Measuring urban edge effects and its impact on restoration potential in Rouge National Urban Park

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Urban development is a driving force behind habitat fragmentation and biodiversity loss in major metropolitan areas. While greenspaces and naturalized areas can provide resources for wildlife, urban areas are organized in such a way that the transition from forest to suburban neighbourhood is abrupt and heavily maintained. This arrangement in conjunction with the intensity of urban activities leaves a limited area to buffer any anthropogenic impacts, negatively affecting species that are unable to adapt. To examine the extent to which urban activities are affecting naturalized areas, a one-sided edge effect study was conducted in Rouge National Urban Park (RNUP) in Toronto, Ontario, Canada. The purpose of this study was to frame what kinds of restoration plans might be possible given the amount of least impacted area, i.e., interior conditions. Data were collected in the largest accessible forest fragment, with one primary edge being sampled. 13 transects of 500 m length were used, with samples taken at the following distances *d* from the edge: 0 m, 50 m, 125 m, 250 m, and 500 m. Reference conditions were categorized as those found at d = 500 m. The Shannon Diversity Index and Pielou Evenness Index were used to compare plant species composition and analyzed using a randomized test of edge influence without blocking. The distance of edge influence was not observable, with no distance found to be significantly different from reference conditions. The results may be due to data noise from other nearby edges, primarily a large informal trail network whose presence was not known prior to data collection. Had these additional sources of fragmentation been observed, it would have resulted in smaller sampling fragments with inherently less potential interior habitat. It may also be the result of non-typical urban edge conditions at d = 0 m as it ran parallel to metal fencing and lay beneath a mature canopy. The edge had a sheltered side-canopy in contrast to an expected open forest edge that is exposed to disturbances such as increased light exposure and heavy anthropogenic activity. Additional observations may indicate limited interior conditions in the studied area of RNUP. Though not examined specifically, evidence of anthropogenic impacts was not contained to the defined fragment edges and permeated every area of the park. Given how impacted RNUP appears to be, improvements to ecological integrity seem unlikely unless accompanied by a broader landscape approach. Restoration activities may help reduce further biodiversity loss and bolster other ecosystem services provided by the park. Due to the complexity of potential influencing factors, this research is the beginnings of a foundational framework that sought to better understand priorities and best practices for ecological restoration in major urban areas. Subsequent research is expected to develop a deeper understanding of the drivers behind observed edge conditions.

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1.1.1 Issues with Urbanization

Urbanization is a driving force of habitat loss and fragmentation, leading to a worsening decline in biodiversity around the globe (Dunford & Freemark, 2004; Fletcher et al., 2007; Ikin et al., 2014; LaPaix & Freedman, 2010; Marull et al., 2018). In the past few decades, urban populations globally have increased by 700 million with the trend expected to accelerate (Durham Region, n.d.; Ikin et al., 2014; Ontario, 2021). As the distances between cities and natural environments continues to decrease, it is no longer just a question of managing nature, but people as well (Burke & Mitchell, 2007; Hall et al., 2021; Standish, Hobbs, & Miller, 2013). Where resources and space are limited, consideration ought to be given to a wide breadth of stakeholders such as wildlife, plants, the watershed in its entirety, different levels of government, communities, and commercial businesses. From a biophysical perspective, this additional layer of complexity amplifies the issues being faced by the natural environments that remain in and around major urban centres. From various kinds of pollutions and road mortalities to invasive species and climate change, anthropogenic pressures continue to grow (Bugiel et al., 2018; Leermakers, 2020; Toronto and Regions Conservation Authority, 2015).

1.1.2 Urban Sprawl in Ontario, Canada

The problems between urbanization and the natural environment can be exemplified by several issues that the province of Ontario, Canada, has faced in recent decades. Like many developed regions, the province is expected to experience major population growth in the coming years. It is estimated to reach 20 million by 2046, which would be a 35.8% increase from 2020 levels (Caldwell et al., 2022). To support the growing number of people it is often necessary to create new housing developments, which can contribute to urban sprawl (Caldwell et al., 2022). With a finite amount of land in the province, land development can become the priority over other land-use types such as agriculture or wildlife habitat. From 1996 to 2016, 1.5 million acres of farmland were lost with increased urbanization being a main contributor (Caldwell et al., 2022). Once developed, the productive capabilities of the land are lost forever (Caldwell et al., 2022). The issue was recognized decades ago, with several policies being implemented by the Ontario provincial government to reduce the impact of urban development such as the *A Place to Grow: Growth Plan for the Greater Golden Horseshoe* and the *Greenbelt Plan.*

Examining the latter, the Greenbelt (2005) is an area of the province that was given protections from commercial and residential development to preserve the integrity of the environment and the agricultural lands within. This designation has not stopped the Province of Ontario from being inundated with proposals for adjustment, with rumours of weakening protections becoming a key political issue. At the end of 2022, the Province of Ontario passed Bill 23 titled the *More Houses Built Faster Act* (2022). Amongst other measures, it removed environmental protections for 2995 ha of land in the Greenbelt to allow for housing development to occur. The move was argued to be necessary to combat an inflated housing market, though critics maintain the environmental damage would have been irreparable (Jones, 2022). In September of 2023 the Province reversed course, returning protections to the selected lands amid speculation about corrupt motives driving the initial decision (CBC News, 2023).

This conflict demonstrates the value of land and how an equitable division of resources can be challenging, especially when located in a highly sought after region. Even amongst two necessary land-use types for modern society, i.e., housing and agriculture, the prioritization of one over the other can fluctuate. In addition to the loss of farmland, increased urbanization has decreased the capacity of lands to support climate change adaptation and mitigation measures (Caldwell et al., 2022). This begs the question, where does ecological restoration lay in relation to these other higher priorities and has its effectiveness been hindered in some way? Environmental issues are notorious for being an afterthought in regard to policymaking, and while progress is being made change is slow and not guaranteed.

1.1.3 Challenges with Private and Commercial Sectors

Another factor that is important to consider is the role of the commercial sector, whose interests are often at odds with those of the environment. While 87% of Ontario is publicly held Crown Land, 95% of it is located in northern Ontario, leaving southern Ontario mostly privately owned (Destination Ontario, 2023). The consequence is that a homogenous landscape management plan is difficult to implement because land managers are dealing with a large number of independent stakeholders each with their own economic interests. It can lead to a patchwork of land-use types that can hinder both agricultural and wildlife activity which may require a greater degree of connectivity than is available (Caldwell et al., 2022; Ikin et al., 2014; Kremer & Merenlender, 2018). There is often a reliance on public education and opt-in programs to encourage private landowners to incorporate more ecologically focused management practices, such as the planting of native trees or installation of pollinator gardens (Credit Valley Conservation, 2023; LEAF, 2022; Meadoway, 2019;). On the other side, it means that private organizations may ignore environmental considerations in favour of their own economic gains.

In 2020, the pursuit of economic recovery from the COIVD-19 pandemic led the Province of Ontario to use a minister's zoning order (MZO) to overrule local planning requirements, allowing them to expedite commercial development (McGillivray, 2020). One was used to enable the development of an Amazon warehouse atop an ecologically significant wetland, though the project was later cancelled by developers (Crawley, 2021; McGillivray, 2020). If it were not for the pushback received from residents, the project would still be in progress. Some politicians, such as the Mayor of Pickering, framed the cancelled project as an economic loss to the community with 2000 jobs and tens of millions of dollars in tax revenue being forfeited (Crawley, 2021).

This event highlights the debate between growing the economy and protecting the environment, and how the solution is not always so clear. Even after the toll imposed on communities by a global pandemic, in this instance sacrificing more wetland area was not a popular development despite encouragement by project proponents. What little naturalized habitat remains is not necessarily safe from development pressures, whether the land itself is threatened or the area just adjacent to it. There remains uncertainty as to how long a specific environment will be allowed to persist into the future if another use for the land is deemed more desirable. Metrolinx (2023) is one example, where improved public transit provides great public utility, but the habitat that must be removed for its construction is being compensated with reforestation plantings at another location. In a heavily developed urban centre, it appears difficult to not only create effective landscape connectivity but to maintain it far into the future. This is an important consideration for attempting to restore the ecological integrity of a given landscape because it does not exist in a vacuum.

1.1.4 Conflicts between People and Nature

On a local scale, the mere cohabitation of adjacent lands can be a source of tension in a community. With a province as diverse as Ontario, there will invariably be a range of opinions on how best to manage wildlife and the environment. Some people may enjoy the outdoors and seeing flocks of birds flying all around, while some may consider local beavers to be a menace. People's relationship with nature is not always mutualistic and can at times result in antagonism. In the city of Toronto, stories of coyote encounters are increasing with the potential risk to public safety forcing discussions around how best to manage the problem, with euthanasia being a regular consequence (Jackson & Bingley, 2022; Lavoie, 2022; Mertz, 2022). In 2020 and 2021, foxes had made a den in a popular recreational area, Woodbine Beach, and though efforts were made to protect them from the public, in both years kits had been found dead (Aguilar, 2021; Fox, 2020).

As much as some would like to live in harmony with nature, the complexity of these urban landscapes means that there will inevitably be a difference of opinion about how best to manage the land and interact with nature. If some residents believe the reforestation of a nearby area will bring unwanted wildlife activity, their opinions need to be considered along with those in favour of such a project. Creating a consensus within a community can be difficult and is likely to result in compromises that may reduce the efficacy of a restoration project from an ecological perspective (Romanelli et al., 2023). One may imagine what Rouge National Urban Park, a large protected area in the Greater Toronto Area (GTA), may be if it was one contiguous naturalized area rather than a majority agricultural land (Parks Canada, 2020). It is important to maintain meaningful community engagement and work within the limits set by them.

1.1.5 Complexity of a Dynamic Urban System

The challenge becomes how should one approach ecological restoration in the face of such complexity. Some studies that looked at the relationship between people and natural areas focused on smaller local communities with easily categorized identities, such as Indigenous groups (Adams & Hutton, 2007; Sarkar & Montoya, 2010). Sarkar & Montoya (2010) examined the "social ecology" model for land and resource management through the issue of protected areas and the impact on different Peruvian Indigenous tribes, the Ashaninka, Awajum, Ese'eja, Huambisa, and Matsiguenka. Through legal loopholes, resource extraction was permitted by the Peruvian government despite previously agreed on protections, giving rise to some violent altercations. While the causal forces driving the conflict are by no means simple, it was framed as these small local communities with a long history managing the land versus large government and corporate interests. The application of something like the "social ecology" model maintains its utility, but more work may be required to adapt it to an urban context.

Urban environments, especially those found in Western developed nations, have their own unique structures and challenges that warrant more targeted research. The city of Toronto is one of the most diverse cities in the world across a multitude of categories such as ethnicity, economic class, and politics. Communities overlap across multiple scales whether it be Toronto, one of its boroughs, or a particular neighbourhood. Some people live in high rise condominiums and apartments in the downtown core while others live in sprawling suburban communities. There may be staunch environmentalists who would do anything to save a wetland while at the same time be struggling families who are looking for steady work just to afford the cost of living. The number of factors that need to be considered are greater because there is a large collection of diverse intertwined communities that are evolving at a near constant rate.

The notion of sustainable development as outlined in the 1987 Brundtland Commission is one response that is gaining traction, whereby the needs of the present are met without compromise to the needs of future generations. It is a guiding principle that seeks to achieve more equitable outcomes when it comes to managing the earth and its resources. Part of this concept is the recognition that the natural environment provides an array of values that ought to be maintained so that it is accessible to all. In other words, the environment should be managed in a way that achieves a multitude of benefits for the greatest number of stakeholders (Kremer & Merenlender, 2018). The application of this principle has yet to yield the desired results with effective stakeholder engagement remaining a significant challenge. Even amongst Indigenous groups, there can be variability in opinions on issues of ecological management, further demonstrating the challenge in dealing with socioecological systems (Adams & Hutton, 2007; Sarkar & Montoya, 2010). Until ideals are translated into appropriate actions, those environments nearest urban centres of development will continue to struggle as the choice between economic growth and ecological conservation is presented dichotomously.

1.1.6 Key Issues Identified

When looking at restoration opportunities in an urban environment, it appears to be a tale of restrictions. Whether it is the political jurisdictions, available land, persistent negative anthropogenic inputs, or needing to appeal to a diverse set of interests, the sandbox in which one has to work is small. These issues are by no means exclusive to urban environments nor are they insurmountable, but they are amplified in such a way that is not comparable to a remote town in northern Ontario. Where a restoration site might have one municipality with which to work, in southern Ontario that could be three. Time and resources are a significant limitation to any project, but especially in an urban environment where budgets can balloon out of control and delays will only be tolerated for so long. For ecological restoration to be both effective and efficient in urban areas, it will require a solid comprehension of the local urban environment so a project can more intentionally be integrated into the landscape.

For the purposes of my research, I narrow my focus to the local level and on indicators of urban impact in naturalized areas. Within a dynamic urban landscape, I sought to understand the limits imposed on ecological restoration at the outset due to broad scale urban influences. The goal was to reframe how one might design a restoration project to better accommodate local urban conditions. I did not attempt to provide an explanation of the complex drivers and systems acting on a given area as such a project would be more appropriate for a longer-term study. Instead, I narrowed my research to measuring how much land is being significantly impacted by adjacent urban environments through changes in vegetative composition. To gain insight on what might be possible given enough land and with stronger regulations, I chose to study a federally protected area, Rouge National Urban Park (RNUP), Canada's first designated national urban park. If a protected area as large as RNUP cannot mitigate anthropogenic influences and provide a safe interior for wildlife, then smaller naturalized areas would suffer from similar issues and new restoration strategies might be reviewed (Adams et al., 2023; Beninde, Veith, & Hochkirch, 2015; Romanelli et al., 2023; Rudd, Vala, & Schaefer, 2002; Santangeli et al., 2023).

1.2 Protected Areas as a Conservation Strategy

1.2.1 Protected Areas and Historic Issues

A common strategy for conservation is the implementation of protected areas (PA). These are high quality natural areas given government protections to preserve what is viewed as pristine environment (Adams & Hutton, 2007). The origins of the PA model can be traced to the development of Yellowstone National Park in 1872 (Adams & Hutton, 2007; Robbins, 2020). Founded on what some call the fortress conservation model, PAs were guided by the idea that nature in its purest form exists outside of human influence (Adam & Hutton, 2007; Robbins, 2020). Humans are inherently destructive and must be separated from nature for it to thrive. This notion of an 'unspoiled Eden' or 'a lost Eden in need of protection' gained legitimacy in the 1990's when it became integrated into modern science-based approaches to PA development (Adam & Hutton, 2007).

