ºC), ground freezes (below ground Tmin < 0ºC), and freeze-thaw days (above ground Tmax > 0ºC and Tmin < 0ºC).

Table 2.2: Climate indices examined for warm and cold seasons, and the period in which they were examined. Data collection for all sites occurred between January 2021 and July 2022.

	Climate Index	**Period Examined**
Warm Season	Mean Daily Minimum Temperature	June-August
	Mean Daily Maximum Temperature	June-August
	Mean Daily Temperature	June-August
	Growing Season Length	Full Year
	Growing Degree Days; base 5ºC	Full Year
	Growing Degree Days; base 10ºC	Full Year
	Number of days with Tmax > 20ºC	Full Year
	Number of days with Tmax > 25ºC	Full Year
Cold Season	Mean Daily Minimum Temperature	December - February
	Mean Daily Maximum Temperature	December - February
	Mean Daily Temperature	December - February
	Frost Days (Tmin < 0ºC)	Full Year
	Freeze Days (Tmin < -2ºC)	Full Year
	Number of Freeze/Thaw Days	Full Year
	Freeze Event Frequency	Full Year
	Freeze Event Duration	Full Year

2.3 Results

2.3.1 Ecological integrity characteristics

2.3.1.1 Stand and site characteristics

Canopy cover, total tree height, and diameter at breast height capture aspects of structural diversity, which contributes to the ecological integrity of an ecosystem. The canopy cover in TU sites were often similar to PU sites, with CBS containing the biggest variation ranging from 68.7 ± 37.3% in TU to 87.8 ± 6.9% in UU (see Table 2.3 for summary statistics of stand and site characteristics). Here, we also observed that TU had a much higher variation in canopy cover between sites than in PU areas (Figure 2.9, see Table 2.4 for model summaries of stand and site characteristics). UU sites overall had the least closed canopy, however, some sites in Corner Brook had up to 75% canopy cover. While the diameters of trees in TU sites were larger than in PU, there was more variation in the DBH of trees in TU sites. In contrast, total tree heights in PU sites were taller than those in TU, and there was little variation in either site type. These results indicate that there is generally more structural diversity in TU sites than PU and UU sites, indicating higher ecological integrity. However, there are other important factors that contribute to the ecological integrity of an ecosystem. Another important consideration in ecological integrity, nutrient availability, was lower in TU sites than PU sites where ground cover was primarily composed of natural materials that provide nutrients to the ecosystem (Lindenmayer and Franklin 2002), like coarse woody debris (CWD), moss, and other trees. No trends were detected in SOM depth.

Table 2.3: Summary statistics of stand and site characteristics. Means are shown with standard deviation in parentheses.

Urban Centre	**Site Type**	**Canopy Cover (%)**	**DBH (cm)**	**Height (m)**	**Overstory Richness**	**Understory Richness**
Corner Brook	PU	81.07 (9.16)	26.88 (8.22)	13.36 (3.42)	4.0 (1.41)	2.7 (2.36)
Corner Brook	TU	81.28 (18.24)	42.77 (20.42)	10.26 (3.25)	0.4 (0.97)	1.9 (0.99)
Corner Brook	UU	29.16 (33.60)	-	-	0.3 (0.50)	1.44 (0.88)
Gander	PU	87.21 (4.65)	18.71 (3.33)	15.2 (8.05)	2.1 (0.88)	3.2 (1.40)
Gander	TU	74.62 (22.21)	22.72 (9.04)	10.91 (1.99)	1.1 (0.99)	2.1 (1.10)
Gander	UU	0.37 (0.66)	-	-	0.2 (0.42)	0.5 (0.51)
CBS	PU	87.78 (6.78)	23.11 (7.64)	13.64 (5.42)	3.38 (1.1)	2.0 (1.20)
CBS	TU	68.70 (37.26)	27.21 (13.14)	8.08 (2.32)	1.2 (0.79)	1.6 (0.84)
CBS	UU	3.28 (4.53)	-	-	0.1 (0.33)	1.22 (0.67)
St. John’s	PU	86.58 (5.20)	18.62 (3.23)	15.45 (7.28)	2.6 (0.70)	3.7 (1.64)
St. John’s	TU	72.27 (24.91)	30.71 (23.67)	8.78 (4.72)	0.6 (0.73)	2.11 (0.93)
St. John’s	UU	3.02 (4.47)	-	-	0.1 (0.35)	1.13 (0.35)

Table 2.4: Summaries of GLM results, comparing stand and site characteristics between urban centres and site types. Parameter estimates are shown with standard error in parentheses. Estimates significant at α=0.05 are bolded.

Coefficients	**Parameter Estimates**				
	Canopy Cover (%)	**DBH (cm)**	**Height (m)**	**Overstory Richness**	**Understory Richness**
(intercept)	**81.07 (5.95)**	**26.88 (4.12)**	**13.36 (1.72)**	**4.00 (0.26)**	**2.70 (0.38)**
TU	0.21 (8.41)	**15.89 (5.83)**	-3.10 (2.31)	**-3.60 (0.37)**	-0.80 (0.54)
UU	**-51.91 (8.64)**	-	-	**-3.67 (0.38)**	-1.26 (0.56)
CBS	6.71 (8.92)	-3.768 (6.19)	0.28 (2.43)	-0.63 (0.39)	-0.70 (0.57)
Gander	6.14 (8.41)	-8.17 (5.83)	1.84 (2.52)	**-1.90 (0.37)**	0.40 (0.54)
St. John’s	5.51 (8.41)	-8.26 (5.83)	2.09 (2.31)	**-1.40 (0.37)**	1.00 (0.54)
TU : CBS	-19.29 (12.27)	-11.79 (8.50)	-2.45 (3.26)	**1.43 (0.54)**	0.40 (0.79)
UU : CBS	**-32.59 (12.58)**	-	-	0.40 (0.56)	0.48 (0.81)
TU : Gander	-12.79 (11.90)	-11.88 (8.25)	-1.19 (3.32)	**2.60 (0.53)**	-0.30 (0.77)
UU : Gander	**-34.93 (12.06)**	-	-	**1.77 (0.53)**	-1.44 (0.78)
TU : St. John’s	-14.53 (12.06)	-3.80 (8.36)	-3.57 (3.21)	**1.56 (0.53)**	-0.79 (0.78)
UU : St. John’s	-31.66 (12.42)	-	-	1.19 (0.55)	-1.32 (0.80)

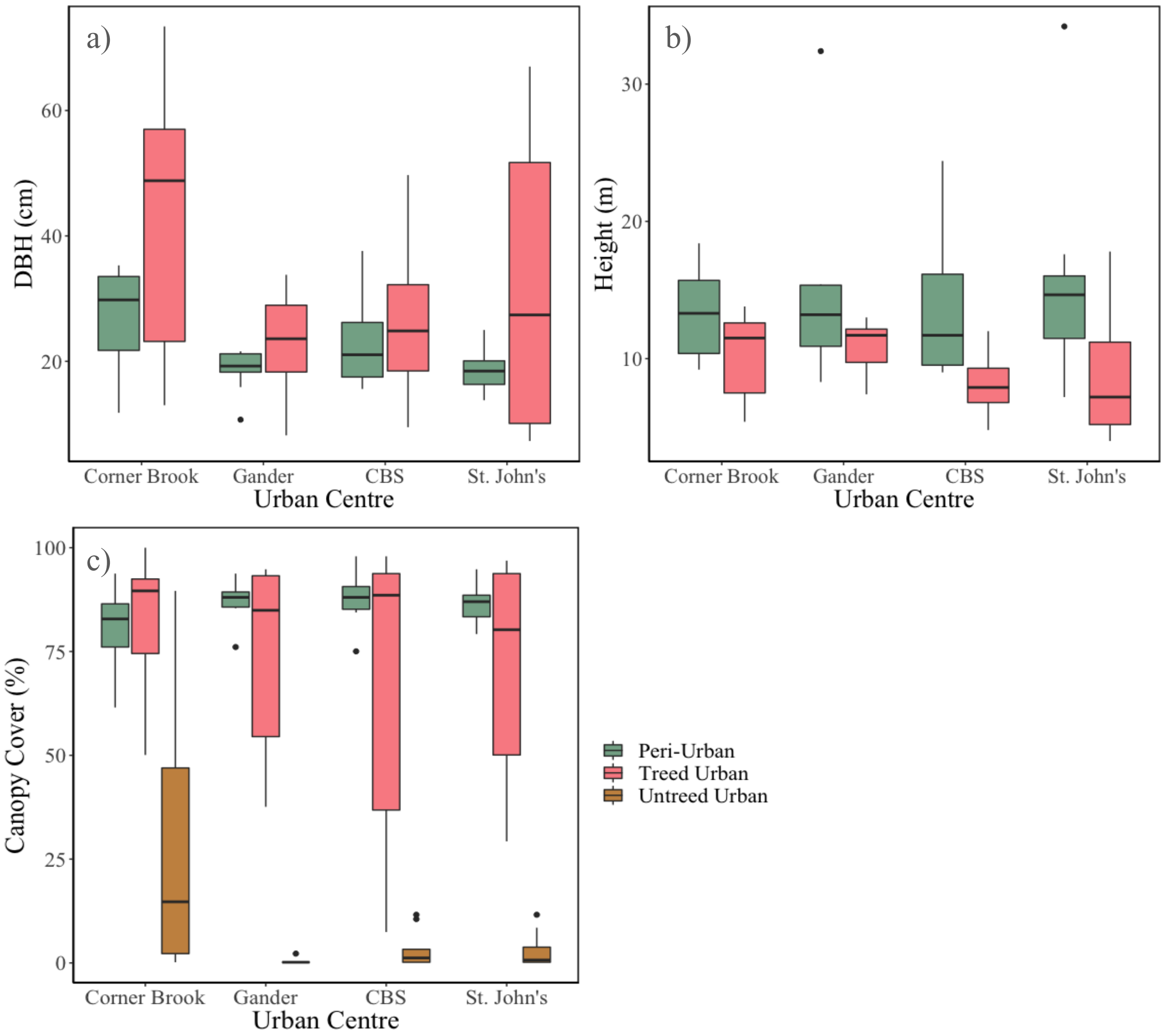

Figure 2.9: Boxplots depicting stand and site characteristics observed in peri-urban, treed urban, and untreed urban site types in four urban centres for a) diameter at breast height (cm), b) height (m), and c) canopy cover (%). The line in the centre of the boxes is the median of that site type and the box encompasses the 25-75% quartiles. The whiskers extending beyond the boxes represent the 95% quartiles, and extreme observations are points beyond the whiskers. Results of GLM summarised in Table 2.3.

Species richness is a commonly used measure of species diversity, which is a major contributor to an ecosystem's ecological integrity (Steedman and Haider 1993; Lindenmayer and Franklin 2002; Ordóñez and Duinker 2012; Rempel *et al.* 2016). The species richness in TU sites ranged from 0.4 ± 0.97 (Corner Brook) to 1.2 ± 0.79 (CBS) in the overstory, and from 1.6 ± 0.84 (CBS) to 2.11 ± 0.93 (St. John's) in the understory (Table 2.3). Both overstory and understory species richness was generally higher in TU sites than in UU sites, and lower than in PU sites (Figure 2.10).

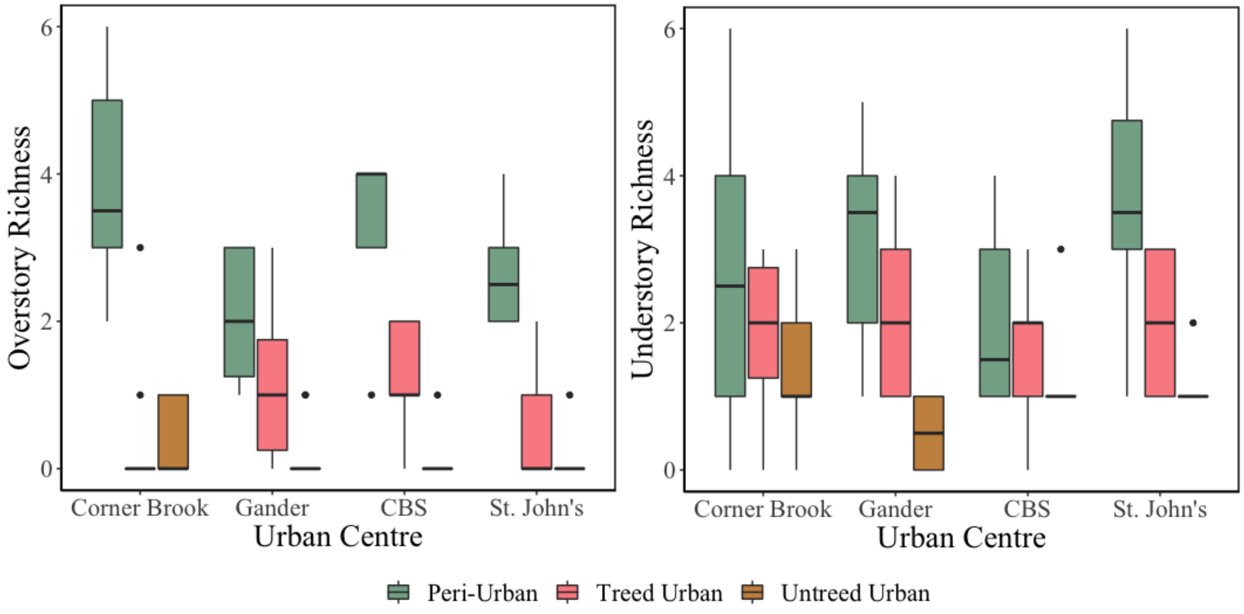

Figure 2.10: Boxplots depicting species richness observed in peri-urban, treed urban, and untreed urban site types in four urban centres for a) overstory and b) understory vegetation. The line in the centre of the boxes is the median of that site type and the box encompasses the 25-75% quartiles. The whiskers extending beyond the boxes represent the 95% quartiles, and extreme observations are points beyond the whiskers. Results of GLM summarised in Table 2.3.