In the context of ecological conservation, in some cases PAs have been shown to be effective. The Galapagos Islands are one area that has had PA status for over 50 years, with 97% of the land being held under the jurisdiction of the Galapagos National Park (Mathis & Rose, 2015). The remaining 3% was set aside for urban and rural development for permanent residents. As the place that inspired Charles Darwin's theory of evolution, the preservation of the Islands was a primary goal and resulted in a concerted effort that successfully retained many of its unique marine and terrestrial biodiversity (Mathis & Rose, 2015). When viewed from a sociological perspective, residents were involuntarily displaced from much of their traditional lands and left to turn to tourism for economic sustenance (Mathis & Rose, 2015). This need for work resulted in the demand for greater infrastructure development, which posed a risk to the ecological stability of the Islands. Traditional PAs that employ a fortress conservation model may be effective when viewed from a narrow ecological perspective, but with them comes a host of other issues that cannot be ignored.

PAs often had consequences for local Indigenous peoples who were forcibly removed from their lands for the sake of conservation (Adam & Hutton, 2007; Clapp, 2004; Sarkar & Montoya, 2010). Their land rights were involuntarily forfeited for a perceived greater good, i.e., the preservation of the natural environment. Not only is such a policy unjust, but it relies on the faulty premise that the environment is most pristine when it is unaffected by humans (Adams & Hutton, 2007; Clapp, 2004; Sarkar & Montoya, 2010). Increasingly areas previously thought to be purely natural are being found to be the result of direct human impact. The prairies, which have long been valued for their biodiversity were in many cases created by Indigenous intervention as fires were used for the establishment of hunting grounds (Adams & Hutton, 2007). The Amazon rainforest, which was thought of as the peak virgin environment in desperate need of protection, has been found to have a long history of human occupancy and co-evolution (Roosevelt, 2013). It is not the presence of human influence in the natural environment that is problematic but the nature of that interaction that is worth examination.

Another consequence of the Garden of Eden mentality is that it assumes one can both create and maintain the ideal conditions for a given ecosystem within a set jurisdiction. The political boundaries of a given PA are just as intangible as those used to divide nation states and do nothing to prevent the movement of things like wildlife or pollutants across borders. It is the reason agreements such as the *Migratory Birds Convention Act* (1994) or the *Montreal Protocol* required international cooperation, because environmental issues do not abide by state border conventions. In the context of PAs, a ruling governing body such as Parks Canada may have control over how to manage the land for which it is responsible, but it can only do so much. If light pollution is impacting nocturnal wildlife (Longcore & Rich, 2004; Newport, Shorthouse, & Manning, 2014), while alternative means of lighting within park lands may be used, Parks Canada cannot fully prevent the impact of pollutants from adjacent communities (Parks Canada, 2020).

1.2.2 Protected Areas in a Modern Context

It has been over a century since PAs were first developed, and while they are by no means perfect, progress has been made in addressing their historic issues. RNUP is one of Canada's newest national parks and represents a more modern approach to PAs. It is part of a trend that is moving away from the idea that conservation requires the exclusion of human elements and instead promotes a strategy that is both more inclusive and adaptive. This idea can be seen in the *Canada National Parks Act 1988* and subsequent *Rouge National Urban Park Act 2015,* which establishes the restoration and maintenance of ecological integrity as the priority when it comes to park management. Ecological integrity is defined as "a condition that is determined to be characteristic of its natural region and likely to persist, including abiotic components and the composition and abundance of native species and biological communities, rates of change, and supporting processes" (Hobbs et al., 2010). In other words, the management endpoint is not to achieve a set of conditions akin to pre-European settlement, but one that is indicative of both current and future environmental factors. It is aligned with the notion of novel/hybrid ecosystems, whereby conditions have changed so dramatically from historical norms that they have achieved a new stable state (Hobbs et al., 2010; Hobbs et al., 2014; Prach et al. 2019). Given the circumstances, a restoration of former states of being would either be impossible or too resource intensive to achieve. This new direction allows for greater flexibility in ecosystem management with presumed higher rates of success, as it shifts from focusing on historic states that are no longer viable to desired effects and future conditions (Hobbs et al., 2010; Hobbs et al., 2014).

Moreover, it acknowledges the role that humans have played in the development of the environment. Nature has co-evolved with local peoples and in some cases can thrive with human intervention (Adams & Hutton, 2007; Roosevelt, 2013). In the context of RNUP, this means Parks Canada recognizes the long history of urbanization that is now characteristic of the area, as well as the influence Indigenous peoples had had prior. The surrounding urban and agricultural lands are unlikely to be removed in favour of naturalized areas. Any restoration efforts will have to improve the ecological integrity of RNUP within the urban context in which it is embedded. These management ideals are explicitly outlined in Parks Canada's 2019 draft management plan and are prioritized as a part of the first two key strategies (Parks Canada, 2020). It is attempting to balance both social and ecological needs, giving wildlife a space while allowing people and farmers to continue using the land alongside them.

1.2.3 Protected Areas in an Urban Setting

RNUP is too early in its development to know whether such a model can succeed where others have not. On paper, it appears to be addressing many historic issues associated with PAs and intends to set management goals that are adaptable to a dynamic urban environment. The issue is that there is a strict designated area where people and the environment can be managed by a given set of guiding principles, while outside of those borders there is no obligation to follow suit. Parks Canada (2020) can regulate the trail network, improve watershed health, and increase the amount of forested area within RNUP itself. It can even work with the farmers to develop and implement more sustainable agricultural practices.

What it does not have control over is the expanding influence of the GTA and the need to develop the requisite infrastructure to support it. Affordable housing, jobs, and the economy are consistently a top priority during every election and every news cycle, and it is further evidenced by the amount of development occurring in the surrounding area. For every cancelled warehouse project, there are three more that occupy former agricultural land or remnant forest area. Housing developments continue to expand northward in formerly rural areas, increasing traffic and stressing roads that were not initially designed for such high usage. While conscious efforts are being made to minimize the spread of invasive species, the fact is the sheer number of people in an urban area with their immense capacity for travel makes it an almost impossible task (Padayachee et al., 2017). Even the most conscious of hikers can inadvertently spread seeds from one site to another.

RNUP can be turned into the ideal naturalized environment that maintains its utility for both residents and farmers alike, but it must still contend with both the positive and negative influences of the surrounding urban environment. It may not be the traditional Garden of Eden approach where all human elements are removed for the sake of ecological integrity, but it is still a defined piece of land where the regulations change one step outside of park borders. The question is how much improvement can be seen in the ecological conditions of RNUP when less than a metre away is a drastically different landscape with a multitude of impacts. Urban areas are also changing at such a rapid pace, where in the blink of an eye everything can shift as demonstrated by the recent COVID-19 pandemic. The management plan Parks Canada (2020) proposed states that it will aim to achieve a set of conditions that is indicative of the landscape and likely to persist into the future. The relevant question for the purposes of my research is given the mitigating effects of urban environments as they are today, what level of ecological restoration is possible. To scale the problem into something manageable, I focused my attention to a more fundamental aspect that could be built upon with subsequent research. I set out to study how a naturalized area as large as RNUP is being influenced by its adjacent urban areas through an examination of the border where these two landscapes meet, i.e., the edge.

1.3 Urban Edge Effects

1.3.1 Urban Dynamics Through an Edge Effect Lens

I chose to view the issue of ecological restoration in an urban context through the lens of an edge effect study. When large sections of native forests are replaced by other ecosystem types, it becomes fragmented with abrupt shifts from one habitat type to the next (Murcia, 1995; Zheng & Chen, 2000). This area of transition creates an edge, resulting in changes in both the biotic and abiotic conditions of the area (Ewers & Didham, 2006; Murcia, 1995). It becomes distinct from the two adjacent environments as variables such as species interactions, trophic structure of communities, movement of individuals, and resource flows are altered (Ewers & Didham, 2006). It is common for invasive exotic species to proliferate in these edges as well as interior specialist species to be reduced. In general, the habitat can be modified to such a degree that conditions move beyond their natural range of variation, impacting the survival of native organisms (Murcia, 1995).

Whether an edge habitat harms or enhances ecological structure and function is dependent on the cumulative impacts, as different species may find certain environmental conditions favourable (Dunford & Freemark, 2004; Fonesca & Joner 2007; Ikin et al., 2014; Watling & Orrock, 2010). Birds better adapted to urban environments have been found to thrive in edge conditions while those ill-adapted are typically those requiring more interior habitat (Ikin et al., 2014). The reaction of a given species will be related to the type of edge response that has been created through the distribution of resources in both habitats along an edge (Ikin et al., 2014). A positive edge response is created when resources in the adjacent areas are complementary to one another, and it is neutral when they are supplementary. The response is transitional when one side of the edge has lower quality resources leading to positive conditions in one habitat and negative conditions in the other. This model of edge response is only applicable at the species level, with more research needed for application at the community level (Ikin et al., 2014).

Dunford & Freemark (2004) examined the effects different land use types had on forest birds near Ottawa, Canada. They found that species richness and abundance was affected by nearby urban and agricultural lands, but not in any way that was predictable. Impacts were dependent on available habitat structures and resources in those external habitats which led them to argue that migratory birds are more sensitive to urbanization at greater distances than previously thought. They suggest that urban areas should not be thought of as inhospitable environments but another piece of the broader landscape that can either support or hinder a species' capacity for survival. Their recommendation of a

200-1800 m buffer between forest patches and urban activity highlights the importance of selecting an appropriate scale in edge effect research, as what is biologically meaningful will vary depending on the observed species.

When evaluating an urban park like RNUP, its landscape is laden with high contrast edges causing heavy fragmentation. It incorporates most of the naturalized area in the region, but outside its borders it immediately transitions to either urban or agricultural land. This shift in land use is further emphasized by physical features like rail lines, high traffic roads, and fenced backyards. These structures not only mark the outer perimeters but run through RNUP in a typical grid-like pattern (Figure 1). Roads create their own set of edge conditions (Eigenbrod, Hecnar, & Fahrig, 2009), with the added consequence of road mortalities for wildlife. The latter is a known impact that is actively being studied in RNUP (Leermakers, 2020; Toronto and Region Conservation Authority, 2015; Wicks, 2019). It is of particular concern to the threatened Blanding's turtle, which is a key species of interest to Parks Canada (2020).

At finer scales, there is a historic trail network that further intersects throughout the area (Figure 1). The effects of trails in PAs are well documented, with studies having shown they can reduce vegetation cover and height, increase soil compaction and erosion, introduce weed species, and disturb wildlife (Ballantyne et al., 2014; Ballantyne & Pickering, 2015). Ballantyne et al. (2014) found a positive correlation between the number of access points and the degree of fragmentation, arguing that trail networks can create an artery of disturbance within a PA with varying degrees of edge effects. The impacts can accumulate over spatial scales, which are further amplified by the addition of informal trails (Ballantyne & Pickering, 2015). Such concerns are echoed by both Parks Canada (2020) and the Toronto and Region Conservation Authority (TRCA) (2015) regarding RNUP. The latter reported that the prevalence of dogs on hiking trails may be contributing to lower breeding success amongst songbirds and is the reason why places like Tommy Thompson Park in Toronto, Canada prohibit access for pets (Tommy Thompson Park, n.d.). It is further evidenced by the TRCA's (2015) recommendation to implement sanctuary areas within RNUP. Parks Canada (2020) acknowledges the need to have a well-regulated trail network, with some sections already having been closed to allow for restoration to occur. There is a clear understanding of the impact that public access has on a PAs, so the question is how RNUP has been impacted and can more intentional alterations be made.

1.3.2 Review of Edge Effect Research

Edge effect research has been conducted for decades, following a rudimentary sampling design with variations in statistical analysis and modeling. At its core, these studies select a response variable which is assumed to experience observable changes as samples are taken from the edge, at distance $d = 0$ m, and along a perpendicular transect at varying distances into the interior (Murcia, 1995). It is flexible in that the response variable is based on the topic of interest, with plant communities being a common subject (Hamberg et al., 2009; Labadessa, 2017; Lehvävirta et al., 2014; Vallet et al., 2010; Watling & Orrock, 2010). Studies have looked at the effects of edges on bird and insect communities, which have been shown to demonstrate various adaptations to these kinds of environments (Dunford & Freemark, 2004; Fonesca & Joner 2007; Watling & Orrock, 2010). This ability to choose one's subject can create variations in the measured distance of edge influence (DEI) as species will have differing sensitives to a given set of conditions (Fonesca & Joner 2007; Laurance and Yensen, 1991; Ries & Sisk, 2004; Zheng & Chen, 2000).

Another characteristic of edge effect studies is the decision to use either a one-sided or two-sided approach. In the past, the former was more commonly used as it sought to measure ecological processes and patterns from the edge and into the interior of one habitat type (Fonesca & Joner, 2007). A two-sided approach examines the entire gradient as it moves from one habitat type, across the edge, and into the adjacent habitat. In a broad literature review of 317 papers, Fonesca & Joner (2007) found 72% of papers used a one-sided approach with 28% using a two-sided approach.

The advantage of a one-sided approach lies in its simplicity, as it seeks to answer how deep into a habitat fragment do edge effects penetrate (Fonesca & Joner, 2007). If DEI was measured up to 100 m from the edge of a 1 km radius, then 64% of the remaining fragment would be labelled as the interior (Fonesca & Joner, 2007). These kinds of studies are more than capable of determining DEI, but the drawback is that they do not result in generalizable principles about the drivers and dynamics of edge conditions (Fonesca & Joner, 2007). This issue is where a two-sided approach becomes beneficial, as it is argued to better account for the interactions between two habitat types and the edge along which they meet. It can describe the flow of energy, materials, and organisms across edges, allowing for the creation of predictive modeling (Fonesca & Joner, 2007; Ries & Sisk, 2004). When nuanced factors such as ecological flows, access to spatially separated resources, resource mapping, and species interactions are incorporated into a single model, one may be able to predict a given species' response to an observed edge (Ries & Sisk, 2004). This kind of information would be favourable in a restoration context, as land managers could more reliably alter environmental conditions to meet their management goals.

Edge effect studies have experienced two common issues, i.e., the failure to select appropriate replicates and the failure to account for confounding variables (Fletcher et al., 2007; Murcia, 1995). Appropriate replicates are necessary for creating generalizable results, limited though they may be depending on the chosen approach. The issue is related to the second problem of accounting for confounding variables. Not all edges are created equal and observed impacts can be influenced by factors related to fragment size, matrix type, orientation, physiognomy, and management history (Murcia, 1995). It would be prudent to describe a study area in detail to acknowledge when certain factors could not be isolated and may be impacting results.

1.3.3 Statistical Modeling of Edge Effects

While edge effect modeling does not require any specific method, there are benefits depending on the one chosen. Generalized linear models are commonly used (Eigenbrod, Hecnar, & Fahrig, 2009; Harper & Macdonald, 2001; Lehvävirta et al., 2014) and offer a simplified way of determining at what distance do interior conditions differ significantly from those at the edge. Laurence & Yansen (1991) developed their "Core-Area Model". This method examines the distribution of *d*, which is defined as the distance at which the edge effect is no longer observable. Another parameter for its use is that at the core of the fragment, there must be some unaltered habitat. It considered the shape of a fragment, which was assumed to have a linear relationship between the shape index, i.e., how much it deviates from a perfect circle, and the loss of core area, or the interior. The core-area model is outdated with more recent iterations having been developed (Ewers & Didham, 2012). It was also problematic because it relied on low fragment shape complexity and assumed that fragment shape, size, and DEI were independent from one another (Ewers & Didham, 2012).

Randomized test of edge influence (RTEI) is a model designed by Harper & Macdonald (2011) in response to the inconsistent results of DEI when implementing different statistical methods of analysis. A review of nine methods of analysis including RTEI with blocking, RTEI without blocking, Critical Values Approach, ANOVA with *post hoc* Tukey comparisons, paired t-tests, Wilcoxon Rank Sum tests, piecewise regression, exponential 2/3 rule, and exponential with CI intersection, found that given the same data set DEI was variable depending on the selected method. They argued that RTEI was more consistent across diverse sampling approaches with only 7-15 transects and a minimum of 3 or more distances from the edge being measured. They do not argue that RTEI is necessarily more accurate, simply that model selection will impact DEI results.

Currently, there is no consensus on what the best methodological approach is to studying edge effects despite the lengthy history of the research (Fonesca & Joner, 2007; Harper & Macdonald, 2011; Murcia, 1995). With no clear best approach, my decision on which model to use was based on the comparisons made by Harper & Macdonald (2011) and the limitations of the data that was collected for this study. Paired t-tests, RTEI, and the exponential 2/3 rule were considered. RTEI without blocking was the selected model based on the sampling design and number of transects able to be sampled.

1.4 Rouge National Urban Park

1.4.1 Case for Studying Rouge National Urban Park

Rouge National Urban Park exemplifies the highlighted issues with ecological restoration in urban areas and is the focus of this study. In 2011, Parks Canada began the process of designating what was previously known as the Rouge Valley as a federally protected area (Parks Canada, 2020). With the enactment of the *Rouge National Urban Park Act* in 2015, Rouge National Urban Park was created. Sitting on the eastern border of the Greater Toronto Area, Canada's largest metropolitan area, RNUP is a new kind of PA that will employ an inclusive management style. It is often promoted as being Canada's first national urban park (Parks Canada, 2020).

The policy framework that the *Rouge National Park Act 2015* produced is key, as it dictates management guidelines specific to RNUP. In 2017, an amendment was added to use the same language as the *Canada National Parks Act 1988*, establishing as the priority in Section 6, the restoration and maintenance of ecological integrity (Parks Canada, 2020). Another significant aspect of the *Rouge National Urban Park Act* is that it legislates the continued practice of agriculture within the park, meaning the existing farmlands in the PA are federally protected. Section 4 of the *Rouge National Urban Park Act* states that the park was established "for the purposes of protecting and presenting, for current and future generations, the natural and cultural heritage of the park and its diverse landscapes, promoting a vibrant farming community and encouraging Canadians to discover and connect with their national protected heritage areas" (Parks Canada, 2020).

Through legislation, it established a set of ecological and social management priorities that Parks Canada will aim to balance. In addition to improving the ecological health of RNUP, the included farmland now possesses similar protections. Parks Canada wants people to enjoy the park and gain a better appreciation of the area as opposed to excluding them from it to limit ecological harm. It is observing both current and potential future conditions to set RNUP on a trajectory that can be sustained for decades. This strategy is an explicit use of novel ecosystem principles, as Parks Canada is acknowledging RNUP cannot reasonably be returned to the vast forest it once was. While it is a large, naturalized area, it is also permanently a feature of a major metropolitan area that is only expected to expand. Parks Canada will attempt to create a thriving ecosystem in an urban setting that can also be enjoyed by the residents who will make the most use of it. What that looks like is the main point of inquiry as I begin building the knowledge necessary to make that determination.

Under the jurisdiction of Parks Canada, RNUP is surrounded by urban, suburban, rural, and agricultural lands (Figure 1). While the land is managed by the Canadian federal government, it must work in tandem with the surrounding stakeholders to manage varying interests, from municipal and provincial governments to private and commercial landowners. Each level of authority will be guided by different rules and regulations, have their own ends and land uses, and belong to different political ecosystems. The GTA and its surrounding regions continue to experience rapid development pressures as local populations are projected to grow, increasing prices on an already inflated housing market (Nuttall, 2023; Ontario, 2021).

The Province of Ontario had set a target for building 1.5 million new homes by 2031 but is already at risk of missing its goal (D'Mello & Callan, 2023). The pressure for more housing is why the Province had initially proposed to expand urban boundaries and develop into the Greenbelt, as it would have opened up almost 4700 ha of land for housing by 2051 (Callan & D'Mello, 2023). At the time, the move had raised enough concern over potential impact on the ecological integrity of RNUP that an environmental study had been commissioned by the federal government to assess the issue (Javed, Allen, & Ballingall, 2023). In this kind of setting, a broad landscape approach to ecological restoration is not always possible and may result in a patchwork of different land use types (Beninde, Veith, & Hochkirch, 2015; Kremer & Merenlender, 2018; Rudd, Vala, & Schaefer, 2002; Standish, Hobbs, & Miller, 2013).

While RNUP is in the early stages of development, should it prove successful it could act as a model for how governments can balance the needs of both natural and urban environments (LaPaix & Freedman, 2010). RNUP's urban setting means there are an abundance of high contrast edges due to the patchwork of land use types – the transitions from forest to farmland and residential neighbourhood are immediate and typically marked with built infrastructure. This creates challenges for ecological restoration – can restoration address impacts of fragmentation given the bordering, permanent development? This larger question will require multiple years of study, but it must be initiated by someone. This was my intent – examine the fragmentation in RNUP as an exploratory research project that is needed to set the stage for, ultimately, determining priorities and management actions for ecological restoration in RNUP.

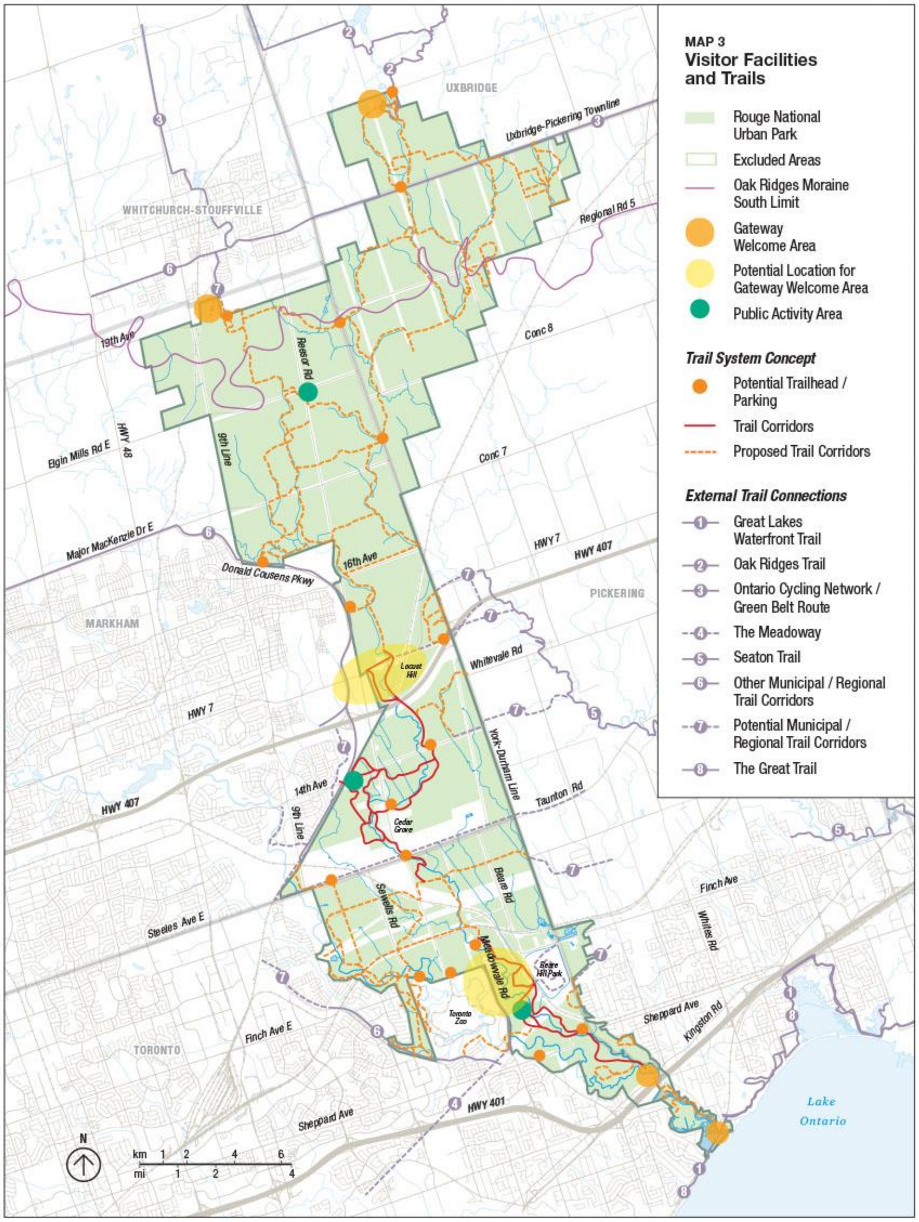

Figure 1 – [Map of Rouge National Urban Park and Proposed Visitor Developments]
This map outlines Parks Canada's (2020) proposed management plans for visitor infrastructure development. The creation of Welcome Centres and the improvement of trail connectivity are a primary focus. Improving watershed connectivity from the Oak Ridges Moraine to Lake Ontario is part of its key strategy.

1.4.2 Review of Biological Inventory and Assessment

The current ecological state of RNUP is best described by a 2015 biological inventory and assessment performed by the Toronto and Region Conservation Authority (2015). It compiles 20 years of data with some information having been provided by the Ontario Ministry of Natural Resources (OMNR) (Toronto and Region Conservation Authority, 2015). Prior to European settlement in 1883, the land was described as a contiguous forest and wetland with windfall areas and sites for natural burns (Toronto Region Conservation Authority, 2015). Up until 1851, there was 73% forest cover in the Scarborough Township, but this decreased to 6% in 1921 (Toronto Region Conservation Authority, 2015). Recent restoration efforts have returned forest cover to 21% based on 2013 aerial photos (Toronto Region Conservation Authority, 2015).

Habitat patch quality was deemed “fair”, though the southern region which is better connected was described as “good” (Toronto and Region Conservation Authority, 2015). RNUP has 226 ecological land classifications (ELC), 26 of which were found to be of regional concern. Over a third of recorded vascular plants were non-native at 37%, leaving 63% native plant species. In the decade preceding the report, 155 breeding vertebrate species were observed in the following numbers: 112 birds, 24 mammals, and 19 herpetofauna. Native biodiversity continues to decline with 39 flora species considered to be extirpated or in sharp decline. The endangered butternut tree (*Juglans cinerea*) was observed in the area. Some bird species have adapted to the surrounding urban environment well, while more sensitive species have experienced decline. In 2005, the TCRA surveyed 112 bird species, which is a 10% reduction from previous reports by the OMNR. Across every metric, there is a clear trend of declining ecological health despite a 15% increase in forest cover over the last century. The decline in biodiversity is to be expected and reinforces the need for new management approaches (Santangeli et al., 2023). RNUP is so entrenched in its urban setting that the amount of historic forest cover lost is unlikely to be recovered and wildlife sensitive to urban environments are unlikely to return (Romanelli et al., 2023).

Anthropogenic impacts within RNUP were also reported (Toronto and Region Conservation Authority, 2015). Thousands of visitors every year combined with a previously unregulated trail network has resulted in general issues with vegetation trampling, soil compaction, and dispersal of non-native species. The usage of trails by humans and dogs was identified as a potential cause of nest abandonment amongst birds, thereby reducing the chances of reproduction and survival. Some residents who border the park have expanded their backyards into the forest, with one reported incident of a stunt bike playground having been created. Harvesting of plants was found to be a common occurrence, though it is now strictly prohibited. Carnivore populations have decreased, enabling herbivores to thrive to the point where over browsing has led to stunted plants and a thinning forest floor (Russell et al., 2017). Urban and agricultural runoff are of particular concern to the ecological state and function of the watershed, especially as it relates to the Rouge River estuary. Atmospheric nitrate deposition from air pollution has increased soil fertility in a manner that favours rapidly growing invasive exotics and weedy native species. Some of these issues can be regulated to a degree, but diffuse ones like managing atmospheric nitrate deposition are beyond the power of Parks Canada. Even local problems like the harvesting of plants will be impossible to prevent completely. Unlike traditional national parks with regulated entrances, RNUP is accessible from virtually anywhere, by anyone. These are complex challenges that affect everything in and around the park.

The TRCA (2015) made several recommendations for management of the area. These suggestions included better management of public use, improvements to habitat quality and connectivity, and continued monitoring and assessment of realized changes. The creation of sanctuary areas, i.e., restricting public access in high-quality areas with more sensitive flora and fauna, was strongly recommended. Some areas typified as prairie remnants or dry oak forests would see some benefit from the reintroduction of a prescribed burn regime. One statement of note was that despite the larger size of some of the forested areas, there were not as many breeding bird species as would be expected given what the area ought to be able to support. The TRCA acknowledged that even if all of its recommendations were enacted upon, without a landscape approach that goes beyond the jurisdictional borders of RNUP there would be little improvement to the biodiversity loss being experienced.

The report by the TRCA (2015) was the main piece of inspiration for this research. The primary factor was the seeming contradiction in recommending the establishment of sanctuary areas while also admitting only a wider landscape approach would yield significant results in terms of reversing biodiversity loss. In conjunction with the statement that RNUP should be able to support more bird species than are currently being observed, it made me question how one should approach a project like RNUP if the source of many of its problems are outside of Parks Canada's control. Part of wanting to better contextualize urban restoration through an edge effect study is to help determine the course of action with the highest chance of success given such limitations.

1.4.3 Existing Research on Rouge National Urban Park

Although I aim to build foundational knowledge on the urban/nature dynamic of RNUP, I am not the first to perform a study in the park. There is a small amount of existing research that was reviewed to see if a previous edge effect study had been conducted, or if there was information that could be used to better contextualize the issue. In conjunction with the detailed report by the TRCA (2015), I was better able to frame the current condition of RNUP and the challenges it faces. No previous edge effect study on RNUP was found, but some studies provided insights on the level of fragmentation.

Research specific to RNUP has (mainly) been the domain of Master's theses, mainly from the nearby University of Toronto; four of these theses have been published in peer-reviewed journals (Barakat, 2017; Bugiel et al., 2018; Deslauriers, 2021; Gill, 2017; Guppy, 2019; Khazaei, 2015; Leermakers, 2020; Livingstone, Cadotte, & Isaac, 2018; Ramsay et al., 2017; Shamirian, 2020; Sodhi et al., 2019; Villiger, 2015; Wicks, 2019). While no article was found to have directly studied edge effects in the park, several insights can be pulled from the existing research.