2.3.1.2 Site type community characteristics

Ordination diagrams show similarities (or dissimilarities) in communities based on the distance between points: points closer together are more similar in their composition. Our NMDS ordination revealed that the ground cover of each PU, TU, and UU site contain similarities to each other, even in different urban centres (Figure 2.11a). This pattern suggests that, for example, the ground cover in Botanical Gardens (St. John’s PU) is similar to the ground cover in Bowater Park (Corner Brook PU), and the ground cover at Memorial University campus (St. John’s UU) is similar to the ground cover in downtown Corner Brook (UU). However, there are areas of overlap between site types. Notably, some TU sites blended components of both PU (e.g., coarse woody debris (CWD), rock, bare ground) and UU ground cover (e.g., grass), resulting in TU sites demonstrating the widest variation of all three site types.

Similar to the stand and site characteristics, vegetation associations – represented by vectors overlain on the ordination diagram – indicate that ecological integrity is lowest in UU sites and highest in PU sites (Figure 2.11b). Here, the length and direction of the vector indicate the strength of the relationship between the vegetation group and site type. Vegetation indicators that are commonly and traditionally associated with high ecological integrity, including high proportions of native species and increased species richness in both the overstory and understory, are highly associated with PU and some TU sites. Non-native species are more associated with UU sites and TU sites that have similar composition to UU sites. The presence of coniferous species, which are most common in intact boreal forest stands in Newfoundland, also increase as we move into PU areas, while deciduous species are most commonly present in TU sites.

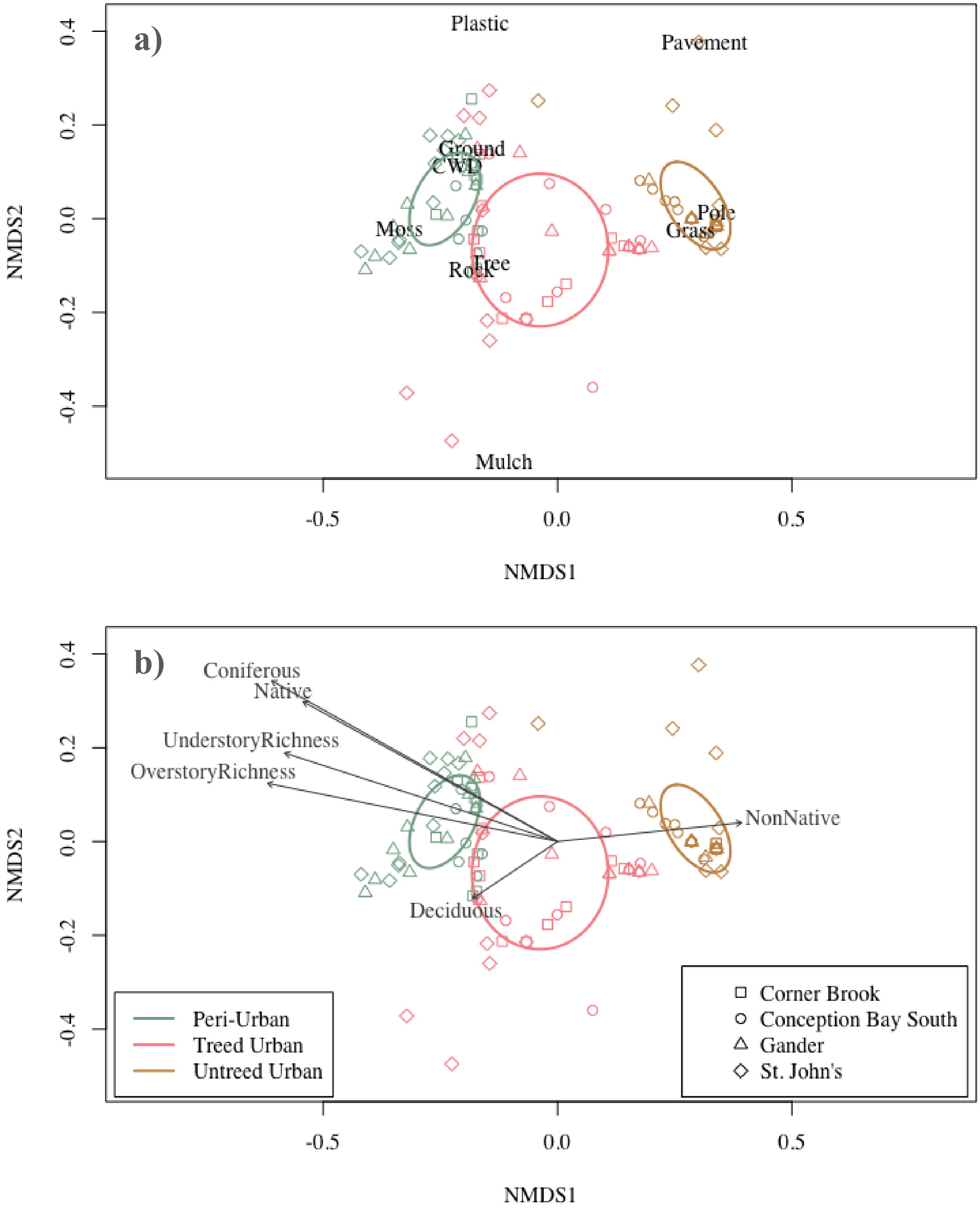

Figure 2.11: NMDS ordination of ground cover types (ground cover types represented in plot a) comparing all urban centres and site types (stress = 0.1104 in 2 dimensions). Colours represent site types and shapes represent urban centres. Ellipses are 95% confidence intervals around the centroids of each site type. Vegetation categories (b) depict how overstory and understory vegetation are associated with the three site types, where the direction and length of the arrow indicate the strength of the relationship.

Looking at each urban centre individually, they each follow similar patterns when compared to the overall NMDS: i) site types are mainly characterized by their ground cover, and ii) of the three site types, TU and PU demonstrate the greatest similarities (Figure 2.12, Figure 2.13, Figure 2.14, Figure 2.15). Some urban centres, however, display this relationship more than others; e.g. Corner Brook shows very little variation in the ground cover of UU, while CBS has much higher variation in ground cover of TU sites than UU and PU. The associations between native and non-native overstory and understory vegetation indicate that native species have a higher presence in PU sites, while non-natives have a higher presence in TU and some UU sites. Similarly, species richness in both the overstory and understory are generally more associated with PU sites and TU sites, but tend to have a stronger association with PU sites. These relationships between vegetation and site types suggest that ecological integrity decreased across the gradient from PU into UU environments in each urban centre.

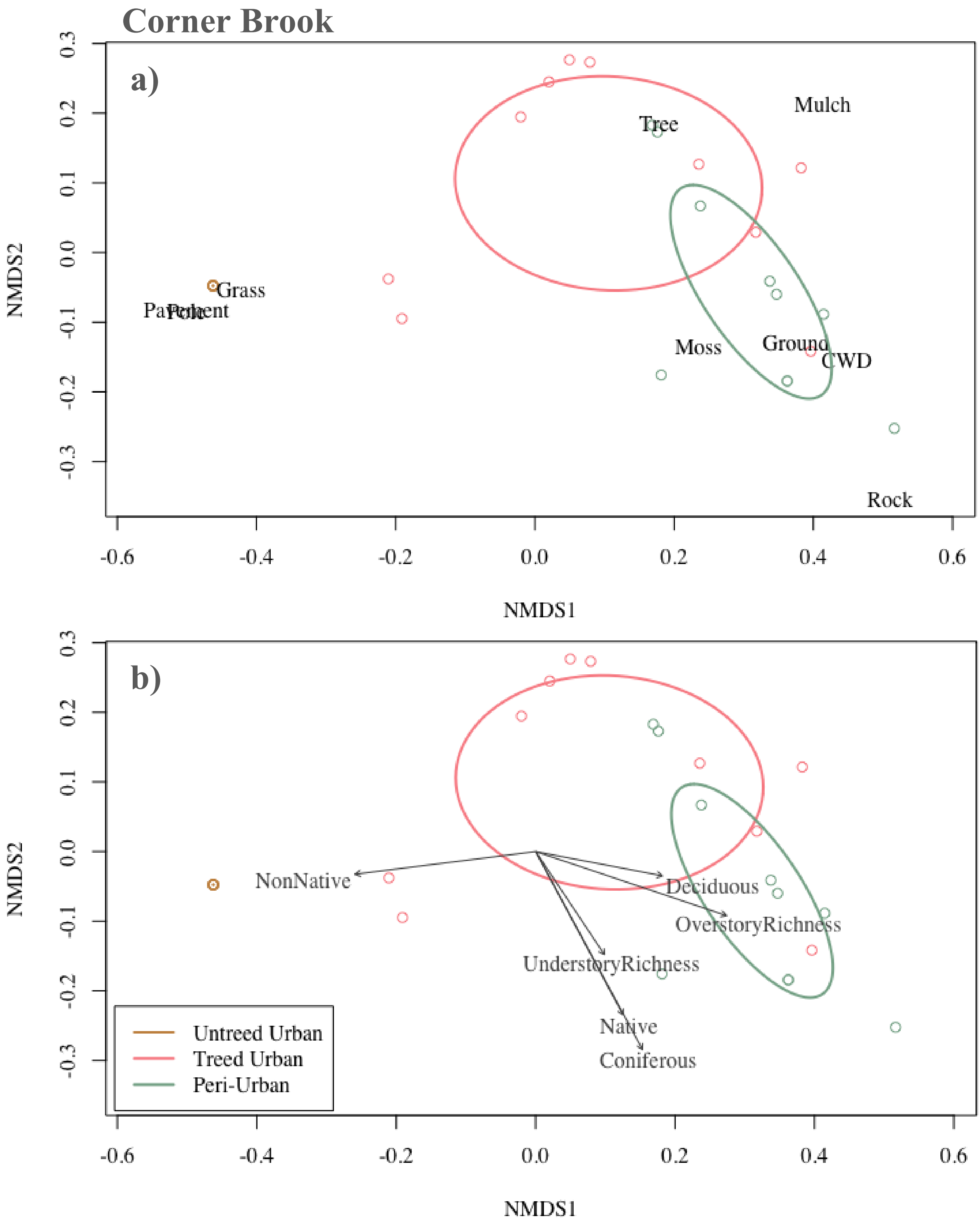

Figure 2.12: NMDS ordination of ground cover types (ground cover types represented in plot a) comparing three site types in Corner Brook (stress = 0.0327 in 2 dimensions). Ellipses are 95% confidence intervals around the centroids of each site type. Vegetation categories (b) depict how overstory and understory vegetation are associated with the three site types, where the direction and length of the arrow indicate the strength of the relationship.

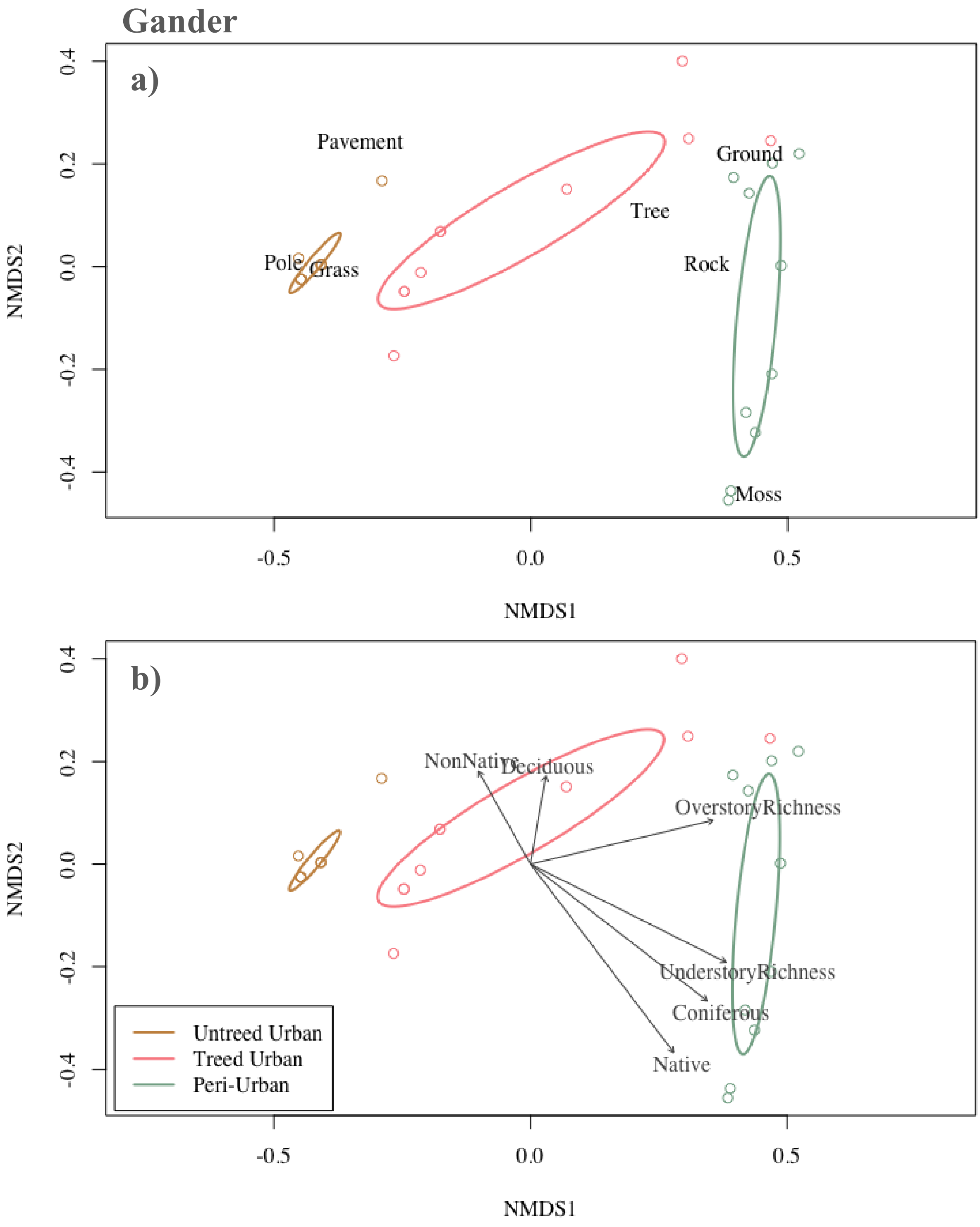

Figure 2.13: NMDS ordination of ground cover types (ground cover types represented in plot a) comparing three site types in Gander (stress = 0.0566 in 2 dimensions). Ellipses are 95% confidence intervals around the centroids of each site type. Vegetation categories (b) depict how overstory and understory vegetation are associated with the three site types, where the direction and length of the arrow indicate the strength of the relationship.