Two theses from the U of T Department of Forestry, one by Deslauriers (2021) and another by Shahmirian (2020), focused on habitat suitability for flying squirrels (*Glaucomys spp.*). The latter examined the methodology for defining suitability while Deslauriers (2021) looked at whether landscape fragmentation was affecting flying squirrel abundance. She concluded that habitat connectivity was correlated to species abundance, and that RNUP was connected enough to support existing subpopulations. This study contradicts the assumption I am making that RNUP is heavily fragmented, but only as it relates to flying squirrels. The impacts of urban activity are diverse and will not affect all species equally, e.g., Leermakers (2020) studied methods of tracking turtle movements to design more effective eco passages and prevent road mortalities. These studies demonstrate the varied consequences of anthropogenic fragmentation in the area. My research is not focused on the impacts on any one species but will seek to study general indicators of urban influence. This choice will affect the broader applicability of my study.

Bugiel et al. (2018) and Sodhi et al. (2019) examined the impacts of dog strangling vine (*Vincetoxicum rossicum*) on soil and habitat composition respectively. Bugiel et al. (2018) found the diversity of soil bacterial communities to be lower in areas heavily invade by *V. rossicum* with the observed changes being independent of other anthropogenic disturbances. Sodhi et al. (2019) found *V. rossicum* to be deterministically altering the functional structure of herbaceous plant communities, creating a filter that excluded different species depending on the habitat in which they were found. As the presence of *V. rossicum* in the park is a direct consequence of modern urban activity and disturbances (Bugiel et al., 2018), these papers provide important context on how RNUP is being influenced. For the purposes of my research, I noted what ecological conditions facilitated *V. rossicum* and if there were areas, such as the interior of the park, where it was struggling to take hold. It may prove to be a useful indicator of urban influence.

Other papers examined the social dimensions of RNUP. Livingstone, Cadotte, & Isaac (2018) used RNUP as a case study, comparing how park users' valuation of different ecosystem services differed from non-users. Ramsay et al. (2017) looked at barriers millennials face when trying to visit urban parks like RNUP. Gill (2017) examined how the agricultural community was engaged when creating RNUP, which is a key tenet of the *Rouge National Park Act*. While these papers are not directly related to an edge effect study, they help paint a fuller picture of the matrix of influence impacting the park. Though my data will be purely ecological, it will be framed within the broader social context of the surrounding urbanized area.

1.4.4 Restoration Efforts and Park Developments

Though it has been eight years since RNUP was established through legislation, it is still in the early stages of development as much of that time had been spent finalizing land acquisitions (Parks Canada, 2020). Restoration plans have yet to be finalized, though several projects and priorities have been outlined (Parks Canada, 2020). Improving watershed health is one of Parks Canada's primary goals, proposing to increase wetland and riparian area by 50 ha. Increasing connectivity with the Oak Ridges Moraine is another concern, with older pieces of infrastructure inhibiting healthy waterflow (Parks Canada, 2020).

Regarding reforestation, efforts are focused on the creation of buffer zones in areas adjacent to agricultural fields (Parks Canada, 2020). The impacts of agricultural activities fall outside the scope of my research, but I do take note that buffer zones are an active management strategy. Whether it would work adjacent an urban environment is what I am studying. Some attention is being paid by Parks Canada to improving greenspace connectivity with nearby areas such as Beare Hill Park, Morningside Creek, and the Godswood Resource Management Tract. Parks Canada states it will use a science-based ecological approach, concentrating on projects with a high return on investment.

Given the urban setting of the park, anthropogenic impacts are a primary feature that must be controlled. There is a great deal of attention being given to visitor infrastructure, as it is looking to expand the trail network in the northern areas of RNUP to improve access (Figure 1). Several "Welcome Areas" and parking infrastructures are currently under construction (Parks Canada, 2020). New infrastructure is being designed to have minimal negative effects, such as the use of dark-sky lightning. Parks Canada's (2020) strategy for minimizing the impacts from human behaviour focuses on education and community engagement to promote a mutually beneficial relationship between visitors, residents, and nature. Of note, there are active discussions with farmers in RNUP about developing more environmentally friendly farming practices (Parks Canada, 2020).

1.5 Studying Edge Effects in Rouge National Urban Park

1.5.1 Purpose of the Study

It is too early to evaluate the effectiveness of RNUP's new management regime, so instead I endeavour to assess the urban context in which restoration will be taking place. With pressures from intense urbanization near RNUP, I wanted to gain insight into not only what kinds of ecological restoration work is possible, but what will achieve the greatest results. Longer-term, the relevant questions include:

- Is the area large enough to be protected from the surrounding urban influences or is it a novel ecosystem that will continue to support the flora and fauna that have already adapted to the urban landscape?
- Does being adjacent to Canada's largest urban centre hinder possible ecological gains, necessitating a broad scale unified approach across the wider region?
- What is the current relationship between RNUP and its urban surroundings and how much of it can be changed?

For the shorter-term, I focused on building a foundation for understanding the urban/nature dynamic of RNUP so that ecological restoration efforts may have a baseline from which to work. This study is meant to be the first in a series, with subsequent research building on one another to achieve a more complex understanding of the urban/nature dynamic. I examined this relationship through an edge effect study to quantify the magnitude of influence the surrounding urban environment is having on RNUP. I hypothesized that due to the fragmented layout of RNUP in conjunction with presumed levels of anthropogenic activity at the edges, I would find a large DEI, leaving a minimal amount of interior habitat.

1.5.2 Research Approach

To explore the urban/nature dynamic in RNUP, I conducted an edge effect study to quantify the degree of influence the surrounding urban environment is having on the naturalized area. By determining DEI, I aimed to gain insight on the context in which restoration may occur and if it will create any limitations that must be observed. RNUP is not only surrounded by intense urbanization separating it from adjacent greenspaces, but the park itself is divided by a range of anthropogenic features such as roads, trails, and rail lines. The fragmented nature of the area with its abundance of high contrast edges lent itself well to this kind of study (Wicks, 2019). Should a large DEI be found with a limited interior forest, strategies such as the use of sanctuary areas may not be effective and allow other options to be explored.

RNUP was studied using a novel ecosystem framework in alignment with growing restoration trends and Parks Canada's (2020) own management plan. Urban woodlands have been drastically altered, with remnant woodlots being more indicative of past conditions that no longer exist (Romanelli et al., 2023). This decision impacted sampling design, as reference site data was collected from within RNUP itself as opposed to a separate location. A remote forest of equivalent size with less anthropogenic impacts may have greater ecological integrity, but it would not provide a useful comparison (Romanelli et al., 2023). Urban development is expected to intensify throughout the region, so an effective reference site was deemed one that is least impacted given the circumstances. For the purposes of this study, it was assumed to be the deepest interior of RNUP relative to the designated fragment perimeter.

1.5.3 Research Scale

Though RNUP occupies a larger area, the project was scaled down to make data collection feasible (Figure 2). The initial proposed study area was the section of RNUP north of Highway 401 and south of Highway 407. This selection was based on the significant disruption to habitat connectivity caused by a major highway (Eigenbrod, Hecnar, & Fahrig, 2009) as well as the shift in habitat type. South of Highway 401, RNUP turns into a wetland with the presence of the Rouge River estuary and would require a different set of considerations for sampling design. North of Highway 407 is predominantly farmland with small patches of forested areas spread across the region. I am less interested in the specific impacts of agricultural activity and more focused on urban activity, so this area was not considered for data collection. These parameters were used for the initial pilot study which was needed to determine the necessary sample size as well as fine tune the sampling design. For the subsequent primary study, the study area was scaled down to the areas directly adjacent to Twyn Rivers Drive to accommodate discovered issues with site access and safety (Figure 3).

1.5.4 Sampling Design

Research was carried out using a one-sided approach, as data was only collected on one side of the observed edge. This contrasts with a two-sided approach, which would have examined the entire edge gradient from the interior of one habitat type, across the edge, and into the interior of the adjoining habitat. In other words, data was only collected within RNUP itself and not into the adjacent neighbourhoods. The decision arose from the need to consider the scope and timespan of a Master's thesis. A two-sided approach directly examines the role that the surrounding landscape is playing in the park's development. However, it produces a level of complexity that is better scoped for a Ph.D thesis. For example, permissions for data collection would need to be gained from a multiplicity of landowners, with ongoing communications needed to maintain effective working relationships (Dyson et al., 2019). There is also increased variability in independently managed landscapes, e.g., I could sample the garden spaces of 20 houses and find 20 different garden types.

A one-sided approach was deemed sufficient as it is more than capable of quantifying DEI. It simply lacks the explanatory power to describe the drivers and principal dynamics of edge interactions (Fonesca, & Joner, 2007). There is a push to perform more experimental studies that seek to gain a deeper level of understanding of the causal forces of edge conditions, but this would require more time and resources than was available. A one-sided approach was selected as the results can still be meaningful and be further built upon by future studies.

Data was collected on plant species diversity and species evenness, measured through the Shannon Diversity Index and the Pielou Evenness Index, respectively. These indicators were selected based on the ease with which they could be measured, requiring no special tools or additional expertise. Research began when COVID-19 restrictions were still in place. With travel and sharing of equipment being discouraged, an accessible sampling design ensured adaptability in the face of changing regulations. They also provided a sufficient indication of how forest structure changes as one moves from the edge of the park and into the interior. A more targeted study that focused on a specific animal or pollutant was possible, but the variable nature of edge effects would not allow the results to be generalized at the community level. Examining the plant structure of RNUP would provide a general understanding of how this complex matrix of urban influences is impacting ecological integrity.

1.5.5 Method of Analysis

RTEI without blocking as designed by Harper & Macdonald (2011) was the selected method of analysis. When compared against parametric testing, which was also considered, it provides more consistent DEI measurements across different sampling designs with a minimum of seven transects and three edge distances (Harper & Macdonald, 2011). Parametric testing requires 15 transects before DEI measurements begin to stabilize. RTEI is sensitive to inherent variations in the reference interior but not at the edge. It is also adaptable to a two-sided approach. When compared against a blocking design, there is minimal impact on results meaning the decision was based on sampling design. RTEI is conducive to this kind of exploratory research as it can more easily be built upon by later studies.

2.0 Measuring Urban Edge Effects

2.1 Study Site

The study took place in Rouge National Urban Park in Toronto, Ontario, Canada. It spans approximately 5810 ha of land located in the ecotones among the Carolinian floristic region, the deciduous forest zone of south-western Ontario, and the Great lakes-St. Lawrence floristic region (Toronto Region Conservation Authority, 2015). It is the most eastern location of Ontario's 38 Carolinian sites (Toronto Region Conservation Authority, 2015). RNUP holds several environmental designations for its protection, such as two Areas of Natural and Scientific Interest (ANSI), eight Environmentally Significant Areas (ESA), and two Provincially Significant Wetlands (PSW) (Toronto Region Conservation Authority, 2015). It is home to the Rouge River estuary, which is Toronto's largest coastal marsh and a prime example of a coastal wetland and river valley system in the GTA (Toronto Region Conservation Authority, 2015).

There are three major sources of anthropogenic fragmentation that cut across the area from east to west: Highway 401, Highway 407, and the Toronto hydro corridor (Figure 1). The Glen Rouge Campground is a primary piece of Parks Canada infrastructure and is currently being renovated (Parks Canada, 2020). A bordering area of note is the newly designated Beare Hill Park, which is a greenspace created atop a decommissioned landfill (City of Toronto, n.d.). RNUP is neighbour to the Toronto Zoo, who aids in some wildlife management and research in the area. In terms of recreational activities, they are typical of most parks with many people using the area for hiking, fishing, and canoeing where allowed. Near the Twyn Rivers parking lot is a large hill which locals use for tobogganing in the winter months. The Rouge Beach, which sits at the southernmost point of RNUP connected to Lake Ontario, is another popular destination for visitors.

a)

b)

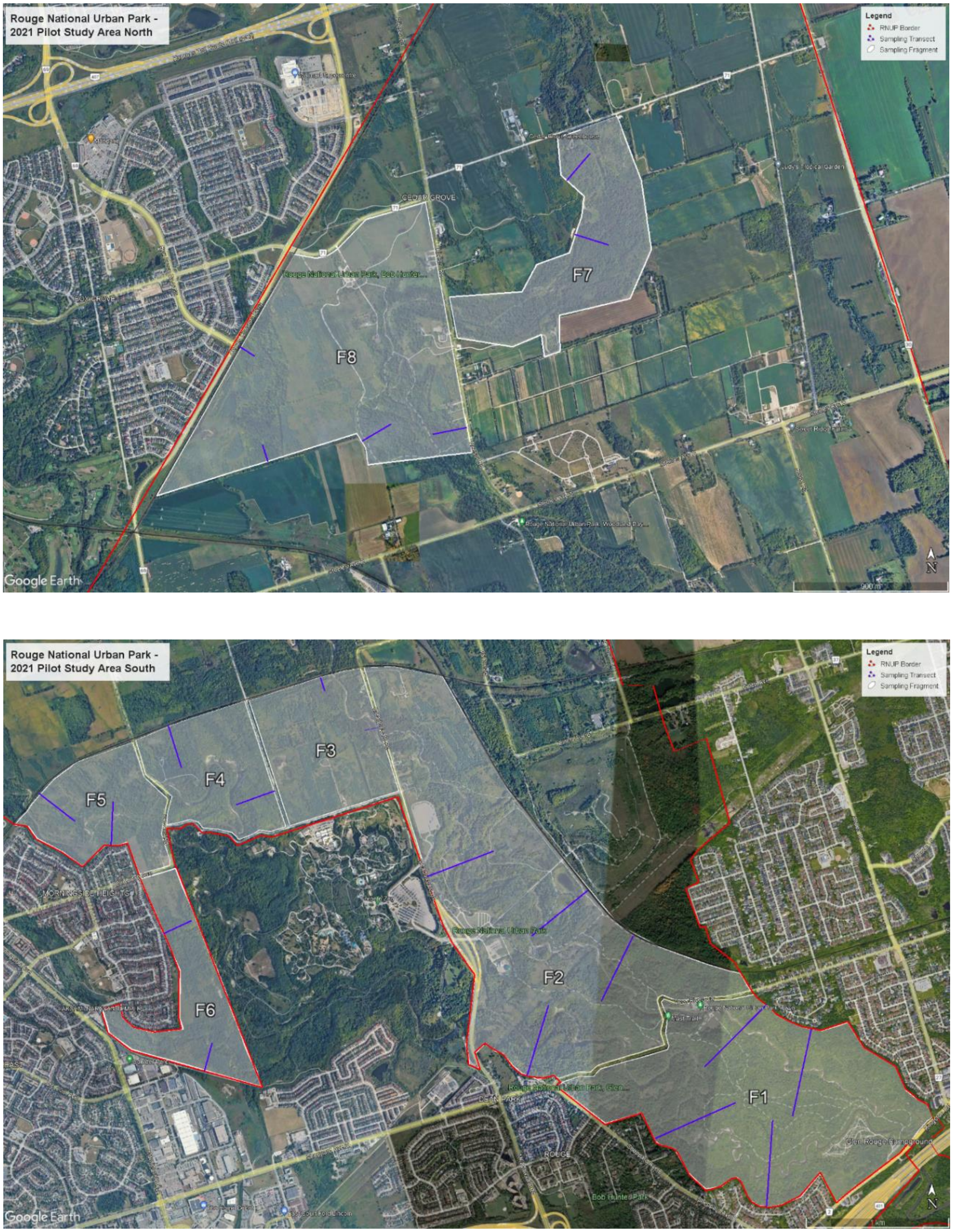

Figure 2 – 2021 Pilot Study Area
Map of initially proposed sampling areas for the 2021 pilot study. (a) Fragments 7 and 8 are south of Highway 407 and are adjacent to Bob Hunter Memorial Park. (b) Fragments 1-6 are north of Highway 401 and constitutes most of the sampling area. Sampling fragments were designated based on high contrast edges such as roads, rail lines, agricultural fields, and jurisdictional borders. Due to issues with land access and transitioning into the winter months, data was only collected in Fragments 1 and 2. Fragments 3 and 4 were not visited due to an active police investigation along Reesor Road. Fragments 5 to 8 were visited with observational notes being taken.

Figure 3 – 2022 Primary Study Area
Map of sampling area for the 2022 primary study, adjusted based on findings from the 2021 pilot. The scale of the study was reduced to areas with good land access, allowing for enough transects to be drawn. The northern edge of Fragments 1 and 2 were merged into a single observable edge with a total of 13 transects being sampled. Data collection remained south of the Hydro Corridor and north of Highway 401.

2.2 Methods

2.2.1 Data Collection

Data was focused on changes in vegetation composition as a function of their distance from a given point along a fragment edge at d = 0 m. There were two phases for this study: a pilot study to determine the necessary sample size and a subsequent primary study based on the formers' results.

The pilot study took place from September to November of 2021. Since sampling of the entire park was not feasible, potential sampling areas were held north of Highway 401 and south of Highway 407 (Figure 2). This decision avoided sampling of wetlands and farmlands. The park was further divided along major sources of anthropogenic fragmentation, such as roads and railways (Figure 2). This division would limit the number of confounding variables from multiple nearby edges. Other potential sources of habitat fragmentation, such as the two rivers running through RNUP and the trail network, were not observed for the purposes of this study, though their potential impacts were acknowledged. A total of eight fragments were outlined all varying in size. Fragments 1 and 2 were the largest (Figure 2). Some areas required further division due to lands being privately held, resulting in smaller accessible park area than appears on a map. For each fragment, random points along the perimeter were selected and a perpendicular transect was drawn. Spacing between transects was a consideration to ensure different areas of the park were sampled, though it was not systematically measured. Both the number and length of the transects were determined based on the relative size of the fragment. For instance, Fragment 1 had the largest total area, so four transects 600 m in length were drawn. Fragment 2 had less area by comparison, so four transects 500 m in length were drawn (Figure 2).

For a given transect, sampling points were predetermined at distances proportional to the transect length: 0, 0.1, 0.25, 0.5, and 1.0. These proportions allowed more samples to be taken closer to the edge where DEI was more likely to be measured while still allowing for larger measurements. A 5 m × 5 m quadrat was used to create the sampling area. Species data was collected on all vegetation found within the quadrat, with some exceptions. Data was not collected on dead/dormant vegetation, vegetation shorter than 5cm, and all grasses. For the remaining specimens, both the species and number specimens were recorded. In addition, the data was categorized by height: ground layer (<1 m), sub-canopy (1-5 m), and canopy (>5 m).

The primary study took place in June and July of 2022 and was adjusted based on the findings of the pilot study. Changes were made as to which areas were sampled, reducing the scale from the areas of RNUP between Highway 401 and 407 to the areas directly adjacent to Twyn Rivers Drive (Figure 3). Issues with safety and site accessibility meant sampling sites had to be more selective to maximize data collection. Due to ease of access, the northern borders of Fragments 1 and 2 (Figure 3) were determined as the sole edge to be sampled. Though this would produce some bias and exclude other regions of RNUP from investigation, it enabled enough transects to be drawn to garner statistically significant results. The labelling of Fragments 1 and 2 were kept to differentiate which side of Twyn Rivers Drive was being sampled, south and north respectively. For the purposes of analysis, the data was merged and considered as coming from a single edge. A total of 13 transects were drawn with nine in Fragment 1 and four in Fragment 2 (Figure 3). Transects were standardized to 500 m in length with the same proportions outlined in the pilot used to determine sampling distance from the edge. Quadrat size and data collected were consistent with the pilot study.

2.2.2 Statistical Analysis

Data was organized using Microsoft Excel (2022) while the analysis was performed using RStudio (2022). The package 'vegan' was installed to enable the calculation of species diversity using the Shannon Diversity Index. For each quadrat, the height categorization was removed and metrics for the entirety of the sample was calculated. For the pilot study, the standard deviation was calculated for the range of species diversity values found in Fragments 1 and 2. A power t-test was then employed to calculate the effective sample size. For the 2022 primary study, once species diversity was calculated, RTEI without blocking was used to determined DEI. Species evenness as determined by the Pielou Species Evenness Index was an added indicator for the primary study. It was calculated and analyzed using RTEI without blocking.

The steps for the RTEI method are as follows and were performed for each response variable for each distance *d* from the edge (Harper & Macdonald, 2011):

1. Calculate the observed magnitude of edge influence (MEI) at distance *d* using samples for the reference system and edge habitat.
    a. MEI = (e-r)/(e+r)
    b. Where 'e' is the response variable for a given distance from the edge and 'r' is the reference data set.
2. Calculation of randomized MEI:
    a. Without blocking: Create a data set using all edge data at distance *d* and all reference data resulting in a total sample size of *x*+*y*. Randomly select *x* values from this data set and calculate MEI using *x* values as the edge sample and *y* values as the reference sample.
    b. With blocking: For each transect, create a data set using the edge sample point at distance *d* and all reference sample points on the same transect for a total of 1+*z* sample points. Randomly select one of these as the edge sample point and use it to calculate MEI, with the remaining sample points acting as the reference sample.
3. Repeat step 2 numerous times. 5000 permutations are the recommended minimum to achieve a 1% confidence interval. This will create the randomized set of MEIs.
4. Within the distribution of the randomized MEIs, determine the percentile of observed MEI. The p-value is equal to this percentile for a one-tailed test, or two times for a two-tailed test.

If the p-value is lower than the determined level of significance (α), then reject the null hypothesis, concluding that MEI is significantly different than 0 at distance *d.* DEI is estimated to be the set of distances where MEI is found to be significant.

Initially for this study, a blocking design was used to link the reference data with an associated transect at *d* = 500 m. The reasoning was that it aligned better with the novel ecosystem framework as reference conditions were categorized as those in the deepest interior of RNUP as opposed to a separate location. After careful consideration, RTEI without blocking was the chosen method of analysis, though no changes were made regarding the selection of the reference sample.

2.3 Results

2.3.1 Pilot Study 2021

Figure 4 – 2021 Pilot Study Fragment 1 Sampling Area
Map of accessible Fragment 1 sampling area, marking where formal data was collected. The northern edge had the most accessible land, allowing for data collection to take place along the entire transect. Movement was aided by an abundance of informal trails. The southern edge was less accessible with physical barriers and hazardous conditions not making site visits possible. Transects 3 and 4 were partially sampled for this reason.

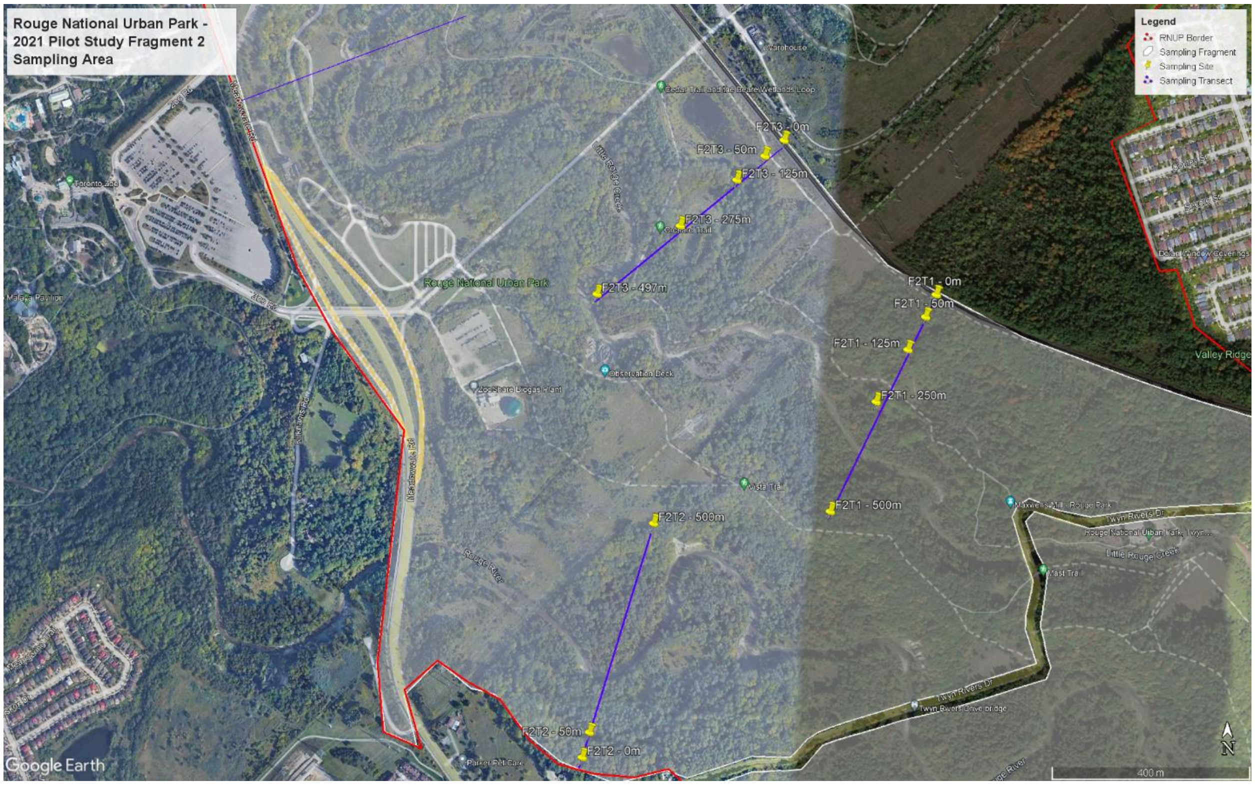

Figure 5 – 2021 Pilot Study Fragment 2 Sampling Area
Map of accessible Fragment 2 sampling area, marking where formal data was collected. Transects 2 and 3 were fully accessible, though some adjustments were made to the latter to avoid wetland areas. Transect 2 had some safety concerns with low visibility and a possible encampment, so no data was collected at d = 125 m and d = 250 m. The remaining northern most transect was not accessible as the land is held by the Toronto Zoo and was therefore restricted. This fact was not obvious through satellite imagery.

For the pilot study, eight fragments and 22 transects were initially identified for data collection (Figure 2). At the conclusion of the pilot, formal data was collected from only Fragments 1 and 2 (see captions in Figures 4 and 5 for why some potential sites were omitted). For F1T3 and F1T4, no data was collected from d = 0 m to d = 150 m due to hazardous site conditions and restricted site access. For F2T2, no data was collected at d = 125 m and d = 250 m due to safety concerns about a potential encampment. No data was collected for F2T4 due to restricted land access. The result was a total of 20 sampling sites. With the information available, it was determined through a power analysis that a minimum sample size of 15 transects in Fragment 1 and eight transects in Fragment 2 would be required for the primary study in the subsequent year.

A total of 60 different plant species were identified (Table 3). Of those found, 52 are considered native to Ontario, one is naturalized, and seven are listed as invasive. Dog strangling vine (a hard-to-separate mix of *Vincetoxicum rossicum* and *V. nigrum*) was the most observed species complex, having been found present in 9/20 sampling sites in high quantities. For example, in sample F1T1 at d = 300 m (Figure 4), it was the only observed species with an estimated 850 individuals counted.

There were logistical hurdles experienced in the field, creating unforeseen safety concerns. While the sloping nature of the terrain was known prior, it was underestimated how much that would impact travel time from one sample site to the next. Though some sites were only 100 m apart on a transect, travel around steep slopes and rivers was often necessary. Some slopes were moderate and thus navigable. Others were steep with dense layers of vegetation creating low ground visibility, hiding potential drops and unstable terrain. In some areas, the shrub layer was so dense that it created a physical barrier. Movement was inhibited at times by built infrastructure, with one instance of an old barbed-wire fence restricting access to sampling sites at F1T3 and F1T4 at d = 250 m. These issues with traversal lengthened the sampling time into early winter. Once it had reached a point where most of the vegetation had gone dormant, it was no longer feasible to collect formal data. At this stage, observational notes were taken at sites that could be visited.

The presence of private property within RNUP became an issue as it did not allow for sampling of previously planned areas. In Fragment 2, a fourth transect in the northern region could not be visited because it crossed over land owned by the Toronto Zoo (Figure 5). This fact was not obvious through online maps and was only discovered once on site. Access to some edges were blocked by residential housing, most notably the southern edge of Fragment 1.

Of note, two instances of reported criminal activity in RNUP increased concerns about travelling the area, especially those sites further away from formal infrastructure. The areas near Reesor Road were restricted as an ongoing police investigation into criminal activity was being conducted. In September of 2021, a body had been found after a grass fire had been reported (Freeman, 2021b). This came a month after another homicide incident had occurred with a body having been found in a burning car (Freeman, 2021a). These reports created a level of fear and uncertainty while traveling the park alone. More isolated areas with low visibility were avoided, and the area around Reesor Road was not visited at the request of Parks Canada. Fragments 3 and 4 were not visited for this reason.

In summary, changes needed to be made for the primary study so that I did not run into the same issues that resulted in missing data points. I required 15 transects for Fragment 1 and eight for Fragment 2. I needed to sample areas that were accessible and safe to navigate. It had to be possible to collect all the necessary data within the two months for which the study had been scheduled to avoid variation due to seasonal changes. The result was a narrowing of scope which focused on the northern edge of Fragments 1 and 2 (Figure 3). The area is well traveled, with an array of formal and informal trails granting access to more remote areas of the park. Visibility at a distance was good and the pitch of the slope was mild enough in certain areas to allow for navigation. The edge to be sampled was long enough to allow for sufficient transects to be drawn and less variable results to be measured due to the use of RTEI.