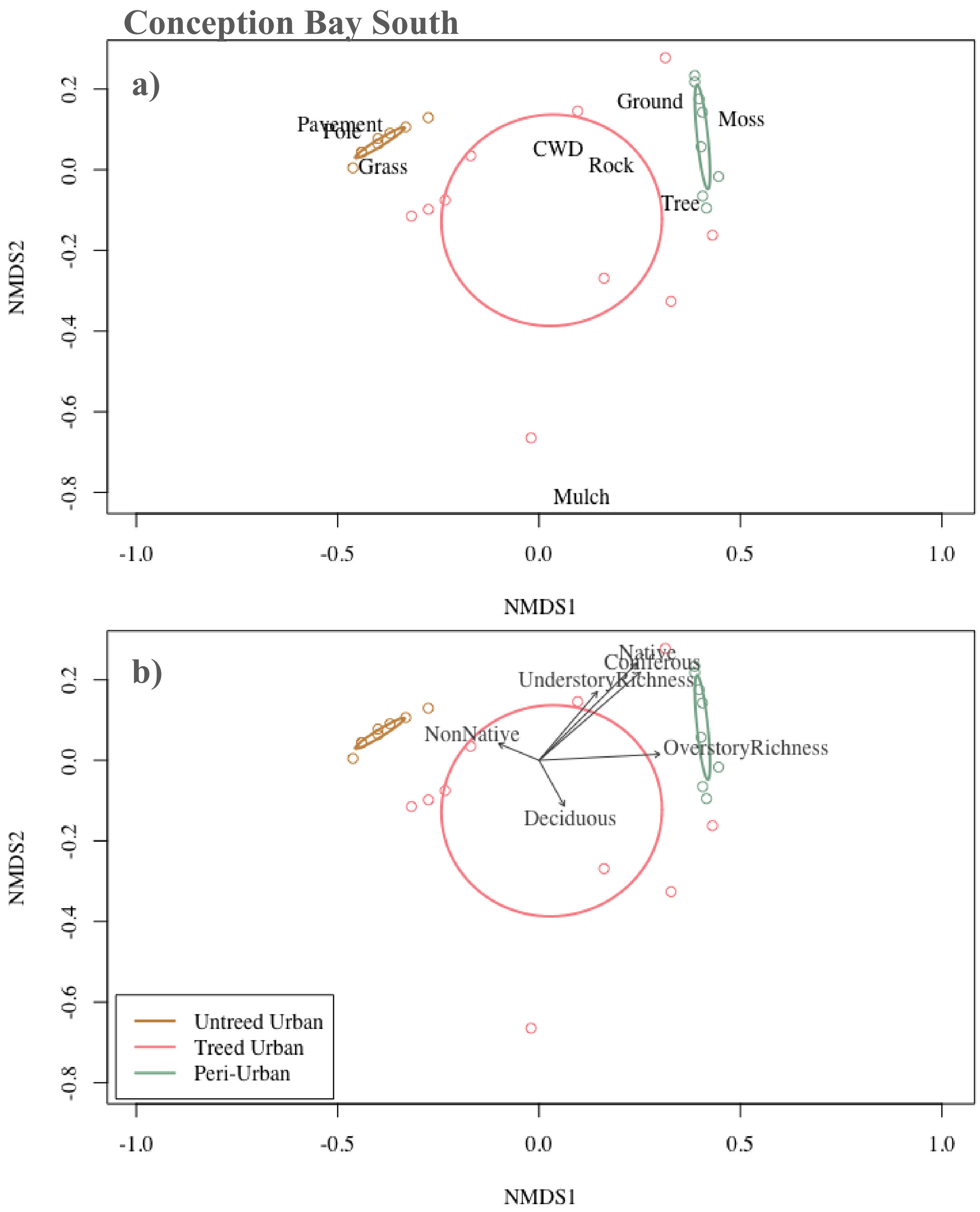

Figure 2.14: NMDS ordination of ground cover types (ground cover types represented in plot a) comparing three site types in Conception Bay South (stress = 0.0675 in 2 dimensions). Ellipses are 95% confidence intervals around the centroids of each site type. Vegetation categories (b) depict how overstory and understory vegetation are associated with the three site types, where the direction and length of the arrow indicate the strength of the relationship.

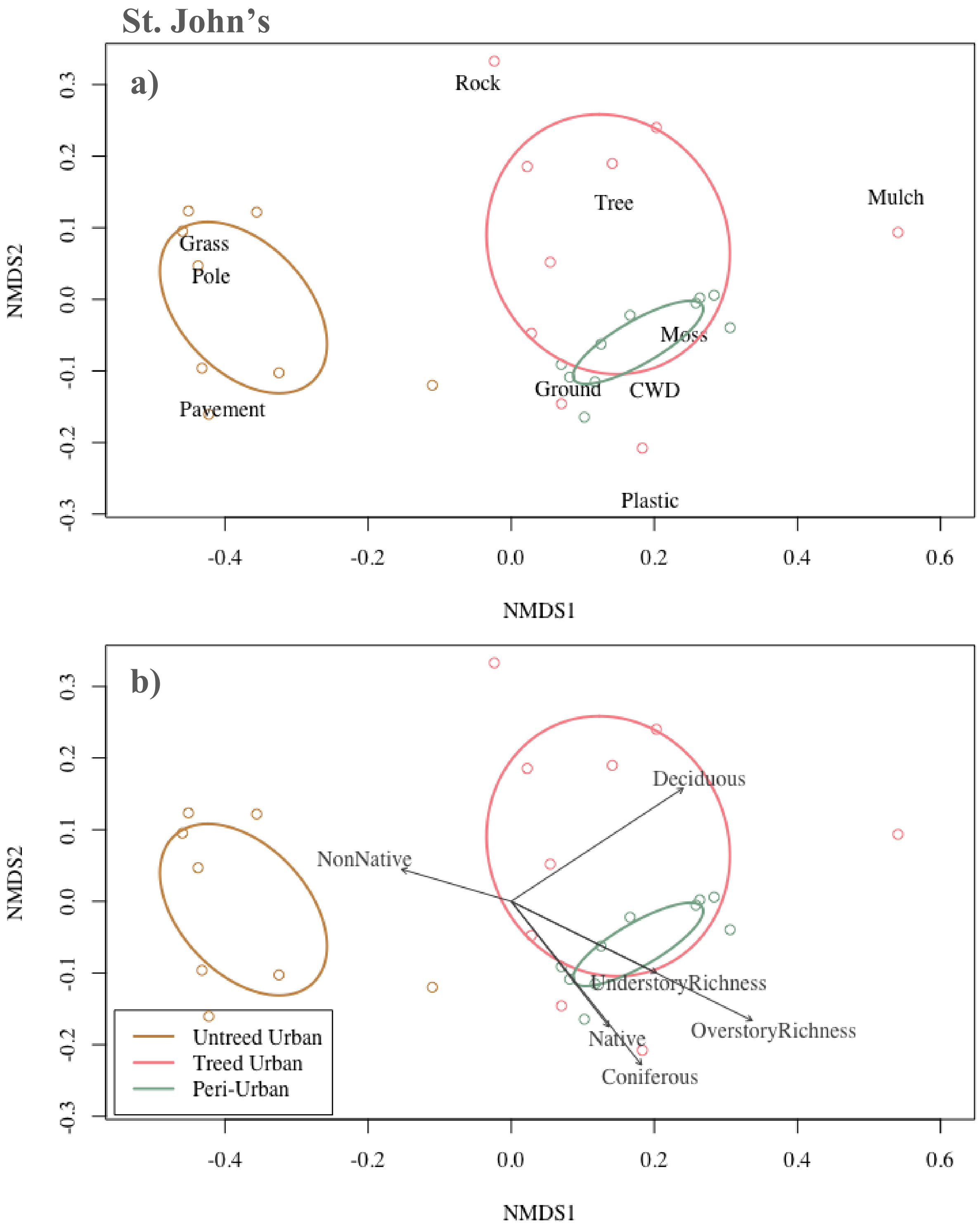

Figure 2.15: NMDS ordination of ground cover types (ground cover types represented in plot a) comparing three site types in St. John's (stress = 0.0823 in 2 dimensions). Ellipses are 95% confidence intervals around the centroids of each site type. Vegetation categories (b) depict how overstory and understory vegetation are associated with the three site types, where the direction and length of the arrow indicate the strength of the relationship.

2.3.1.3 Urban centre community characteristics

The ground cover within PU sites were composed exclusively of natural surfaces (bare ground, rock, moss, CWD), though some urban centres had higher presence of some ground cover types than others (Figure 2.16). For example, St. John's had high composition of CWD and moss, while CBS had high composition of bare ground, rock, and trees. Deciduous species in PU sites were strongly associated with Corner Brook and CBS, leading to higher overstory species richness, while PU sites in St. John's and Gander had higher compositions of coniferous species (Appendix IV). Non-native species in PU sites were also strongly associated with Corner Brook and CBS while natives were more common in St. John's and Gander. There were more instances of non-natural surfaces, like mulch and plastic, in the TU sites, particularly in sites located in St. John's (Figure 2.17). Within these site types, St. John's additionally had the highest variation in ground cover types. The area with most overlap between urban centres were associated with ground cover types composed of natural surfaces (i.e., trees, grass, moss, ground). Overstory species richness and both deciduous and coniferous species were strongly associated with Gander and CBS in TU sites, while understory richness was strongly associated with St. John's in both PU and TU sites. St. John's additionally had strong associations with both native species (in PU sites) and non-native species (in TU sites).

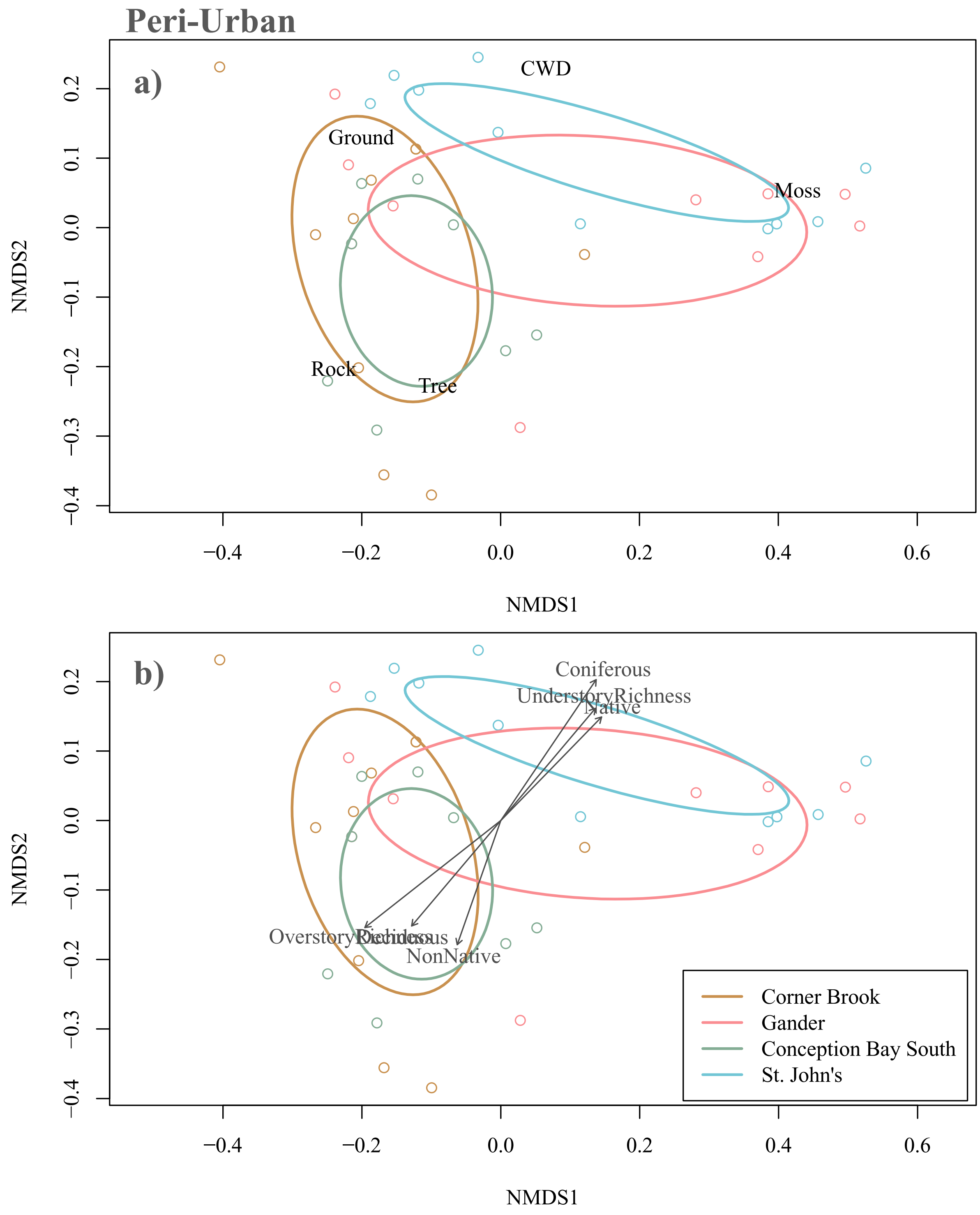

Figure 2.16: NMDS ordination of ground cover types (ground cover types represented in plot a) comparing peri-urban environments in four urban centres (stress = 0.0709 in 2 dimensions). Ellipses are 95% confidence intervals around the centroids of each urban centre. Vegetation categories (b) depict how overstory and understory vegetation are associated with the four urban centres, where the direction and length of the arrow indicate the strength of the relationship.

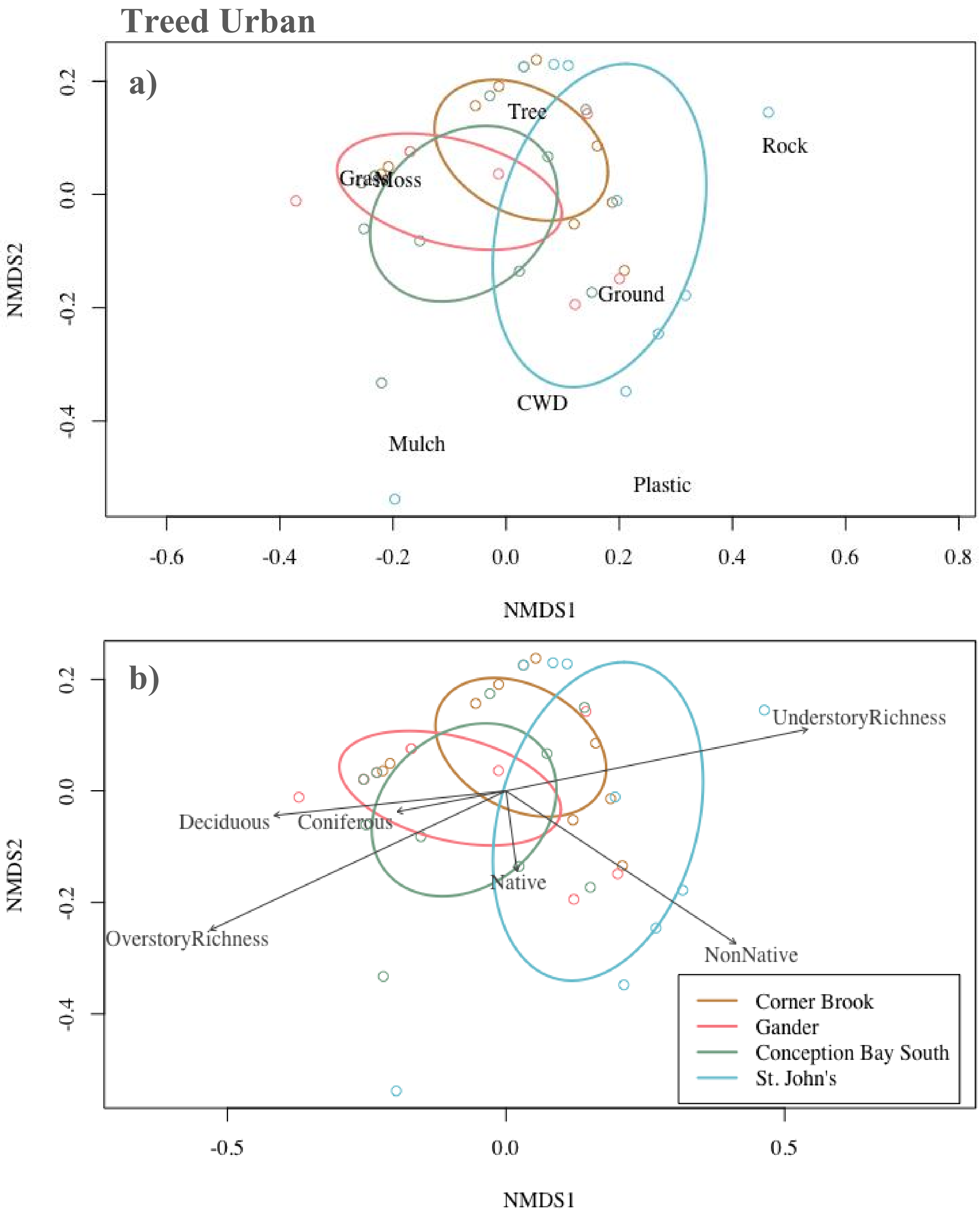

Figure 2.17: NMDS ordination of ground cover types (ground cover types represented in plot a) comparing treed urban environments in four urban centres (stress = 0.0709 in 2 dimensions). Ellipses are 95% confidence intervals around the centroids of each urban centre. Vegetation categories (b) depict how overstory and understory vegetation are associated with the four urban centres, where the direction and length of the arrow indicate the strength of the relationship.

2.3.2 Climate mitigation results

During the summer months, we are able to clearly observe the benefit of urban forest canopy cover. While the average summer air temperature appeared to display little change between sites – only ranging from 5.5°C (PU) to 6.5°C (UU) (see Table 2.5 for summary statistics of summer climate indicators) – given that differences between urban and rural settings is 2.5°C (Ziter *et al.* 2019), the difference in average temperatures often appears insignificant since it masks the extremes. Other indicators of summer climate are also able to show the temperature increases as canopy cover lessens (Figure 2.18). GDD (base 5°C) increased, both in terms of number of GDD and variation in GDD, from PU to TU and finally to UU sites, where St. John's had the highest number of GDD between all four urban centres. Similarly, the number of days where temperatures were above 20°C and 25°C increased as canopy cover lessened (Figure 2.18). The increase between sites was constant when looking at temperatures above 20°C, however, this benefit was stronger between TU and UU sites when looking at temperatures above 25°C. Here, the greatest benefit was observed in St. John's, where the number of days above 25°C reduced by more than 50% between UU sites (77 days) and TU sites (31 days), and was reduced by more than 75% between UU sites and PU sites (17 days). Though there was little change in the average temperature resulting from increasing canopy cover, urban forests were successful in mitigating temperature extremes in the summer seasons.

Table 2.5: Summary statistics of summer (June, July, August) climate indicators. Values of each indicator are shown with averages and standard deviation calculated for each site type. Note that while days above temperature thresholds are measured as number of days in a year, growing degree days are a measure of temperature degree accumulation.

Site Type	City	Average Temperature (°C)	Growing Degree Days	Days Above 20°C	Days Above 25°C
Peri-Urban	Corner Brook	5.2	742.7	87	15
	Gander	4.6	676.4	91	24
	CBS	6.5	717.8	110	28
	St. John's	5.6	960.6	92	17
	Average (SD)	*5.5 (0.79)*	*774.4 (127.13)*	*95 (10.2)*	*21 (6.1)*
Treed Urban	Corner Brook	5.5	820.5	103	31
	Gander	5.2	786.3	116	38
	CBS	7.3	762.3	126	36
	St. John's	7.2	1391.1	117	31
	Average (SD)	*6.3 (1.12)*	*940.1 (301.65)*	*116 (9.5)*	*34 (3.6)*
Untreed Urban	Corner Brook	6.0	962.5	131	63
	Gander	5.1	802	113	40
	CBS	7.8	876.6	151	64
	St. John's	7.0	1808.9	143	77
	Average (SD)	*6.5 (1.21)*	*1112.5 (468.88)*	*135 (16.5)*	*61 (15.4)*

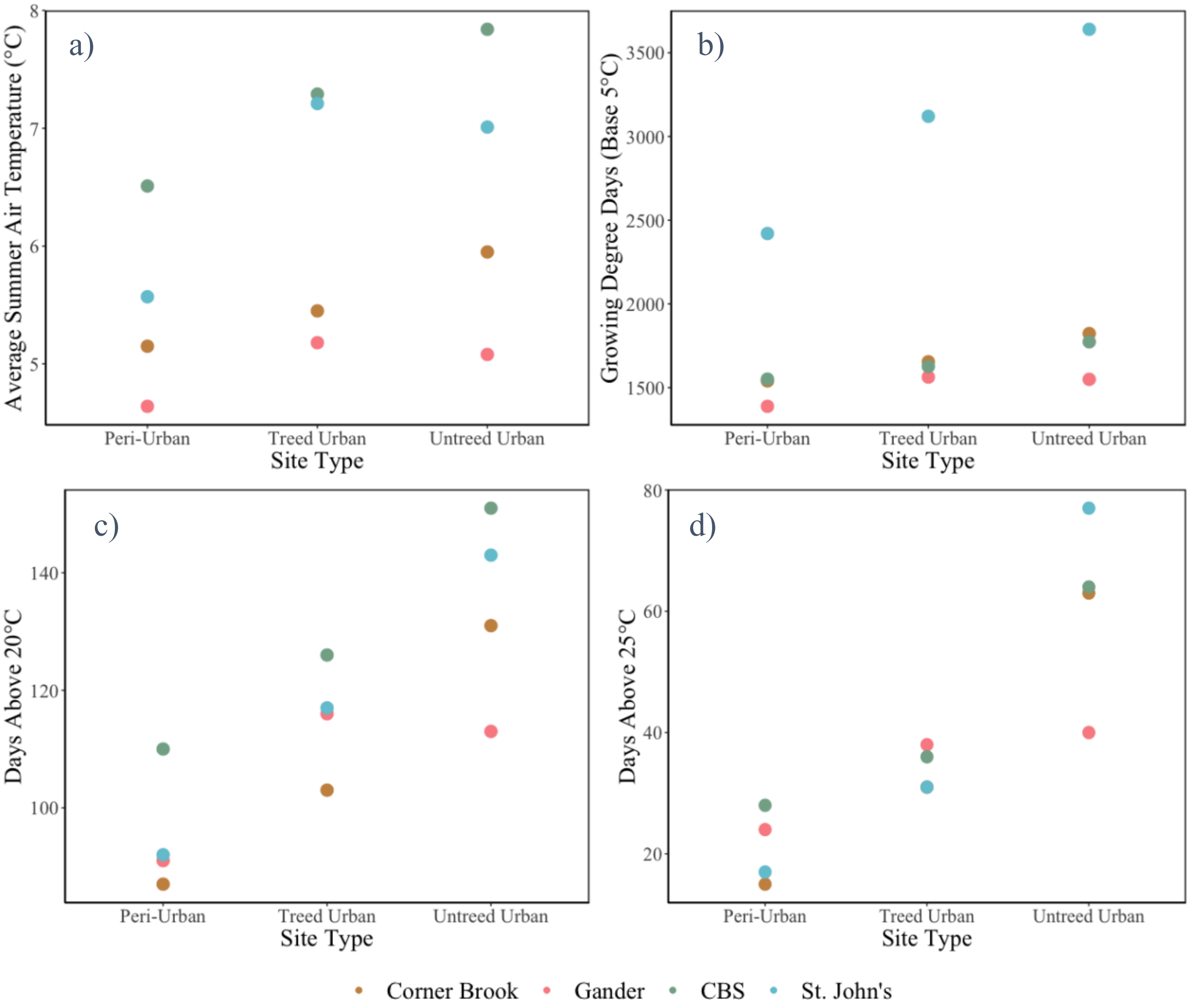

Figure 2.18: Scatterplots depicting indicators of summer climate observed in Corner Brook, Gander, CBS, and St. John's in three site types for a) average air temperature, b) growing degree days (base 5℃), c) days above 20℃, and d) days above 25℃. Summer season incorporated the months of June, July, and August 2021.

The benefits from urban forest canopy cover, however, were less obvious during the winter season. There was less variation in average temperature between site types in the winter than in the summer, ranging from -2.0°C in PU to -1.7°C in UU (see Table 2.6 for summary statistics of winter climate indicators), and remaining climate indices generally demonstrated inconsistent trends with canopy cover across urban centres. For example, when all study sites are examined together, the number of freeze-thaw events and ground freeze days generally increase as canopy cover is reduced (Figure 2.19), suggesting a net benefit from increased canopy cover.

However, freeze-thaw in St. John's and CBS – the two urban centres with traditionally high freeze-thaw cycles – don't follow this trend: TU sites in these two urban centres had the lowest number of freeze-thaw events, rather than the anticipated PU minimum. In addition to the number of ground freeze days increasing from PU to UU sites, the variation between urban centres was also lowest in PU sites. Aboveground freezing temperatures followed an opposing trend to freezing temperatures belowground, with UU sites having the least number of freezing days in most urban centres. Island-wide variation in the number of freezing days remained high in all three site types. Finally, the number of frost events was lowest and had the least variation in TU sites; on average the differences between site types was approximately 5-10 events.

Table 2.6: Summary statistics of winter (December, January, February) climate indicators. Values of each indicator are shown with averages and standard deviation calculated for each site type.