Table 1 – Mean species diversity for 2021 pilot study Fragment 1, calculated using the Shannon Diversity Index.

	H'
0 m	1.59
60 m	1.73
150 m	0.15
300 m	0.94
600 m	0.82

Table 2 – Mean species diversity for 2021 pilot study Fragment 2, calculated using the Shannon Diversity Index.

	H'
0 m	0.72
50 m	0.68
125 m	0.70
250 m	0.78
500 m	0.83

Table 3 - Summary statistics for the 2021 pilot study, merging data from both Fragments 1 and 2. It outlines how much data was collected and breaks down the kinds of plant species that were observed. During data collection, species were categorized by height as a general indicator of age structure: canopy (>5 m), sub-canopy (1-5 m), and ground layer (<1 m). The data was later merged for the purposes of calculating species diversity.

Transects sampled	7
Sampling sites	20
Unique species observed	60
Native species observed	52
Naturalized species observed	1
Invasive species observed	7
Species observed in canopy	17
Species observed in sub-canopy	18
Species observed in ground layer	44

2.3.2 Primary Study 2022

Figure 6 – 2022 Primary Study Fragment 1 Sampling Area
Map of sampling area for Fragment 1 for the 2022 primary study. There were nine transects 500 m in length. Quadrats were selected systematically with distances calculated at the following proportions of the total transect length: 0, 0.1, 0.25, 0.5, 1.0. The edge was dictated by the jurisdictional borders of RNUP and marked by fenced suburban backyards. The area around F1T7 was a public park with no physical barriers dividing the RNUP from the adjacent municipal area.

Data was collected from 13 transects each with five distances sampled using the northern edge of Fragments 1 and 2 for a total of 65 sampling sites (Figure 6; Figure 7). The naming convention for the fragments from the pilot study were maintained, but the data was merged into one data set. Once the mean species diversity for each distance from the edge was calculated, RTEI without blocking was used for the remainder of the analysis. The MEI for d = 0 m to d = 250 m were each compared to the MEI at d = 500 m, which was designated as the reference sample. 5000 permutations were performed with α = 0.01. Statistical analysis showed the MEI was not significantly different at any distance measured (Table 4). Each step was repeated for comparison of species evenness, with similar results (Table 5). For both species diversity and species evenness, DEI was not observable.

A total of 101 unique plant species were identified, with 79 being native to Ontario, 12 having been naturalized, and 10 listed as invasive (Table 6). *V. rossicum/nigrum* was the most observed species having be found in 43/65 sampling sites, with quantities ranging anywhere from 1-450 individuals.

Common buckthorn (*Rhamnus cathartica*) was the second most observed species, followed by sugar maple (*Acer saccharum*).

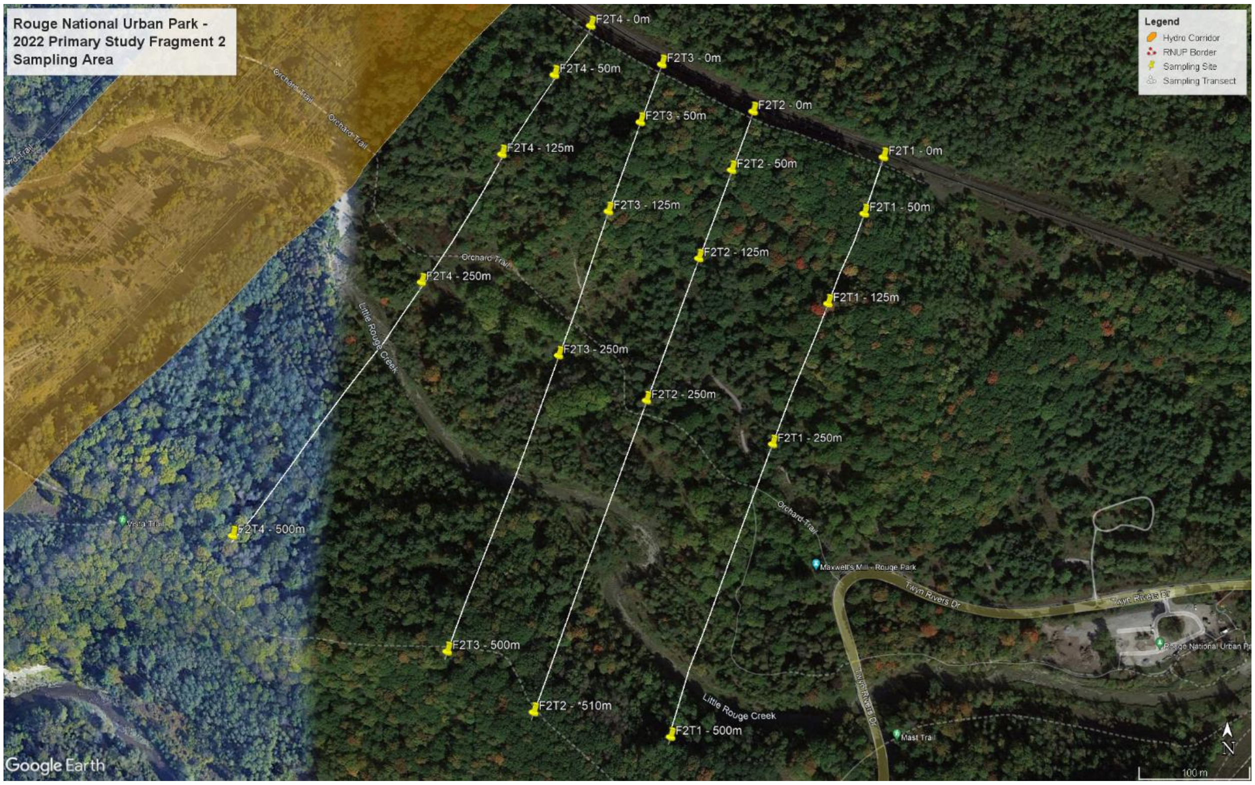

Figure 7 – 2022 Primary Study Fragment 2 Sampling Area
Map of sampling area for Fragment 2 for the 2022 primary study. There were four transects 500 m in length. Quadrats were selected systematically with distances calculated at the following proportions of the total transect length: 0, 0.1, 0.25, 0.5, 1.0. The edge was dictated by an active rail line that was further separated with a chain link fence. The forest continues along the northern side of the tracks before reaching a suburban neighbourhood in the city of Pickering. F2T2 was adjusted to d = 510 m to avoid sampling on an active trail.

Additional observations were noted in conjunction with the formal data collected on species diversity and species evenness. Traversal was made easy in large part due to an extensive informal trail network north of Little Rouge Creek. These trails are not officially recognized and do not appear on Google Maps imagery, but their presence is known to residents of the area. Evidence of human activity, such as a rusted bed frame, hanging ropes from tire swings, and even an old dirt bike area with built up mounds (Figure 11), were found up to 150 m from the edge, near the bottom of the valley slope. Deer were consistently sighted in Fragment 1 just north of Little Rouge Creek, specifically while sampling F1T4, F1T5, and F1T6 at *d* = 250 m (Figure 6). From all areas of the park, urban noise was audible whether from planes flying overhead or traffic on Highway 401.

Table 4 – Results of RTEI analysis of species diversity for 2022 primary study. Mean species diversity, calculated using the Shannon Diversity Index, is displayed and was the selected response variable for the study. Using RTEI without blocking, the set of randomized MEI for each distance from d = 0 m to d = 250 m was compared against reference data at d = 500 m. 5000 permutations were used for a significance level of α = 0.01 with a critical value of ±2.79. DEI is estimated to be the set of distances where MEI was found to be significantly different.

	H’	Observed MEI	Randomized MEI	Standard Deviation	Z-score
0 m	1.43	0.013	-0.005	0.23	0.30
50 m	1.24	-0.06	-0.006	0.46	-0.60
125 m	1.15	-0.096	0.0005	0.29	-1.72
250 m	1.26	-0.051	-0.0007	0.37	-0.70
500 m	1.40	N/A	N/A	N/A	N/A

Table 5 – Results of RTEI analysis of species evenness for 2022 primary study. Mean species evenness, calculated using the Pielou Evenness Index, is displayed and was an additional response variable selected for the primary study. Using RTEI without blocking, the set of randomized MEI for each distance from d = 0 m to d = 250 m was compared against the reference data at d = 500 m. 5000 permutations were used for a significance level of α = 0.01 with a critical value of ±2.79. DEI is estimated to be the set of distances where MEI was found to be significantly different.

	J’	Observed MEI	Randomized MEI	Standard Deviation	Z-score
0 m	0.62	-0.08	0.01	0.26	-1.76
50 m	0.58	-0.15	-0.01	0.37	-1.99
125 m	0.70	-0.02	-0.0006	0.20	-0.45
250 m	0.66	-0.05	-0.003	0.31	-0.71
500 m	0.73	N/A	N/A	N/A	N/A

Table 6 – Summary statistics for 2022 primary study. It outlines how much data was collected and breaks down the kinds of plant species that were observed. During data collection, species were categorized by height as a general indicator of age structure: canopy (>5 m), sub-canopy (1-5 m), and ground layer (<1 m). The data was merged for the purposes of analysis.

Transects sampled	13
Sampling sites	65
Unique species observed	101
Native species observed	79
Naturalized species observed	12
Invasive species observed	10
Species observed in canopy	21
Species observed in sub-canopy	19
Species observed in ground layer	87

2.4 Discussion

2.4.1 Distance of Edge Influence in Rouge National Urban Park

The purpose of this study was to begin building an understanding of the inherent restoration limitations of an urban protected area by measuring the amount of least impacted area, i.e., the interior forest. In Rouge National Urban Park, when comparing vegetative species' response to presumed anthropogenic forces at an observed urban edge, in terms of both species diversity and species evenness, distance of edge influence was not observable. One possibility for the results is that there was too much interference from other sources of fragmentation for which could not reasonably be controlled. In addition, the observed edge did not resemble typical open urban edge conditions. The more closed nature of the edge may be producing a weak edge effect signal that was not detectable at the distances measured. Though an exact measurement of DEI was not obtained, field observations still allow for some insight into how RNUP is being impacted by its surrounding urban environment.

DEI based on vegetation sampling has been found from 15 m up to 1 km, though in temperate forests 50 m is typically the maximum value (Beacon Environmental, 2012; Hamberg et al., 2009; Lehvävirta et al., 2014; Vallet et al., 2010). It must be noted that most studies examined forest edges in a rural matrix, with little information on urban areas (Vallet et al., 2010). Of those that measured urban edge effects, the distinction between the edge and forest interior was less clearly marked with no species having been found to be characteristic of urban edge environments (Guirado et al., 2006; Hamberg et al., 2009; Lehvävirta et al., 2014; Vallet et al., 2010). By virtue of my sampling design, $d = 50\ m$ would have been the minimum DEI value that could have been obtained being the shortest sampled distance from the edge. Sampling distances were set up with the expectation of finding a significantly larger DEI due to the perceived magnitude of anthropogenic influence. It is possible that more granular increments may have produced a DEI measurement less than $d = 50\ m$ which would be more in line with the literature. It is important to note that even at $d = 0\ m$ samples were not significantly different from the reference site, signalling other factors may be at play.

A direct comparison of my results to the broader literature is difficult due to the differences in matrix type, sampling design, and statistical modelling (Harper & Macdonald, 2011; Vallet et al., 2010). As previously highlighted, known DEI measurements are based on forest edges in a rural matrix and would not have encompassed the same influencing forces found in an urban matrix (Vallet et al., 2010). Differences in sampling design will have produced varied results despite similar statistical modelling (Harper & Macdonald, 2011). My use of RTEI, while not necessarily more accurate, would have accounted for variations in sampling design, but only as it compares to other studies that have used RTEI (Harper & Macdonald, 2011). There are only a few studies, and they examined both different response variables and matrix types (Harper et al., 2014; Harper et al., 2015; Medeiros et al., 2023). For example, Harper et al. (2015) used RTEI, both with and without blocking depending on the site, to measure edge influence in boreal forests because of timber harvesting. They found edge responses to generally be weak, with varied DEI measurements less than *d = 50 m.*

I had considered if DEI was of a large enough magnitude to not be captured by the distances measured. As *d = 500 m* was the designated reference sample, *d = 250 m* was the furthest measured distance from the edge. It is possible that the true DEI falls somewhere in between these distances, though additional measurements would be needed for verification. A DEI greater than *d = 500 m* seems unlikely since beyond this distance the transect begins approaching the southern edge of RNUP.

It seems likely that interference from multiple nearby edges may have produced confounding variables, leading to an inconclusive DEI. Eigenbrod, Hecnar, & Fahrig (2009) measured what they called the road-effect zone of Highway 401 to see at what distances was it negatively impacting anuran populations. For five of the seven species studied, they found a road-effect zone of 250-1000 m. Highway 401 directly intersects RNUP and sits 100 m south of Fragment 1 (Figure 1). Twyn Rivers Drive is another road that intersects the study area, providing another road-effect zone between Fragments 1 and 2 (Figure 3). Estimates will vary depending on the design and usage of the road, as well as the species or response variable being studied (Peaden et al., 2015; Pocock & Lawrence, 2005; Shanley & Pyare, 2011). Low traffic country roads have been found to have a 230 m road-effect zone in relation to the Mojave Desert tortoise (*Gopherus agassizii*) (Peaden et al., 2015). In terms of changes in vegetation, Pocock & Lawrence (2005) found exotic species to extend up to 50 m from the road. Though varied in degree, it is possible that the road-effect zone from these pieces of infrastructure in RNUP would interfere with the precision of my results as they overlap the selected study area.

Trails are another source of anthropogenic fragmentation that may have created data noise. Edge effects can range anywhere from 5-50 m depending on factors related to usage and their physical makeup (Ballantyne et al., 2014; Pickering et al., 2012). In the studied area of RNUP, the formal trail network lays in the interior and was thought to have a low enough signal as to not interfere with the expected results. From initial satellite imagery, there were only a few major trails that ran perpendicular to the sampling transects (Figure 6; Figure 7). In Fragment 2, they were closest to sampling sites at *d* =

250 m and d = 500 m. In Fragment 1, the trails were closest to reference sites at d = 500 m. While the presence of informal trails was a possibility, the scale of the network was unknown prior to site visitation. They were found densest up to 100 m from the observed edge, though some areas had trails leading deeper into the interior. Both the age, amount, and kinds of usage are unknown, but even a modest estimate of DEI would remove significant amounts of what would have been contiguous forest (Ballantyne et al., 2014).

The fragmented nature of RNUP coupled with recreational use meant that some level of interference was inevitable. While planning the initial pilot study, efforts were made to limit the impact by observing anthropogenic edges with a significant level of disturbance such as high-volume roads, railways, and the outer RNUP border. During the primary study, the moderate traffic of Twyn Rivers Drive was thought to have a low enough signal to allow for the combining of Fragments 1 and 2 into a single studied edge. It also constituted a different source of fragmentation than the fenced border of RNUP. Had it been possible to divide the region by these additional sources of fragmentation, the sampling fragments would only be smaller, decreasing the amount of potential interior conditions.