Site Type	**City**	**Average Temperature (°C)**	**Freeze-Thaw**	**Freeze Days**	**Ground Freeze Days**	**Frost Events**
Peri-Urban	Corner Brook	-3.0	113	82	66	32
	Gander	-3.0	130	81	96	38
	CBS	-0.5	151	27	62	47
	St. John's	-1.4	154	57	91	41
	Average (SD)	*-2.0 (1.25)*	*137 (17.2)*	*62 (25.9)*	*79 (17.2)*	*40 (6.2)*
Treed Urban	Corner Brook	-2.9	125	63	203	30
	Gander	-2.7	159	54	132	34
	CBS	-1.4	122	27	58	31
	St. John's	-1.4	131	24	93	33
	Average (SD)	*-2.1 (0.82)*	*134 (16.9)*	*42 (19.4)*	*122 (62.2)*	*32 (1.8)*
Untreed Urban	Corner Brook	-2.5	133	49	190	36
	Gander	-2.8	162	67	201	38
	CBS	-0.9	135	23	135	39
	St. John's	-0.5	181	24	50	46
	Average (SD)	*-1.7 (1.14)*	*153 (23.0)*	*41 (21.2)*	*144 (69.0)*	*40 (4.3)*

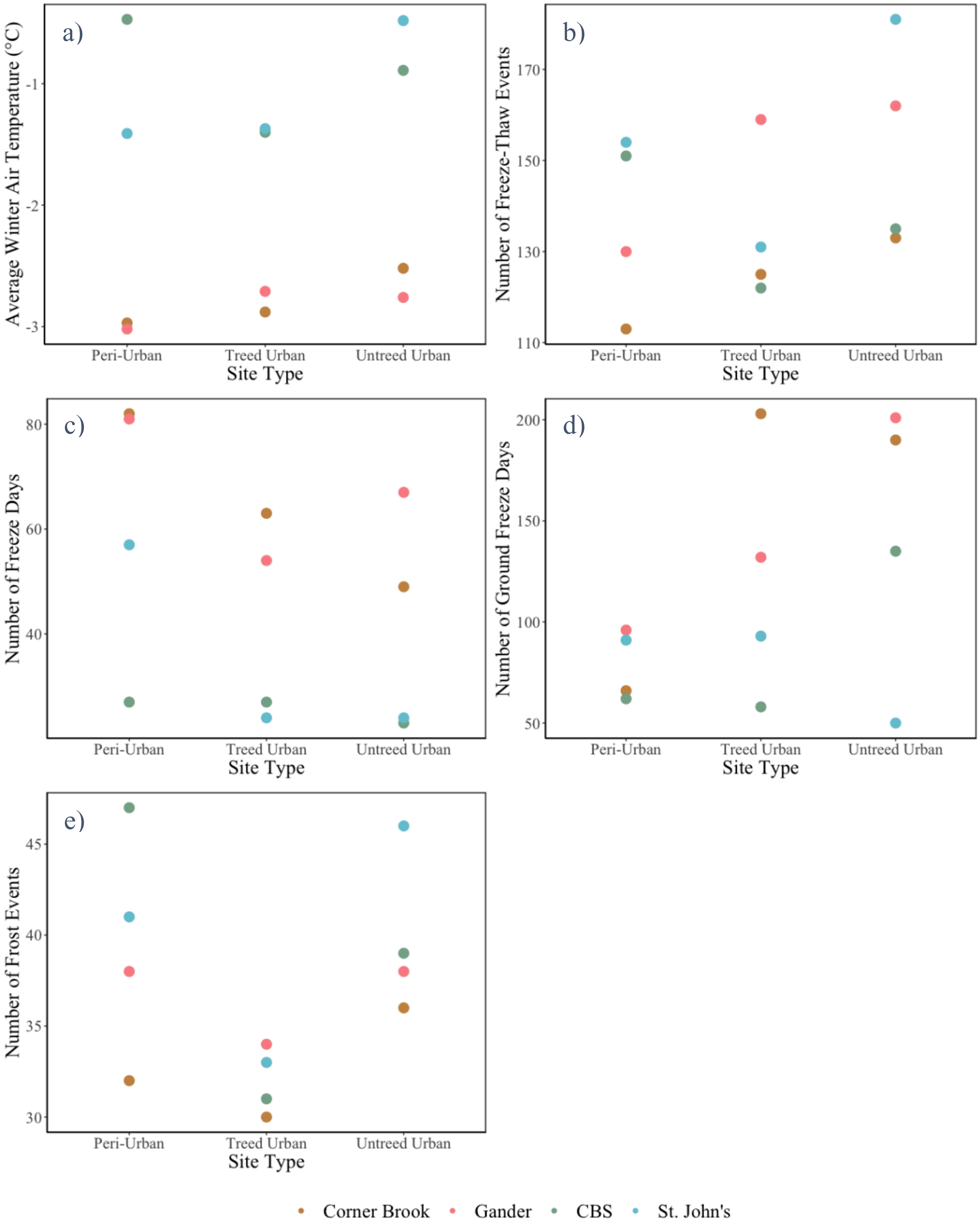

Figure 2.19: Scatterplots depicting indicators of winter climate observed in Corner Brook, Gander, CBS, and St. John's in three site types for a) average air temperature, b) number of freeze-thaw events, c) number of freeze days, d) number of ground freeze days, and e) number of frost events. Winter season incorporated the months of December, January, and February 2020-2022.

2.4 Discussion

2.4.1 Characterization of urban forests on the island of Newfoundland

Our aim was to characterize urban forests on the island of Newfoundland, and interpret them in the context of relatively natural forests and untreed urban areas. We found that Newfoundland's urban forests share few characteristics with intact boreal forest stands located within or adjacent to urban centres. Species composition differed between peri-urban (PU) and treed urban (TU) sites, where PU generally contained higher species richness, in both the overstory and the understory, than their associated TU sites. While the increased structural diversity in TU sites suggests higher ecological integrity (Lindenmayer and Franklin 2002), they are not reflective of more natural intact boreal forest stands present on the island of Newfoundland which are more homogeneous than many other forest ecosystems. We discuss the potential mechanisms and implications of these findings in detail, below.

The total number of overstory species observed in TU sites in Gander and Conception Bay South was higher than in PU sites (Appendix IV), though species richness was still higher in PU at the plot level. This pattern is likely because PU sites were located in larger homogeneous forest stands, each of which contained generally the same number of species, while TU sites were more variable. The type of species found in TU sites also varied from those found in PU. Within intact boreal stands, there is generally a single species that creates the dominant canopy in mature stands (Thorpe 1992); on the island of Newfoundland the main species is balsam fir, with black spruce occupying the role in central regions of the island where the ecosystem is driven by wildfires (Damman 1983). This stand dominance is reflective of the most common species found in PU sites: balsam fir in each urban centre, except in Gander where black spruce was the most common species. Coniferous dominance in PU sites is very different from the most common species observed in TU sites, where deciduous species dominate. Non-native Norway maple (*Acer plantanoides*) was the most common species observed in TU sites of both Corner Brook and Conception Bay South, and had only one less observation than the most common species observed in St. John's, golden chain (*Laburnum spp.*). Further, any coniferous species that were encountered in TU sites each had only one observation, with no coniferous species in TU sites in St. John's. This composition is a stark contrast to PU sites on the island of Newfoundland, which were primarily composed of coniferous species. Similarly, non-native species were highly associated with TU sites, almost exclusively composing the observed overstory canopy in Corner Brook, Conception Bay South, and St. John's. This is, again, in stark contrast to PU sites where no non-native species were present in the overstory, despite these sites having a much higher number of individual trees than their paired TU sites.

Higher density of individuals created closed canopy forests in PU sites. Given that PU sites were located in large homogenous forest stands, there was little variation in canopy cover. TU sites, however, had much lower species density, which leads to a more open canopy in many instances, though this varied both within and between sites. Closed canopy homogenous forest stands also lead to more uniformity in the diameter and height of trees (Wilson 1984), of which there was little variation in PU sites. Diameter was more variable in TU sites, often greatly exceeding the maximum DBH of trees in PU sites. There was little variation in the height of trees in both TU and PU sites, however, trees were often taller in PU areas. In closed canopy environments, like intact boreal forest stands, trees are primarily competing for light, and therefore allocate their resources to primary growth (Wilson 1984). In open grown environments, or those with low canopy cover, trees have more access to light and therefore can more evenly allocate resources to both primary and secondary growth, leading to shorter trees with a more tapered shape (Gray 1956; Muhairwe 1993; Rhoades and Stipes 1999; McHale *et al.* 2009). This significantly alters the size of both diameter and height between trees in open and closed canopy environments. Similarly, many trees in TU sites are not native to the region, and therefore do not reach their full height potential due to limited resources and less than ideal growing conditions for their needs (Rhoades and Stipes 1999; McHale *et al.* 2009).

While structural diversity is an indication of increased ecological integrity in many circumstances (Wicklum and Davies 1995), it may not play a significant role in determining the ecological integrity of urban forests in Newfoundland. The diversity in structure, while important for creating new habitats and occupying various ecological functions (Lindenmayer and Franklin 2002), may simply be a by-product of the nature of urban forests and a lack of coordinated planning. In many instances on the island of Newfoundland, trees on properties are chosen and planted by residents with little guidance or consideration of the larger urban forest ecosystem. Urban forests generally have an open canopy, and planting locations and years are dictated by residents, which leads to diverse age classes, and variable diameters resulting from more tapered trees (Muhairwe 1993). Some municipalities have by-laws in place to ensure property owners contribute to the urban forest (e.g., Town of Conception Bay South 2011; City of St. John’s 2018), leading to a complex matrix of urban forest canopy cover throughout the urban centre. More natural intact boreal forest stands in Newfoundland, however, are driven by large-scale disturbance cycles – primarily insect outbreak and wildfire – which leads to more homogenous forest stands with little diversity in age and structure (Engelmark 1999).

Species selection is also often dictated by residents in urban centres of Newfoundland, which often leads to low diversity of both species and provenance, creating large vulnerabilities in the longevity of the urban forests. Local climates and ongoing climate change also impact the vulnerability of these urban forests that are primarily composed of non-native deciduous species. Winter climates in Newfoundland are in many respects harsher than in other mainland regions, with heavy snow loads, frequent freeze-thaw cycles (Banfield 1983), and high winds. Coniferous boreal forest species native to Newfoundland – including balsam fir and black spruce – are well adapted to withstand challenges associated with winter climates on the island (Messaoud *et al.* 2007a, b, 2019). However, many non-native deciduous species in urban forests are at the northern edge of their temperature ranges and experience damage and losses during the winter (Schlaepfer *et al.* 2020) due to wind, snow, and ice. Shallow rooting depths, high winds, and an increasingly lengthened growing season in Newfoundland further compound this issue, as trees that are not evolved to withstand these extremes will be more likely to have full foliage during seasons with extreme wind events and sustained high winds, with limited abilities to anchor themselves (Wiersma *et al.* 2012).

In contrast, non-native species in urban forests are often better equipped than natives to take advantage of the ongoing lengthening of the growing season. Phenological patterns in trees are dictated by complex interactions between chilling (cool temperatures between autumn and late winter), forcing (warm temperatures between late winter and early spring), and photoperiod (Flynn and Wolkovich 2018). There is less plasticity in the ability of species to change the timing of "starting" and "ending" the growing season, despite favourable conditions. Therefore, species that are adapted to regions with a longer growing season (i.e., non-native species) may be better equipped to take advantage of the increasingly favourable growing conditions (Davidson *et al.* 2011; Esperon-Rodriguez *et al.* 2020). The increased plasticity of non-native species does, however, create issues in PU environments: non-native species begin to reproduce on their own and encroach on nearby "natural" spaces, dominated by native species (Paquette *et al.* 2012).

Their ability to take advantage of a longer growing season and increased resources means they often become invasive in these areas, out-competing native species (Ehrenfeld 2010; Vitule *et al.* 2012). Given that our study locations in PU sites were often located further into the natural forests, we didn't observe high amounts of non-native species in PU sites. However, encroachment of Norway maple has been anecdotally observed in PU areas surrounding the St. John's metro region by urban forest managers, botanists and conservation biologists, and residents.

2.4.2 City-specific urban forest characteristics

Looking further into the urban forests in Newfoundland, we found that stand and site characteristics and species compositions differed in each urban centre. Given the variety in both ecoregion characteristics and climate regimes across the island (Banfield 1983; Damman 1983), we expected urban forests to differ from each other. Many major differences in species compositions between PU sites were driven by the ecology of the region: as previously mentioned, PU sites in the dry, fire-driven forests surrounding Gander were primarily dominated by black spruce, while PU sites in other urban centres were primarily dominated by balsam fir. The favorable growing conditions in the Western Newfoundland Ecoregion and valleys of the Maritime Barrens Ecoregion (Damman 1983) likely also contributed to the higher presence of deciduous species and increased species richness in PU sites of Corner Brook and CBS. However, the size and amount of use in PU sites may also influence the species composition, primarily when considering the abundance of non-native species. The location chosen as a PU site in Gander is a set of cross-country ski trails primarily used in the winter season, with limited accessibility in the spring, summer, and fall. Similarly, the location chosen as a PU site in St. John's does not have free entry as many other parks and trails in the city do, which likely limits use by the public. PU sites in these two urban centres had the lowest abundance of non-native species, indicating that the increased use of PU sites in Corner Brook and CBS may create more opportunities for non-native species to colonize in these areas.

As the largest and oldest urban centre in Newfoundland, the urban landscape of St. John's is more developed than others in this study, with a longer period in which anthropogenic changes to the landscape have occurred. This history may have facilitated high presence of ground cover types that would not occur in natural environments (e.g., plastic, mulch, pavement, fence) and an increased number of ornamental and non-native understory species, leading to increased understory species abundance in St. John's TU sites. In the overstory, however, species richness was highest in Gander and CBS, while the overstory of St. John's and Corner Brook had a more uniform species composition. While this reflects the uniformity of intact boreal forest stands throughout Newfoundland, those in TU sites in St. John's and Corner Brook were primarily non-native deciduous species, introducing potential vulnerabilities to the urban ecosystem. The urban forest of St. John's, for example, is primarily composed of Norway and sycamore maple (*Acer pseudoplantanus*), with 50% of the urban forest population composed of these non-native maple species (City of St. John's 2006). The homogeneity of these stands creates a significant vulnerability for the urban forest of St. John's should a damaging insect or disease reach the city, as it has in the past with the elm spanworm (*Ennomos subsignaria*; City of St. John's 2006). Similar losses of significant proportions of urban forests have occurred in cities with forests primarily composed of ash (*Fraxinus spp.*) and elm (*Ulmus spp.*) due to the emerald ash borer (*Agrilus planipennis*) and Dutch elm disease (*Ophiostoma ulmi*), respectively, in other regions of North America (Arnberger *et al.* 2017; Bajeux *et al.* 2020).

2.4.3 Microclimate mitigation by urban forests in Newfoundland

Our aim was to determine the level of mitigation of temperature-related climate change impacts by urban forests, in the context of relatively natural forests and untreed urban areas. The background climate conditions of Newfoundland (i.e., the conflicting influences of the Gulf Stream and Labrador Current) and urban context (e.g., relatively small urban centres and low urban heat island effects) shape the potential benefits of urban forests, while also influencing their capacity to address stakeholder needs. It is necessary to consider these carefully when designing urban forest strategies to address current and anticipated climate concerns. Here, we found that even in the context of the climate conditions in Newfoundland, urban forests provide temperature mitigation services, while other factors may dominate microclimatic influences on winter temperatures. Below, we discuss in detail the potential mechanisms and implications of these temperature mitigation patterns by urban forests.

During the summer season, urban trees proved capable of reducing the microclimatic influence of urban infrastructure, creating a microclimate that more closely resembles intact boreal forest stands (PU sites) than UU sites. Urban forest canopy cover reduced the overall average summer temperature between site types by 1°C, and mitigated temperature extremes, measured as the frequency of days with temperatures above 20°C and 25°C. Urban trees reduced the number of hot days in both TU and PU sites compared to UU sites, with substantial reduction of days above 25°C. The impact was notable in TU sites (e.g., a reduction of 14 days in St. John’s), but often much more pronounced in PU areas (e.g., a reduction of 60 days in St. John’s). There was a proportionally higher reduction of hot days when assessing the warmer temperature of 25°C compared to 20°C. This finding is potentially useful; although very hot days (>30°C) are uncommon in Newfoundland’s present climate, this is expected to increase under ongoing climate change: the maximum average number of very hot days in the central Newfoundland region (area of most change on the island) will increase from 1.2 days (1976-2005) to 11.5 days (2051-2080) under RCP 8.5 (Pacific Climate Impacts Consortium 2019). As the climate continues to warm, hot and very hot days become more common, these mitigations in temperature become increasingly important.

The number of growing degree days (GDD) is a metric determined by accumulated temperatures above a set temperature (5°C is most commonly used in boreal-temperate forests; Sykes and Prentice 1996; Messaoud *et al.* 2007a, 2019) during the growing season. Here, GDD was highest in UU sites and lowest in PU sites. In many cases, increasing the number of GDD is beneficial to trees, as it indicates an earlier start to the growing season, which has been found to increase cone production and create favourable habitat for more southerly species (Sykes and Prentice 1996; Messaoud *et al.* 2019). However, not all trees have the plasticity to adjust their growing season. Given that this plasticity is more commonly associated with non-native invasive species, an increase in GDD has the potential to improve conditions for (and, thereby presence of) invasives (Davidson *et al.* 2011; Vitule *et al.* 2012; Esperon-Rodriguez *et al.* 2020), while rendering habitat less suitable for native species (Sykes and Prentice 1996). Increasing GDD also reflects various environmental stresses. For example, higher GDD have been linked to an increased occurrence of droughts and an earlier snowmelt (Matthews *et al.* 2018), which impacts the phenology, growth, and reproduction processes in trees (Wipf *et al.* 2009), leading to synchronicity in their relationships with other organisms. Various pest populations also respond to increasing GDD, which has previously been used to understand other pest- and climate-driven reductions in ecological integrity; for example, increasing frequency of western balsam bark beetle outbreaks in British Columbia, coinciding with climate-driven susceptibility to such outbreaks (Howe *et al.* 2022).

During the winter season, however, we did not observe clear temperature mitigation benefits from urban forests. Although most winter climate indices examined vary between site types, very few demonstrated consistent, monotonic trends relative to canopy cover (Banfield 1983). The lack of clear winter climate trends between site types indicates that during the winter season there are likely other dominant influences on winter microclimates in urban centres, overwhelming weaker signals related to tree density. One important consideration in Newfoundland's climate is wind exposure. High wind events are generally more common during the cold season than summer, and such events are capable of moderating (or even erasing) microclimatic influences of individual sites differently, depending on their exposure and orientation relative to changing wind directions. Given the complexity of Newfoundland topography and high wind variability, it was not feasible to fully correct for this influence when selecting a handful of study sites in each region.

The composition and characteristics of the urban forest influence its ability to mitigate and ameliorate temperatures and weather events when compared to non-forested areas (Rowntree 1986; Akbari *et al.* 2001; Joshi and Joshi 2015; Ziter *et al.* 2019; Pataki *et al.* 2021). Structurally, we observed that there was more canopy cover in TU sites when compared to UU sites, apart from Corner Brook where a more extensive urban forest is present throughout the downtown region of the city. The difference in level of canopy cover between site types likely greatly influenced the trends observed in the summer months, when shading and humidity regulation play key roles in temperature amelioration (Roy *et al.* 2012). Ground cover in TU sites was primarily composed of more natural surfaces, like CWD and other vegetation, while UU sites generally had an increased presence of non-native grasses with compacted soils, pavement, and plastic, further contributing to the urban heat island effect that urban centres experience – particularly in untreed areas during the summer months (Oke 1982; IPCC 2001).

While intact boreal forest stands in Newfoundland are primarily composed of coniferous species (Damman 1983), TU and UU sites throughout the island were primarily composed of deciduous species. The high presence of these foliage-shedding trees in the overstory greatly influences the ability of urban forests to create an insulating effect under the canopy that contributes to the mitigation of winter temperatures (Akbari and Taha 1992; Akbari *et al.* 2001; Akbari and Konopacki 2004; Joshi and Joshi 2015). Similarly, as previously mentioned, wind is a major factor in the climate across the whole island of Newfoundland; even less windy parts of the island receive substantially more high wind days than mainland Canada, reducing the ability of the urban forest to create an insulating effect under the canopy. Topography and aspect associated with sites may have more of an impact to the amount of sun exposure and snowpack between sites in the winter months after leaf drop (e.g., Lugo-Pérez and Sabat-Guérnica 2011; Zuo *et al.* 2018), which would further influence the microclimates at each site. The level of urbanization also influences microclimate, with higher populations and increased levels of infrastructure contributing to a higher level of urban heat island effect (Oke 1982; IPCC 2001), and in some cases – as was observed here – reducing the number of freezing days and increasing the number of freeze-thaw cycles.

2.5 Conclusions

Urban forests in Newfoundland were often found to have different ecological characteristics than the homogeneous coniferous boreal forests that are naturally occurring on the island. Ecological characteristics additionally changed throughout regions of the urban ecosystem, driven by the level of canopy cover, location, microclimates, and level of development. These findings further amplify the need to create unique sets of differing indicators of ecological integrity throughout the urban landscape to observe and understand changes to the health, productivity, and adaptability of urban forests under the sociocultural context in which they are located. To ensure their benefits and services are realized into the future, there needs to be more intentional management and planning of urban forests in Newfoundland and throughout North America, including in planting locations and species selections to promote diversity and adaptability.

We have also shown that urban forests in Newfoundland are effective at mitigating the temperature related impacts of climate change in the summer season, particularly in their ability to reduce the exposure of residents to increasing numbers of very hot days. Other factors of the urban landscape and background climate (e.g., wind exposure) may influence their mitigation abilities in the winter season, highlighting the need to conduct more extensive, fine-scale research into how urban forests contribute to winter climates, and how changing climate conditions could impact urban tree health. Such future work could prove useful in predicting potential impacts of climate change on urban forests throughout mainland Canada. Predictions indicate that the effects of climate change will be most prevalent in urban environments, meaning urban forests are more vulnerable than natural forests, further reaffirming the need to make management decisions that will ensure their longevity into the future.

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Chapter 3: Summary and conclusions

3.1 Summary of findings

Urban forests are the backbone of green infrastructure (FAO 2016) and play a key role in our ability to adapt to and mitigate the effects of climate change and urbanization continues. The variety of influences on the layout and structure of urban forests (e.g., social, cultural, economic, environmental) and unique stressors present in urban landscapes mean that they rarely mirror the characteristics and processes of naturally occurring forests. Since the impacts of climate change are predicted to be higher in urban areas, urban forests are increasingly vulnerable, and the proper planning and management of these spaces is increasingly important. While the characteristics, benefits, and services of urban forests have been studied throughout Canada and North America, we recognized the opportunity to contribute empirical knowledge regarding the urban forests on the island of Newfoundland, which are subject to harsh climate conditions, representing a possible scenario of highly variable and extreme future climate conditions of mainland Canada.

Here, we took an island-wide approach to examining the characteristics of urban forests across the island of Newfoundland, and empirically assessing the temperature mitigation benefits they potentially provide to the urban landscape. We compared measures of ecological integrity at three distinct site types (peri-urban, PU; treed urban, TU; and untreed urban, UU) in each of four urban centres (Corner Brook, Gander, Conception Bay South (CBS), and St. John’s) along a climate gradient to assess how ecological characteristics change throughout the urban forest based on their similarity to more natural intact boreal forest stands. Our results suggest that structural and ecological characteristics of urban forests differ both between and within urban centres at different site types. Looking at structural diversity alone, our results suggest that ecological integrity may have been highest in TU sites; however, this high diversity in structure is not representative of the more homogeneous boreal forest stands that naturally occur throughout the native forests of Newfoundland (Damman 1983; Wilson 1984; Thorpe 1992).

Closer observations of perhaps more meaningful ecological indicators, like species diversity and species types present in different site types, suggest that the unmanicured peri-urban sites likely have the highest ecological integrity of the three site types, as they most closely represent natural conditions. Species richness was still often much higher in TU sites than UU sites. However, in stark contrast with the conifer-dominated natural forests of Newfoundland, urban forest canopies were primarily composed of deciduous species. This difference alone does not necessarily imply that urban (deciduous) forests lack integrity; rather, it is necessary to also consider the various socio-economic influences that influence forest composition, health, and function in urban centres (Ordóñez and Duinker 2012; Steenberg *et al.* 2019). Still, the high species evenness observed in TU and UU sites does raise major ecological integrity concerns with the urban forests of Newfoundland. As with other (urban or natural) forests, dominance of just one or two non-native species (e.g., Norway and sycamore maple in St. John's) creates potentially significant vulnerabilities to various stresses, including pests, disease, or changing environments.

Differences in ecological characteristics observed between the four urban centres in this study appeared to be driven by both level of urbanization and the underlying characteristics of their respective ecoregions. We observed high presence of deciduous and non-native species in Corner Brook and CBS PU sites, which are both located in favorable growing conditions and are heavily used by residents. St John's, as the oldest and largest urban centre in Newfoundland, had the highest understory diversity in TU sites, resulting from the high numbers of ornamental species on residential properties, but had low diversity in the overstory. Differences in the urban development planning additionally appeared to influence our results, as was observed with the high canopy cover in UU sites in Corner Brook, which has an extensive urban forest throughout the urban centre.

Temperature mitigation by urban forests was much stronger in the summer season than in the winter season. Key indicators of summer climate – growing degree days, days above 20°C, days above 25°C – were highest in UU sites which contain less canopy cover, demonstrating the ability of urban forests to both regulate the rate of increase in the growing season and mitigate high temperature days. These patterns, however, were not present during the winter months. Here, our indicators of winter climate showed varying results in the ability of trees to regulate instances and durations of freezing, frost, and freeze-thaw events, suggesting that there are likely other dominant influences on cold-season urban microclimates, including species types, level of development, and wind exposure.

3.2 Study limitations

One major limitation of this study was in the length of time our iButton temperature data loggers were deployed. Ideally, we would have data for multiple years, which would allow us to identify and account for significant climate anomalies and/or irregular weather; this would allow greater ability to observe patterns of climate mitigation by urban forests. While a long-term multi-year setup would have been beyond the scope of a Masters' project, our original intent was to deploy loggers in May 2020, which would have allowed us to collect 2 years of temperature data for both summer and winter seasons. However, logistical constraints in deploying and retrieving data loggers, imposed in response to the COVID-19 pandemic, limited our temperature data to just one winter and one summer season; issues included limits on travel social interaction (e.g., interacting with homeowners). Consequently, we are only able to draw broad-scope patterns rather than definite trends in our temperature data. Additionally, in choosing plot locations within site types, we limited the locations of our sites to single residential neighbourhoods in TU sites, and relatively small and consistent commercial areas in UU sites. This decision allowed us to pool our data within each site type, knowing there would be little variation in site conditions such as elevation, aspect, exposure, and proximity to coastlines. However, this also limited our findings to relatively small areas, precluding us from making broad scale conclusions about forest influences over of an entire urban centre.