Figure 8 displays hypothetical maps of Fragment 1 that help to demonstrate the issue. If a DEI of 50 m is assumed (Ballantyne et al., 2014; Vallet et al., 2010), it would leave a modest amount of interior habitat (Figure 8a). The informal trail network, however, extends up to 100 m into the interior with as much as 50 m of edge influence. That would create a potential cumulative DEI of 150 m into the interior. It is now approaching Little Rouge Creek which, while natural, can produce its own edge effects up to 50 m (Ballantyne et al., 2014). The result would be virtually no undisturbed habitat north of Little Rouge Creek and south of the observed edge (Figure 8b). When examining the reference interior, it runs into similar issues with edge effects from both the adjacent rivers and formal trail network. This map does not highlight the presence of informal trails south of Little Rouge Creek, which were observed while collecting data. To get an accurate measurement of DEI in this section of RNUP, a more granular scale may be required.

a)

b)

Figure 8 – Map of Theoretical Edge of Influence

Map of a theoretical edge of influence based on average estimates for a temperate forest (Ballantyne et al., 2014; Vallet et al., 2010). (a) This map presents a DEI of 50 m, which is typical of temperate forest in a rural matrix. (b) This map presents a cumulative DEI if the edge effects of the informal trail network are included.

Another potential cause of an indeterminate DEI may be the lack of assumed edge disturbances leading to weak interior penetration. In this instance, the selected response variables of species diversity and species evenness may not have adequately captured the changes in vegetative composition. I had assumed the edges of RNUP would be marked with an abrupt transition to urban infrastructure. In this case there would be higher amounts of solar radiation and increasing air temperature (Hamberg et al., 2009; Lehvävirta et al., 2014; Vallet et al., 2010). Urban edges often have a high abundance of non-native species and increased light levels depending on the canopy structure (Hamberg et al., 2009; Vallet et al., 2010). I had expected to find similar conditions based on observing urban edges around RNUP, which are often populated with species like Canada goldenrod (*Solidago canadensis*), *V. rossicum/nigrum*, or phragmites (*Phragmites australis*). Hamberg et al. (2009) found similar results when examining the effects of canopy structure on edge influence. They defined closed edges as those with a high abundance of trees, shrubs, and saplings, and if it was difficult to see into the forest patch, i.e., 75-95% side-canopy cover. At these edges they found edge response to generally be weak or even absent due to the decreasing influence of light, heat, and wind.

In the studied area of RNUP, while the change in habitat was abrupt and usually marked with a chain-link fence, the edge was more sheltered than anticipated and could be categorized as a closed edge (Figure 9). Rather than transitioning into open urban development, the sampled edge ran along private suburban backyards with diverse management styles ranging from large patios and pools to typical gardens and turf-type lawn grass. In addition to a mature canopy cover with a north facing edge, across the yards were two-story suburban homes. The exception was Fragment 2 which bordered an active rail line with the forest continuing on the other side. In this section, the side-canopy was thick with vegetation growing into the bordering fence. The result was that samples at $d = 0\ m$ had moderate to heavy shade, favouring low-light species (Hamberg et al., 2009). Wind and anthropogenic disturbances may be low when compared to edges that open toward a busy road, as the fence would be expected to contain most human activities to their backyards (Hamberg et al., 2009). It may hinder wildlife movement, further reducing the exchange of materials across the edge. RNUP does possess more typical open urban edge conditions, but it was not characteristic of the sampled edge. Relative to the former, the amount of disturbance at the sampled edge may be lower, leading to a weak or absent signal. Additional response variables that better capture vegetative changes at this kind of edge may reveal a measurable DEI.

It will be important to note observed conditions throughout the park to further contextualize the results and pull some insights in the absence of a measurable DEI. While I did not examine specific forest species, a key distinction I observed between conditions near the identified edge and relative interior was the lack of forest floor vegetation. At d = 500 m, vegetation large enough to sample were few and far between and at times non-existent (Figure 9b). These observations were made despite this area having the most mature tree canopy. There was one instance where a break in the canopy allowed for a section of garlic mustard (*Alliaria petiolata*) to proliferate, but in general there was a severe lack of young vegetation. Over-browsing by deer in the area is known to be contributing to a thinning forest floor, but trampling may also be a significant factor (Lehvävirta et al., 2014; Russell et al., 2017; Toronto and Region Conservation Authority, 2015).

Unlike other studies which used an "intact" interior (Laurence & Yansen 1991; Lehvävirta et al., 2014; Vallet et al., 2010), as an urban national park RNUP is regularly used for recreation. The reference sites for this study were in an area that holds a heavily used artery of the trail network, connecting the Glen Rouge Campground to the Rouge River Lookout and Toronto Zoo. The presence of this trail network was known prior to data collection and its potential influence noted. It was thought that by comparison to an anticipated urban edge with higher degrees of activity and disturbance, the area would be least impacted despite the trail network. The selection of a reference site within RNUP with fewer and less travelled trails may prove a good comparison for future studies.


a) F1T1 at d = 0 m (left) and d = 500 m (right)


b) F1T6 at d = 0 m (left) and d = 500 m (right)


c) F2T2 at d = 0 m (left) and d = 500 m (right)

Figure 9 – Comparison photos of edge and reference sites

Sample photos from the 2022 primary study comparing edge conditions at d = 0 m to reference site conditions at d = 500 m for (a) F1T1, (b) F1T6, and (c) F2T2. The edge along Fragment 1 bordered a row of fenced suburban backyards. A 10 m buffer was maintained to respect residents' privacy. The edge along Fragment 2 lay directly parallel an old chain link fence with an active rail line directly across. Reference sites varied in the amount of ground vegetation. In general, they were closest to the most actively used formal trails in RNUP.

The prevalence of invasive species was noted *as V. rossicum/nigrum*, *A. petiolata*, and *R. cathartica* were commonly observed. No pattern was immediately evident between distance from the edge and level of abundance. The heterogenous nature of RNUP meant that where conditions were favourable, pockets of these invasive species would form, generally overtaking the entire area. *V. rossicum/nigrum* was the most observed species, having been found in almost every area of the park. In at least one sampling site, it completely covered the forest floor and was the only species recorded. Bugiel et al. (2018) examined the impacts of *V. rossicum* in RNUP on soil biodiversity and found that plant invasion was contributing to reduced variation in soil bacterial communities. They found no relation with the distance from an anthropogenic edge. During both the pilot and primary study, dense areas of *V. rossicum/nigrum* were often observed in areas with increased light exposure from natural breaks in the canopy. The dominance of this species may have long-term impacts on plant-soil interactions and future restoration opportunities.

Based on the data in conjunction with observed landscape features, it seems more likely than not that the studied area of RNUP is fragmented to such a degree that it consists of mostly edge conditions. This line of reasoning is consistent with my initial hypothesis, which stated that the impact of nearby urban activities would be so great as to leave a diminished amount of interior habitat, having implications for future restoration plans. The difference is the lack of interior conditions may be due to micro fragmentation from anthropogenic activities occurring throughout the park as opposed to intense activity outside of its borders. More research is needed to better understand the drivers behind these local conditions.

a) F1T3 at d = 0 m

b) F1T3 at d = 50 m

c) F1T3 at d = 125 m

d) F1T3 at d = 250 m

e) F1T3 at d = 500 m

Figure 10 – Photos of Fragment 1 Transect 3
Photos of Fragment 1 Transect 3 from the 2022 primary study at every measured distance: (a) d = 0 m, (b) d = 50 m, (c) d = 125 m, (d) d = 250 m, and (e) d = 500 m. While analysis showed there was no significant difference in terms of species diversity and species evenness across distances from the edge, observed species were more indicative of a wetland habitat at (d) d = 250 m as sampling neared the Little Rouge Creek.

2.4.2 Assumptions and Gaps in Knowledge

Data was collected along a single edge, meaning these results cannot be generalized to other areas of RNUP. Other sources of fragmentation such as roads, trails, and agricultural fields will produce varied edge conditions and require targeted data collection for that specific edge type (Ballantyne et al., 2014; Eigenbrod, Hecnar, & Fahrig, 2009; Vallet et al., 2010). While the 2022 primary study focused on the northern edge of Fragments 1 and 2, I had explored other edges during 2021 pilot study. The southern border of Fragment 1 had a more exposed forest edge typical of urban environments, as it backed Sheppard Avenue East (Figure 1) as opposed to fenced backyards. The slope was steeper, canopy immature, and the ground vegetation was denser. Some edges in RNUP backed a busy rural road while others faced an active agricultural field. Fragments 1 and 2 were dynamic in their elevation changes, while others had consistently level ground. Other parts of RNUP see less foot traffic and may have fewer informal trails with different patterns of use. The amount of variability in RNUP in terms of both edge types and environmental conditions means that the lack of observable DEI is not necessarily applicable to other edges of the park. Vallet et al. (2010) compared the impact of rural and urban edges on woodland plots and found differences in species composition, soil conditions, and the degree of clarity seen when transitioning from edge to interior conditions. Rural edges had unique community compositions with several species only occurring in that range, whereas urban woodlands had no clear demarcation. The diversity of activities occurring in and around RNUP is remarkable, but it means that research must adapt to these varied conditions.

The choice of species diversity and species evenness as indicators bears discussion. These metrics were chosen to capture data on the complex matrix of influences acting on RNUP. There are too many anthropogenic forces to effectively measure them all in the allotted time, and to choose one over the other would have narrowed the scope of the study more than was preferred. The assumption was that the culmination of these forces would influence the vegetative make-up at the edges of RNUP and lose power as one moved further into the interior. Species diversity was expected to be highest at the edge where conditions were thought to be more favourable to a wider variety of vegetation due to things like frequent disturbances and increased sun exposure (Hamberg et al., 2009; Lehvävirta et al., 2014; Vallet et al., 2010). The sampled edge for the 2022 primary study did not follow this pattern, so differences in species diversity across distances may not have been as distinct. Species evenness was compared in a response to this result, as some areas being dominated by select species led me to wonder if a different DEI would be found. The results were the same. A wider set of response variables with a closer examination of the kinds of species growing at the edge would benefit future studies (Harper et al., 2015; Vallet et al., 2010).

The primary question that remains is what can be understood from a DEI that was not observable. As discussed in the previous section, additional research would be needed to verify whether a forest edge adjacent to suburban backyards exhibits weak edge influence or an urban park like RNUP has so much edge influence that they need to be parsed out more meticulously. In either event, this research has revealed a deeper level of fragmentation than is apparent, but one that is also more complex. The number of edges coupled with the variety of edge types creates a challenge for measuring the amount of interior habitat and subsequently creating plans to manage it. The research conducted here was based on assumptions that, upon in field investigation, proved to be incorrect. The sampled edge was not heavily disturbed by ongoing anthropogenic activity and the fragment was not as contiguous as displayed on satellite imagery. These results provide a useful foundation for subsequent research in RNUP as well as ongoing management plans.

Assuming heavy fragmentation and no observable differences between edge and interior conditions regarding vegetative species diversity and species evenness, one must ask if this is a desirable environment. Per Parks Canada's (2020) management framework, evaluating the priorities for conservation and ecological restoration will be done based on community engagement and evidence-based science. Certain species of trees, birds, or insects may benefit from high levels of habitat fragmentation (Hamberg et al., 2008; Ikin et al., 2015; Lehvävirta et al., 2014). Lehvävirta et al. (2014) studying the effects of fragmentation and recreation on tree regeneration found that fragmentation favoured the regeneration of deciduous trees up to 80 m into the interior. This was supported by Hamberg et al. (2008) who found that urban edges favoured deciduous trees well adapted to sunny and dry conditions, which may be increasing soil fertility at the edge. Examining bird community responses to edges, Ikin et al. (2015) found that non-native birds such as the house sparrow (*Passer domesticus*) and common blackbird (*Turdus merula*) exhibited a weak edge response due to their ability to exploit nearby urban resources. Some native birds such as the pied currawong (*Strepera graculina*) and the red wattlebird (*Anthochaera carunculata*) relied more heavily on nearby nature reserves with good tree and shrub cover.

As Parks Canada finalizes its management goals, it will be useful know how different edges are impacting its target species, leading to potential control options. If a given species exhibits a weak edge response, then other management strategies aside from edge control may be explored (Ikin et al., 2015). The amount of edge habitat is only one of many factors that should be considered when making restoration plans.

2.4.3 Future Research Opportunities

My research was exploratory; it was designed to be the first of a series and intended to provide the necessary context to develop targeted restoration plans with increased efficacy. I recommend that the next phase expands my research into a two-sided study. This should provide increased explanatory power regarding the interactions of two adjacent environments (Fonesca & Joner 2007). Collecting data on the specific interactions between the natural and urban environments around RNUP would help identify disturbances impacting RNUP, as well as how different species are adapting to available resources (Dunford & Freemark, 2004). Numerous studies have explored the ways in which both wildlife and vegetation are changing as a result of urban environments, such as songbirds altering the frequency of their songs in response to urban noise (Bateman & Fleming, 2012; Dubois & Cheptou, 2017; Nemeth et al., 2013). These evolving adaptions may provide new management opportunities not previously realized. In addition, there is movement to provide more natural habitat such as planting native trees, creating pollinator gardens, and redesigning stormwater management (Canadian Wildlife Federation, n.d.; LEAF, 2022; Meadoway, 2019). The nature/urban dynamic is a two-way interaction with each influencing the other. If restoration in RNUP is to be successful, it will need to develop a deeper understanding of its urban context as it continues to evolve.

Figure 11 – Retired Dirt Bike Mounds
Old dirt bike mound found approximately 200 m into the interior while sampling F1T7. It is possible this is the same dirt bike area mentioned in the TRCA (2015) Biological Inventory. Such features highlight the history of unregulated use of the area that would have impacted RNUP's ecological development.

Effective ecological restoration in urban environments will require comprehension of the social and political dynamics of urban communities. This study was narrowly focused on ecological interactions with some observations made about the impacts of anthropogenic disturbances. I speculated about the impacts of informal trails and the placement of two-storey houses along the observed edge. I did not examine the ways in which people use these trails or manage their yards. There are people who follow best practices when on trails to limit their impact on nature, while others with less awareness may do some harm during their visit. Where a rail line cuts through, the density of new housing developments, how nearby greenspaces are managed, and what kinds of restoration projects are approved, these are all decisions that have both social and political elements to them with physical consequences to natural environments. The culture of the surrounding communities is a factor that cannot be taken for granted. The people of Pickering are not the same as the people of Scarborough, or even the farmers managing the agricultural fields in RNUP itself. The complexity of urban environments is only increased by the complexity of urban communities. Future studies should be more explicitly interdisciplinary. Doing so will not only create a more comprehensive view of the restoration context of RNUP, but it may reveal previously unforeseen barriers to effective urban restoration.