3.3 Future study suggestions

We approached the study of urban forests on the island of Newfoundland from a coarse-scale, observational perspective. We took advantage of the climate gradient that exists across the island to assess both how characteristics of the urban forest change throughout the urban landscape, and how temperature-related climate change impacts are mitigated by urban trees. Future research of urban forests in Newfoundland should build on this framework as we have demonstrated the observational characteristics of ecological integrity and empirical assessments of broad scale temperature mitigation services under varying levels of urban forest canopy cover and in differing local climates.

Our study assessed climate mitigation services provided by urban forests across the island of Newfoundland over approximately 18 months, which limited our ability to make specific statistical conclusions about the microclimatic influence of urban forests. Our results suggested that there are likely many factors other than urban forest canopy cover that influence the microclimate in urban centres, particularly in the winter months. Therefore, we suggest implementing a finer-scale approach to determining the influence of urban forests on microclimates. Within one urban centre (e.g., St. John’s), paired long-term climate monitoring stations should be set up in various regions within proximity to permanent long-running weather stations (e.g., maintained by Environment and Climate Change Canada (ECCC)) that can act as a reference, recording air and ground temperature, humidity, wind, and precipitation. For example, in the city of St. John’s, we would recommend setting up research sites near ECCC weather stations (i.e., St. John’s Airport and St. John’s West CDA), along with the area surrounding Memorial University campus, which has its own weather station; these could be supplemented with temporary placement of experimental weather stations where necessary. Having approximately 10 sites each within areas that represent PU, TU, and UU sites in each region for 5+ years would likely reduce the influence from other site characteristics like aspect, proximity to roads/ocean, topography, and level of urbanization. A longer-term climate monitoring setup would additionally increase the ability to decipher abnormal seasonal temperature trends in the data. Using this setup, it would be easier to observe statistically significant patterns of climate mitigation and amelioration in both the summer and winter seasons provided by the urban forest.

Another possible climatology-focused study would be to assess and model the relative microclimate influence of specific tree assemblages within urban tree stands to assess how different species or species groups influence the level of climate mitigation. Using the four urban centres assessed in this study, or those with urban tree inventories (i.e., St. John’s and Corner Brook), temperature loggers can provide continuous monitoring of ambient temperatures in each of four or five stands with distinct species compositions. Stands should be selected to minimize differences in geographic characteristics (e.g., slope, aspect, exposure), and neighbouring developed and closed canopy areas should be monitored in a similar manner to allow comparisons to untreed and more natural environments, similar to the setup for this study. If this information is supplemented with data from climate monitoring stations of temperature, humidity, precipitation, and wind, statistical models relating urban microclimate mitigation services to stand composition can be developed. This will allow for empirical assessments of microclimatic effects by specific tree assemblages, facilitating planting recommendations for achieving climate-related ecosystem services (e.g., wind breaks, cooling, humidity regulation).

As previously discussed, Newfoundland has experienced five distinct climate ‘epochs’ since the late 1800s, with a notably cold climate that dominated the region from the early 1970s until the late 1990s, followed by periods of warmer climates, and period of less frequent high winds between the 1980s and mid 2000s, followed by a gradual increase (Banfield and Jacobs 1998; Finnis *et al.* 2015). Urban trees in Newfoundland have survived despite these shifting climates, and therefore both native and non-native trees established on the island may be more resilient to variations in climate than other mainland genotypes. A potential future study could assess this prediction by testing the performance of multiple tree species and provenances from urban centres in Newfoundland and a reference population from Atlantic Canada across a temperature, humidity, and wind gradient. Germinated seeds of these different provenances would then be planted at either experimental test sites throughout Atlantic Canada, which allow an assessment of viability under a range of climate conditions, or they could be planted in experimental greenhouses where they can be subject to a manufactured range of climate conditions. Measures of tree growth, productivity, and health over multiple years could then be quantified as a function of their microclimatic environment, creating a set of guidelines for matching local tree stocks to current and future climate scenarios.

Non-native species are often prevalent in urban areas (e.g., ~85% of St. John’s urban forests; City of St. John’s 2006), and anthropogenic disturbances, including urban development, can act as corridors for non-natives to become established and invasive in natural areas (e.g., Sullivan 2020). Anecdotal evidence and observations additionally suggest that many non-native species (e.g., sycamore maple) are highly fecund and have high natural recruitment in urban centres in Newfoundland. An important future study would be to assess the fecundity and recruitment of non-native urban species on the island by comparing annual reproduction rates of both native and non-native species within the same genus over an extended period (e.g., red maple and sycamore maple). The impact to adjacent boreal forest stands could also be assessed with surveys for naturally recruited (i.e., not planted) non-native urban trees in urban-adjacent natural areas. Results from this study would inform recommendations for planting non-native species in urban areas based on the level of impact they have on nearby natural areas.

3.4 Recommendations of the management of urban forests on the island of Newfoundland

Urban forests are a vital way in which cities adapt to and mitigate the effect of ongoing climate change. However, we found that the majority of urban centres in Newfoundland have insufficient information on the current status of their urban forests, which would provide important foundational information for future urban forest management and planning. During the summer of 2021, the City of Corner Brook partnered with the Conservation Corps to undertake a full inventory of their urban forest through one of their Green Teams. The City of St. John’s most recent urban tree inventory, which took place in 2005, was conducted by students in the Forest Resources Technician program at the College of the North Atlantic. Students enrolled in this program are required to complete a three-week internship (which can be paid or unpaid) prior to graduating from the program in the forest industry or forestry related employment setting. While conducting a full urban forest inventory would likely take longer than three weeks, incorporating this internship into the hiring of a summer student team could significantly reduce the costs required by the municipality. There are additionally provincial and federal grant programs, including the Student Summer Employment Program (Department of Immigration, Population Growth and Skills, Government of Newfoundland Labrador 2022) and Canada Summer Jobs (Employment and Social Development Canada 2021) that provide wage subsidies to employers from the public sector and other organizations that municipalities may be able to take advantage of to offset costs. Hiring a team of summer students or recent graduates of the Forest Resources Technician program would likely be more expensive than using a Conservation Corps Green Team, however both options could give municipalities throughout Newfoundland a relatively inexpensive method to conduct an up-to-date urban tree inventory to gain a better understanding of the status of their urban forests.

Of the four urban centres we assessed in this study, the City of St. John's was the only municipality that had an urban forest management plan, which, at the time of this study, has been implemented for 16 years of its 20-year management plan (City of St. John's 2006). Once municipalities know the current status of their urban forests and individual trees, short-and long-term urban forest management plans should be created or updated. Assessments of urban forest inventories will allow municipal staff to identify vulnerabilities in their urban forests from a variety of factors, including low species diversity or high species evenness, low foliage health, and areas with high instances of insects or disease. This information can then be used to inform short-term operational plans to address and attempt to mitigate these vulnerabilities, along with long-term plans to expand and maintain their urban forests. It is also important to note that urban forest expansion can include both large parks and green spaces (like PU sites) or individual trees throughout residential and commercial areas (like TU sites). While PU areas may provide increased benefits over TU sites because of their high canopy cover, we found here that while they may be at a lesser extent, TU sites still provide many benefits to urban areas. Expanding urban forests into TU sites is also likely more feasible in many municipalities that are seeing increased population sizes and residential development, and those that may not have the space to allocate to PU sites. Management plans should additionally include more active planning and planting actions. Presently, a lot of species and location planning in residential neighbourhoods is dictated by property owners. Municipalities could play a more active role in this process by providing more direction around species selections for certain regions of the urban centre, or discouraging species that are overrepresented or that are particularly vulnerable to insect, disease, or future climate predictions.

3.4.1 The role of non-native species in urban forests

In the face of ongoing climate change, it is increasingly important to recognize the positive aspects of incorporating select non-native species into urban forests. Many argue that urban forest health is contingent on the naturalization of the ecosystem (Zipperer *et al.* 1997), and cities have recently been prioritizing native species in an effort to combat the negative implications of non-native species (Heckmann *et al.* 2008; Nock *et al.* 2013). There is increasing evidence, however, that many ecological functions and ecosystem services are maintained or enhanced with species and structures not reflective of natural conditions (Kendle and Rose 2000; Ordóñez and Duinker 2012), arguing that they play a critical role in the ability of urban forests to persist into the future (Conway *et al.* 2019). Therefore, non-native species play a significant role in maintaining and improving the ecological integrity of urban forests by increasing species richness and maintaining wildlife connections (Schlaepfer *et al.* 2011; Ordóñez and Duinker 2012; Chalker-Scott 2015; Sjöman *et al.* 2016), and improving the resilience of the urban ecosystem (Raupp *et al.* 2010; Riley *et al.* 2018; Schlaepfer *et al.* 2020). While invasive species have negative impacts on the urban forest and surrounding natural areas, not all non-natives are invasive (Sjöman *et al.* 2016), and their impact can be reduced by controlling their abundance (Sádlo *et al.* 2017). When invasive species are properly controlled, non-natives improve the resilience of the urban forest and enhance the connections to other ecosystems (Alvey 2006; Muller and Bornstien 2010; Schlaepfer *et al.* 2011; Ordóñez and Duinker 2012).

Assisted migration ameliorates actual or anticipated loss of habitat or diversity by intentionally translocating species outside their natural range (Zhu *et al.* 2012; Ordóñez and Duinker 2012). There are widespread positive and negative impacts from assisted migration making them a highly debated, controversial topic (e.g., McLachlan *et al.* 2007; Sax *et al.* 2009; Minteer and Collins 2010; Woodall *et al.* 2010; Hewitt *et al.* 2011; Aubin *et al.* 2011; Pedlar *et al.* 2011; Almas and Conway 2016; Conway *et al.* 2019). This range-expansion strategy has been proposed as a means to maintain healthy urban forests into the future under a changing climate (Kowarik 2011; Ordóñez and Duinker 2013), and has been explicitly implemented in some urban forest management plans (e.g., Peel Region 2011; City of Halifax 2013). Nursery stocks for urban species are rarely from local regions and provenances, meaning assisted migration has already been occurring in urban spaces in a passive way (Hitchmough 2011; Conway *et al.* 2019). To ensure minimal loss and maximum benefits, a shift to active assisted migration by choosing species from climates that are representative of current and future conditions needs to occur (e.g., planting species and provenances from temperate climates in urban centres in Newfoundland; Ng 2018; Conway *et al.* 2019). This strategy will prevent species and provenance selection based on availability and growth rates, and encourage selection based on adaption to climate and site conditions, and provision of benefits and services.

Management of the urban forest needs to shift away from prioritizing species based on their native status and toward promoting species diversity, functional diversity, and species richness (Morgenroth *et al.* 2016; Conway *et al.* 2019; Paquette *et al.* 2021). This approach would both better align management objectives with current practices, and improve the resilience of the urban forest while producing benefits and services (Conway *et al.* 2019). Promoting a combination of native and non-native species positively impacts the diversity, stability, and sustainability of the urban forest (Sjöman *et al.* 2016). Instead of the exclusion of non-natives, urban forest management should implement regional assessments of species suitability and risk of invasiveness (Sjöman *et al.* 2016). Under this management regime, non-native species pose low risk and greatly benefit the urban forest; native-only urban forest policies have no scientific basis (Johnston *et al.* 2011; Chalker-Scott 2015). These are the species most likely to persist in future climate scenarios (Schlaepfer *et al.* 2011) as a result of their increased tolerance to environmental and climate conditions.

Focusing instead on the function or benefit desired from the urban forest, and current and future climate predictions, is more productive strategy for urban forest management (Chalker-Scott 2015; Amini Parsa *et al.* 2020). Stressors that are present in the urban ecosystem can also be considered in species selection decisions based on the desired functions and benefits (Ordóñez and Duinker 2012). These shifts in management strategies optimize ecosystem services while improving the stability, resilience, and ecological integrity of urban forests into the future.

3.5 Conclusion

Urban forests are a vital part of global strategies to mitigate and adapt to the effects of ongoing climate change, especially in urban areas that house the majority of the global population, and are expected to experience the harshest effects of climate change. While we were able to demonstrate the climate mitigating services of Newfoundland’s urban forests in the summer months, warming is expected to be more extreme in the winter season, and winter temperatures are a major limiting factor for tree establishment (Ramage *et al.* 2013; Jenerette *et al.* 2016; Kendal *et al.* 2018). Our results indicate that there are likely other environmental factors that influence winter microclimates, limiting our ability to decipher mitigation services provided by urban forests during the winter months. Here, the island of Newfoundland can still act as a worst-case scenario model for empirically assessing the climate mitigation services that urban forests provide during the winter seasons through finer-scale analyses of winter temperatures. We have also shown that structural and ecological characteristics change throughout the urban forest, driven by a variety of factors, which in turn impacts how closely different parts of the urban forest resemble naturally occurring forests in the region. Given the stark difference between urban ecosystems and naturally occurring ecosystems, this is not unexpected, and when these factors are integrated into management practices, urban forests can still have high ecological integrity even though they may not resemble natural forests (Ordóñez and Duinker 2012). A perhaps more important takeaway from this study is that this integration of socioeconomic influences is often lacking in ecological assessments and planning; municipalities in Newfoundland lack both the general understanding of the composition and characteristics of their urban forests, along with the differentiated indicators of integrity that would allow them to identify vulnerabilities in their urban forests. We hope that this project encourages municipalities in Newfoundland, and throughout North America, to be more intentional in the management and planning of their urban forests, to ensure the longevity of urban forests and their benefits into the future.

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Appendix I: Plant identification references

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Appendix II: Outcome of generalized linear models

AII.1 A generalized linear model comparing the effects of i) urban centre and ii) site type along with combined effects of iii) urban centre and site type on urban forest canopy cover (%) in 120 peri-urban, treed urban, and untreed urban sites (2021).

```
Call:
glm(formula = c.cover ~ site.type * city, data = site.data)

Deviance Residuals:
        Min        1Q        Median     3Q         Max
        -61.256    -3.224    -0.208     8.216      60.436


Coefficients:
                    Estimate    Std. Error    t value     Pr
      (intercept)     81.072       5.949       13.672    <2e-16      ***
      tu               0.208       8.414        0.025     0.98033
      uu             -51.908       8.644       -6.005     3.03E-08   ***
      cbs              6.708       8.924        0.752     0.45399
      gan              6.136       8.414        0.729     0.46751
      sj               5.512       8.414        0.655     5.1387
      tu:cbs         -19.292      12.265       -1.579     0.11886
      uu:cbs         -32.592      12.581       -2.591     0.011      *
      tu:gan         -12.792      11.899       -1.075     0.2849
      uu:gan         -34.932      12.063       -1.896     0.00463    **
      tu:sj          -14.525      12.063       -1.204     0.23135
      uu:sj          -31.656      12.424       -2.548     0.1234     *

      Significance codes: 0 ‘***’; 0.001 ‘**’; 0.01 ‘*’; 0.05 ‘.’; 0.1 ‘’; 1

(Dispersion parameter for gussian family taken to be 353.9444)

Null deviance: 167789 on 112 degrees of freedom
Residual deviance: 35748 on 101 degrees of freedom
AIC: 997.21

Number of Fisher Scoring interactions: 2
```

AII.2 A generalized linear model comparing the effects of i) urban centre and ii) site type along with combined effects of iii) urban centre and site type on urban forest diameter at breast height (cm) in 80 peri-urban and treed urban sites (2021).

```
Call:
glm(formula = dbh ~ site.type * city, data = site.data)

Deviance Residuals:
        Min         1Q          Median      3Q          Max
        -29.770     -5.412      0.530       5.490       36.289


Coefficients:
                     Estimate      Std. Error     t value       Pr
       (intercept)       26.88          4.125       6.517     9.88E-09   ***
       tu                15.89          5.833       2.724     0.00816    **
       cbs              -3.768          6.187      -0.609     0.54458
       gan               -8.17          5.833      -1.401     0.16583
       sj                -8.26          5.833      -1.416     0.16128
       tu:cbs          -11.792          8.504      -1.387     0.16998
       tu:gan           -11.88           8.25       -1.44     0.15438
       tu:sj            -3.799          8.364      -0.454      0.6511

       Significance codes: 0 ‘***’; 0.001 ‘**’; 0.01 ‘*’; 0.05 ‘.’; 0.1 ‘’; 1

(Dispersion parameter for gussian family taken to be 170.1453)

Null deviance: 16014 on 76 degrees of freedom
Residual deviance: 11740 on 691 degrees of freedom
AIC: 623.59

Number of Fisher Scoring interactions: 2
```

AII.3 A generalized linear model comparing the effects of i) urban centre and ii) site type along with combined effects of iii) urban centre and site type on urban forest height (m) in 80 peri-urban and treed urban sites (2021).

```
Call:
glm(formula = height ~ site.type * city, data = site.data)

Deviance Residuals:
        Min         1Q          Median      3Q          Max
        -8.2500     -3.5269     -0.0113     1.9944      18.7500


Coefficients:
                      Estimate     Std. Error    t value       Pr
      (intercept)       13.363        1.718        7.777     7.97E-11   ***
      tu                -3.103        2.305       -1.346        0.183
      cbs                0.275        2.43         0.113        0.91
      gan                1.837        2.515        0.731        0.468
      sj                 2.087        2.305        0.906        0.369
      tu:cbs            -2.455        3.26        -0.753        0.454
      tu:gan            -1.188        3.324       -0.357        0.722
      tu:sj             -3.57         3.209       -1.112        0.27

      Significance codes: 0 ‘***’; 0.001 ‘**’; 0.01 ‘*’; 0.05 ‘.’; 0.1 ‘’; 1

(Dispersion parameter for gussian family taken to be 170.1453)

Null deviance: 2024.4 on 71 degrees of freedom
Residual deviance: 1511.5 on 64 degrees of freedom
(41 observations deleted due to missingness)
AIC: 441.51

Number of Fisher Scoring interactions: 2
```

AII.4 A generalized linear model comparing the effects of i) urban centre and ii) site type along with combined effects of iii) urban centre and site type on urban forest overstory species richness in 120 peri-urban, treed urban, and untreed urban sites (2021).

```
Call:
glm(formula = over.rich ~ site.type * city, family = poisson, data = site.data)

Deviance Residuals:
          Min        1Q         Median     3Q        Max
          -1.5492    -0.8165    -0.3880    0.4811    2.6248


Coefficients:
                       Estimate     Std. Error    t value     Pr
        (intercept)           4         0.2626     15.231    <2e-16      ***
        tu                 -3.6         0.3714     -9.693    4.26E-16    ***
        uu              -3.6667         0.3816     -9.609    6.52E-16    ***
        cbs              -0.625         0.3939     -1.587     0.11574
        gan                -1.9         0.3714     -5.116    1.49E-06    ***
        sj                 -1.4         0.3714     -3.769    0.000276    ***
        tu:cbs            1.425         0.5414      2.632     0.00982    **
        uu:cbs           0.4028         0.5554      0.725    0.469994
        tu:gan              2.6         0.5252       4.95    2.98E-06    ***
        uu:gan           1.7667         0.5325      3.318    0.001262    **
        tu:sj            1.5556         0.5325      10921   0.0004301    **
        uu:sj            1.1917         0.5484      1.173    0.032133    *

        Significance codes: 0 ‘***’; 0.001 ‘**’; 0.01 ‘*’; 0.05 ‘.’; 0.1 ‘’; 1

(Dispersion parameter for poisson family taken to be 1)

Null deviance: 209.918 on 112 degrees of freedom
Residual deviance: 73.533 on 101 degrees of freedom
AIC: 270.36

Number of Fisher Scoring interactions: 6
```

AII.5 A generalized linear model comparing the effects of i) urban centre and ii) site type along with combined effects of iii) urban centre and site type on urban forest understory species richness in 120 peri-urban, treed urban, and untreed urban sites (2021).

```
Call:
glm(formula = under.rich ~ site.type * city, family = poisson, data = site.data)

Deviance Residuals:
        Min        1Q         Median     3Q         Max
        -2.3238    -0.7211    -0.1201    0.5831     1.7269


Coefficients:
                    Estimate     Std. Error    t value        Pr
      (intercept)        2.7         0.3829      7.052    2.24E-10   ***
      tu                -0.8         0.5414     -1.478      0.1426
      uu             -1.2556         0.5563     -2.257      0.0262   *
      cbs               -0.7         0.5743     -1.219      0.2257
      gan                0.4         0.5414      0.923       0.358
      sj                   1         0.5414      1.847      0.0677   .
      tu:cbs             0.4         0.7893      0.507      0.6134
      uu:cbs          0.4778         0.8096       0.59      0.5564
      tu:gan            -0.3         0.7657     -0.392       0.696
      uu:gan          -1.444         0.7763     -1.861      0.0657   .
      tu:sj          -0.7889         0.7763     -1.016      0.3119
      uu:sj          -1.3194         0.7995      -1.65       0.102


      Significance codes: 0 ‘***’; 0.001 ‘**’; 0.01 ‘*’; 0.05 ‘.’; 0.1 ‘’; 1

(Dispersion parameter for poisson family taken to be 1)

Null deviance: 121.509 on 112 degrees of freedom
Residual deviance: 76.205 on 101 degrees of freedom
AIC: 362.28

Number of Fisher Scoring interactions: 5
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Appendix III: Ground cover types recorded during 2021 urban forest assessments

Table AIV.1: List of ground cover types observed during urban forest assessments along with the description of ground observations included in each cover type.

Ground Cover Type	**Description**
Tree	Any tree stems located within the plot, including the centre tree
Coarse woody debris	Sticks, large and small branches, fallen dead trees
Moss	Mosses
Ground	Areas of bare ground or soil (i.e., no coverage of other ground cover types)
Rock	Large boulders, small, medium, and large size rocks
Pole	Utility pole or other poles (e.g., flag post) located in the plot
Grass	Native and non-native grasses
Pavement	Roadways, sidewalks, and other concrete surfaces
Mulch	Landscaping mulch located in gardens and surrounding planted tees
Plastic	Any plastic coverings, litter, or other landscaping objects located in the plot

Appendix IV: Species observed during 2021 urban forest assessments

Table AIV.1: List of overstory species observed in each urban centre, along with their native status, and species count and proportion of canopy in each site type.

Urban Centre	Site Type	Species	Latin Name	Native Status	Type	Species Count	Proportion
Corner Brook	Peri-urban	Balsam fir	*Abies balsamea*	Native	Coniferous	85	40.3
		Red maple	*Acer rubrum*	Native	Deciduous	40	19.0
		White spruce	*Picea glauca*	Native	Coniferous	30	14.2
		White birch	*Betula papyrifera*	Native	Deciduous	18	8.5
		Mountain maple	*Acer spicatum*	Native	Deciduous	15	7.1
		Choke cherry	*Prunus virginiana*	Native	Deciduous	7	3.3
		Yellow birch	*Betula alleghaniensis*	Native	Deciduous	7	3.3
		Speckled alder	*Alnus incana*	Native	Deciduous	5	2.4
		Black spruce	*Picea mariana*	Native	Coniferous	3	1.4
		Chuckley pear	*Alemanchier canadensis*	Native	Deciduous	1	0.5
	Treed urban	Norway maple	*Acer platanoides*	Non-native	Deciduous	15	50.0
		White birch	*Betula papyrifera*	Native	Deciduous	9	30.0
		Apple	*Malus spp*	Non-native	Deciduous	6	20.0
	Untreed urban	Basswood	*Tilia americana*	Non-native	Deciduous	1	33.3
		Mountain ash	*Sorbus americana*	Native	Deciduous	1	33.3
		Slippery elm	*Ulmus rubra*	Non-native	Deciduous	1	33.3
Gander	Peri-urban	Black spruce	*Picea mariana*	Native	Coniferous	251	71.9
		Balsam fir	*Abies balsamea*	Native	Coniferous	65	18.6
		Eastern larch	*Larix laricina*	Native	Coniferous	19	5.4
		Trembling aspen	*Populus tremuloides*	Native	Deciduous	12	3.4
		Green ash	*Fraxinus pennsylvanica*	Native	Deciduous	1	0.3
		White birch	*Betula papyrifera*	Native	Deciduous	1	0.3
	Treed urban	Trembling aspen	*Populus tremuloides*	Native	Deciduous	24	44.4
		White birch	*Betula papyrifera*	Native	Deciduous	17	31.5
		Norway maple	*Acer platanoides*	Non-native	Deciduous	4	7.4

		Apple	*Malus spp*	Non-native	Deciduous	3	5.6
		Mountain ash	*Sorbus americana*	Native	Deciduous	3	5.6
		Balsam fir	*Abies balsamea*	Native	Coniferous	1	1.9
		Red pine	*Pinus resinosa*	Native	Coniferous	1	1.9
		Scots pine	*Pinus sylvestris*	Non-native	Coniferous	1	1.9
	Untreed urban	Norway maple	*Acer platanoides*	Non-native	Deciduous	2	100.0
Conception Bay South	Peri-urban	Balsam fir	*Abies balsamea*	Native	Coniferous	126	63.3
		Black spruce	*Picea mariana*	Native	Coniferous	40	20.1
		Chuckley pear	*Alemanchier canadensis*	Native	Deciduous	13	6.5
		Eastern larch	*Larix laricina*	Native	Coniferous	8	4.0
		White birch	*Betula papyrifera*	Native	Deciduous	6	3.0
		Choke cherry	*Prunus virginiana*	Native	Deciduous	2	1.0
		White spruce	*Picea glauca*	Native	Coniferous	2	1.0
		Mountain ash	*Sorbus americana*	Native	Deciduous	1	0.5
		Yellow birch	*Betula alleghaniensis*	Native	Deciduous	1	0.5
	Treed urban	Norway maple	*Acer platanoides*	Non-native	Deciduous	15	35.7
		Mountain ash	*Sorbus americana*	Native	Deciduous	8	19.0
		Golden chain	*Laburnum spp*	Non-native	Deciduous	7	16.7
		White birch	*Betula papyrifera*	Native	Deciduous	4	9.5
		Horse chestnut	*Aesculus hippocastanum*	Non-native	Deciduous	2	4.8
		Sycamore maple	*Acer pseudoplatanoides*	Non-native	Deciduous	2	4.8
		Apple	*Malus spp*	Non-native	Deciduous	1	2.4
		Blue spruce	*Picea pungens*	Non-native	Coniferous	1	2.4
		Lilac	*Syringa spp*	Non-native	Deciduous	1	2.4
		Yellow birch	*Betula alleghaniensis*	Native	Deciduous	1	2.4
	Untreed urban	Norway maple	*Acer platanoides*	Non-native	Deciduous	2	100.0
St. John's	Peri-urban	Balsam fir	*Abies balsamea*	Native	Coniferous	212	66.5
		Black spruce	*Picea mariana*	Native	Coniferous	95	29.8
		Mountain ash	*Sorbus americana*	Native	Deciduous	6	1.9
		White birch	*Betula papyrifera*	Native	Deciduous	4	1.3

		Chuckley pear	*Alemanchier canadensis*	Native	Deciduous	2	0.6
	Treed urban	Mountain ash	*Sorbus americana*	Native	Deciduous	4	36.4
		Norway maple	*Acer platanoides*	Non-native	Deciduous	3	27.3
		Trembling aspen	*Populus tremuloides*	Native	Deciduous	3	27.3
		Copper beech	*Fagus sylvatica f. purpurea*	Non-native	Deciduous	1	9.1
	Untreed urban	Black pine	*Pinus nigra*	Non-native	Coniferous	1	100.0