3.0 Implications for the Restoration Potential of Rouge National Urban Park

3.1 Diffuse Urban Impacts

In the context of restoration potential in RNUP, it puts into perspective some of the challenges Parks Canada may face moving forward. This study provides evidence of how embedded RNUP is within its urban context. Throughout the entirety of the observed area there were clear signs of human impact from the extensive informal trail network to old rope swings and discarded metal scraps. What was less obvious was the amount of diffuse urban influence that would be difficult to capture in an edge effect study. For instance, atmospheric nitrate is a non-point source air pollutant that may be impacting the entirety of the park. It would negatively affect native plants poorly adapted to high nitrogen environments (Toronto and Region Conservation Authority, 2015; Vallet et al., 2010).

Noise pollution was notable during the study, as whether it was Highway 401, first responder sirens, or planes flying overhead, it was near impossible to escape the feeling that I was in an urban environment. These high sound levels are known to have negative consequences for wildlife, resulting in a higher risk of predation and reduced breeding success (Eigenbrod, Hecnar, & Fahrig, 2009; Newport, Shorthouse, & Manning, 2014). The effects on songbirds have been studied extensively, as it has caused them to alter the pitch of their songs as a form of adaptation (Dowling, Luther, & Marra, 2012; Nemeth et al., 2013; Perillo et al., 2017). Despite this evolutionary change, increases in noise pollution have been correlated with decreases in species richness (González-Oreja, 2017, González-Oreja et al., 2012; Perillo et al., 2017). It has been recommended to increase greenspace area while decreasing noise levels to mitigate these impacts (González-Oreja, 2017, González-Oreja et al., 2012; Perillo et al., 2017), but as has been discussed, RNUP cannot expand beyond the lands that have already been acquired.

Ecological light pollution is another well documented issue with similar impacts on wildlife's ability to survive (Longcore & Rich, 2004; Newport, Shorthouse, & Manning, 2014). Urban environments, such as those just outside the borders of RNUP, are filled with numerous sources of artificial lighting that do not follow a natural day and night cycle. Like edge effects, the consequences can vary across both spatial and temporal scales as they can affect foraging, reproduction, communication, and other critical wildlife behaviours (Longcore & Rich, 2004). One key aspect is the disruption to natural predator-prey relations. For instance, it creates a new "night light niche" whereby insects are attracted to lights then creating a new concentration of food for animals such as bats on which to prey (Longcore & Rich, 2004). The issue is that this dynamic favours fast moving bats while disadvantaging slower-flying ones that tend to avoid these kinds of lit environments (Longcore & Rich, 2004). In RNUP, Parks Canada (2020) is implementing dark sky strategies on park lands in collaboration with park lessees. It states it will be working with external agencies to implement similar features in neighbouring communities. The question then becomes how much of the adjacent cities' lighting infrastructure needs to change to have an observable impact.

The fact that the sheer amount of data noise made it difficult to find a measurable DEI may mean it will be just as difficult to manage. Though Parks Canada has increased federal resources relative to land managers previously, it is not unlimited. As with many things reliant on government, should priorities change then Parks Canada's ability to manage the land effectively may be hindered. It cannot practically address every issue with the first iteration of its management plan, nor does it claim to be attempting it. Since restoration work must deal with the uncertainty of novel ecosystems, Parks Canada will need to continue monitoring the effects of its actions and adapt as needed (Prach et al., 2019). There is only so much it can change in RNUP to minimize some of these anthropogenic impacts. The key has always been to invest in a strategy that has the highest likelihood of meeting their predetermined restoration goals.

3.2 Buffer Areas and Regulating Human Impact

Parks Canada (2020) explicitly acknowledges it is working within the inherent limits of a naturalized space adjacent to an ever-growing GTA. Given the number of externalities influencing ecological development, it begs the question, what does restoration of RNUP then look like? The TRCA (2015) recommended the installation of sanctuary areas, where public access to high quality sensitive areas is restricted to prevent further biodiversity loss. I question the efficacy of such a strategy. Research is limited on the effectiveness of buffer zones in reducing the impact of anthropogenic activities (Beacon Environmental, 2012). At best, there are approximations about how much distance would need to be maintained based on calculated DEI (Beacon Environmental, 2012). In response to human activity plants have shown a response up to 50 m, invertebrates up to 100 m, and deer up to 390 m (Beacon Environmental, 2012). Whether or not these measurements would be sufficient for a buffer area is open to examination.

The amount of interior forest in RNUP is limited, with virtually no room for expansion due to the surrounding urban environments. Dunford & Freemark (2004) recommended a 200-1800 m buffer between forest patches and urban activity to support migratory bird species. Regarding vegetative composition, Lehvävirta et al. (2014) recommended a fragment diameter larger than 160 m while Hamberg et al. (2008) suggested a minimum area of 2-3 ha. Parks Canada would not be able to establish anything close to these distances, especially since the continued use of the land by residents is a key principle of their management strategy. It may be able to close some trails and key areas to prevent pedestrian traffic and limit impacts from pets. It is difficult, however, to say how much of that would result in a tangible difference given the amount of fragmentation and recreational use of an urban park like RNUP.

Another consideration is that such controls may be difficult to implement given the history of use of the area. Prior to Parks Canada, regulation of the Rouge Valley was relaxed with the responsibility being shared amongst a group of local organizations and government agencies (Livingstone, Cadotte, & Isaac, 2018). Many of the informal trails are old and well-travelled by residents. They know where the breaks in the fences are for entry, where they can have secret bonfires, and where their dog likes to walk. In the past, the newly created Beare Hill Park and nearby section of hydro corridor were areas that were supposed to have restricted access but have openly been used by locals for decades. The point is that any new measures for control may give rise to conflict or simply be ignored because for some, it will not be easy to change a lifetime's worth of behaviour (Soliku & Schraml, 2018). This potential source of tension may be further amplified by the growing popularity of RNUP and the lack of controlled entrances. The ease of access combined with the sheer population of the GTA means proper use may be difficult to encourage (Bueckert, 2021; Mathis & Rose, 2016; Swenson, 2019).

Parks Canada is not dealing with only local Indigenous groups with a long history on the land. It is attempting to manage RNUP with respect to multiple highly dynamic communities each with their own values and beliefs. Livingstone, Cadotte, & Isaac (2018) surveyed visitors of RNUP to see how they valued different ecosystem services. While agricultural services consistently ranked low, there were variations in what was valued more depending on how aware visitors were of local environmental issues, what they defined as being "ecologically engaged". In this sampled group, they found that the average RNUP visitor valued the park for its recreational benefits (i.e., cultural services) while those considered ecologically engaged prioritized supporting services such as pollination. When examining RNUP users, there appears to be three sets of values that need to be balanced. An improved trail network is needed to help people explore the park and better connect with Canada's natural heritage, in line with one of Parks Canada's guiding principles. These trails, however, will disturb and fragment the land people wish to appreciate and conserve (Ballantyne et al., 2014; Ballantyne & Pickering, 2015).

The low valuation of agricultural services is notable, as a majority of RNUP is reserved for crop production (Livingstone, Cadotte, & Isaac, 2018; Parks Canada, 2020). Protected farmland as a feature of the Canada National Park system is relatively novel and may be unfamiliar to the average park visitor (Livingstone, Cadotte, & Isaac, 2018). Farmers, on the other hand, may see this as a continuing trend of underappreciation from governments and environmentalists. Gill (2017) examined the power dynamics involved in the creation of RNUP and found some latent resentment amongst farmers. They felt environmentalists tend to overlook their value or underestimate the impact that restoration activities like tree planting have on their livelihoods. RNUP has an abundance of prime agricultural land with every hectare needed to support a stable crop production. When lands are marked for naturalization, it hurts a farmer's ability to make a living. It is made even worst when discussions seem non-existent, with farmers talking to government officials alone and rarely alongside self-identified environmentalists. As with so many PAs prior, putting the principles of effective community engagement into practice can be difficult. Gill (2017) provides some evidence as to the struggles involved in RNUP's initial establishment. Working to resolve these issues in the long term will be key to its restoration success.

3.3 Need for a Landscape Approach

Though the results of this research are preliminary, the main challenge for RNUP relates to the question of restoration versus conservation, i.e., can the ecological integrity of the park be significantly improved, or can it only be protected from further degradation? Protected areas like RNUP are often held up as a prime tool for reversing biodiversity loss (Santangeli et al., 2023). Yet their effectiveness is often overestimated with little research having adequately quantified their impact (Adams et al., 2023; Santangeli et al., 2023). Santangeli et al. (2023) found PAs to have mixed effects on boreal biodiversity with most species showing little to no response. Of those that did, the rate of extinction merely slowed. Effects were dependent on the species as well as the size and age of the PA. They noted that the low connectivity of the PA network may be contributing to the lack of improvement in biodiversity, with changes in species' populations outside of PAs also being uncommon. The issue is further highlighted by the Protected Planet Report which stated that, in 2020, only 7.84% of the earth's lands were both protected and geographically connected (Adams et al., 2023). This is made worse by that fact that many critical routes of connectivity are unprotected and susceptible to development (Adams et al., 2023).

Though I only explored one aspect of the urban/nature dynamic between RNUP and its adjacent communities, a broader landscape approach would be necessary for impactful restoration results (Adams et al., 2023; Dunford & Freemark, 2004; Fletcher et al., 2007; Hobbs et al., 2010; Hobbs et al., 2014; Kremer & Merenlender, 2018; Romanelli et al., 2023; Santangeli et al., 2023; Toronto and Region Conservation Authority, 2015). Urbanized areas like the GTA are a mosaic of historical, hybrid, and novel ecosystems offering a variety of resources and environmental conditions of which different species can take advantage (Dunford & Freemark, 2004; Hobbs et al., 2014). Managing a habitat patch in relation to the broader landscape enables a more comprehensive and transparent approach, whilst giving managers more options for restoration than they would have otherwise (Hobbs et al., 2014). Having a diversity of management options is critical because of the uncertainty associated with dynamic anthropogenic impacts (Hobbs et al., 2010). Due to climate change, current climatic conditions will represent less than 10% of PAs in 100 years, and a landscape approach is argued to be necessary for buffering these impacts (Kremer & Merenlender, 2018). It also creates opportunities to employ different strategies, either within a single area or across multiple greenspaces, so that community values can be better observed and learning opportunities can be maximized (Hobbs et al., 2010).

More importantly, a landscape approach is needed to better design for patch connectivity (Beninde, Veith, & Hochkirch, 2015; Hobbs et al., 2014; Rudd, Vala, & Schaefer, 2002). While patch size is important, connecting naturalized areas with functional habitat corridors can have just as great an impact on improving biodiversity (Beninde, Veith, & Hochkirch, 2015; Rudd, Vala, & Schaefer, 2002). In urban settings where historical areas are few and far between, landscape functionality can be supported through corridors of novel ecosystems (Hobbs et al., 2014). When sites are managed in isolation, it leads to a patchwork design with broad variations in forest composition and structure depending on their management goals (LaPaix & Freedman, 2010). Ideally, restoration efforts should be coordinated across a larger scale with deliberate actions that consider how a given area fits within the broader landscape (Hobbs et al., 2010). Like edge effect studies, habitat connectivity in an urban landscape requires more research to better understand their functionality in this setting (Beninde, Veith, & Hochkirch, 2015).

3.4 Emerging Restoration Opportunities in the Greater Toronto Area

It is difficult to say where these opportunities for landscape connectivity lie within the Greater Toronto Area and adjoining Durham region. As discussed in previous sections, this region of southern Ontario is heavily developed and only expected to intensify. RNUP itself is not a contiguous landscape, though examining the amount of functional connectivity was not within the scope of this research. That is not to say an urban PA provides no benefit. Controversies over development of the Ontario Green Belt have shown that the new protections afforded the lands, whether they be agricultural or natural, will prevent the further encroachment of urban development (CBC News, 2023; Jones, 2022; More Homes Built

Faster Act, 2022). This factor alone could yield numerous benefits related to ecosystem services, such as the continued maintenance of quality farmland, large amounts of accessible greenspace for the public, and improved connectivity of the watershed with the Oak Ridges Moraine (Kremer & Merenlender, 2018; Livingstone, Cadotte, & Isaac, 2018; Parks Canada, 2020). What it may struggle to be is a thriving naturalized habitat with major improvements to biodiversity (Adams et al., 2023; Romanelli et al., 2023; Santangeli et al., 2023).

Due to their highly transformative nature, urban environments struggle to manage biodiversity loss as conditions often favour generalists and disturbance-adapted species (Romanelli et al., 2023). Despite this fact, researchers continue to search for different ways of improving the urban landscape. Though there is some debate over the effectiveness of a stepping stone approach, whereby small habitat patches laid across the landscape help to bolster connectivity of larger fragments, it remains an available option for heavily urbanized areas (Beninde, Veith, & Hochkirch, 2015; Kremer & Merenlender, 2018; Romanelli et al., 2023; Valente et al., 2023). The creation of mini-forests in vacant lots is one trend that is growing in popularity (Romanelli et al., 2023). Backyard gardens are another potential source of connectivity producing both ecological and social benefits (Gaston et al., 2005; Rudd, Vala, & Schaefer, 2002; Standish, Hobbs, & Miller, 2013). To achieve their full ecological potential, more work is needed to improve the design of these smaller patches as they must also contend with issues related size, connectivity, and edge effects (Romanelli et al., 2023).

The areas surrounding RNUP are slowly progressing toward an urban design that is more conscious of its ecological needs. Organizations like Local Enhancement and Appreciation of Forests (LEAF), a non-profit promoting the planting of native trees, has partnered with local municipalities to help subsidize native tree purchases (LEAF, 2022). The Canadian Wildlife Federation (n.d.) advocates for the establishment 'Wildlife Friendly' gardens through their certification scheme, with at least 30 certified gardens around the RNUP area. The Meadoway (2019) project led by the TRCA is creating a pollinator highway beneath the Toronto hydro corridor, taking advantage of unused space that has been dominated by non-native invasive species such as *V. rossicum/nigrum*. RNUP is not this isolated area, acting as a potential oasis in an otherwise barren landscape full of brick and concrete. It is but one area in a variable mosaic offering a range of resources to those able to access them (Dunford & Freemark, 2004). How these various projects work together across the landscape will be key to restoration success. Long-term monitoring will be needed, however, to understand if the purported benefits of these urban programs will be realized and to adapt to changing climatic conditions.

4.0 Conclusion

Measuring urban edge effects in Rouge National Urban Park is the first step in developing a set of best practices for ecological restoration in urban environments. The significance of this research is highlighted by the relative few studies that have examined the edge effects in urban forests that also see regular recreational use. Existing research has tended to focus on rural matrices or undisturbed urban woodlots. Though a measurable distance of edge influence was not found, the groundwork has been laid on which future studies may build. This research has revealed the variability in the kinds of edges present in RNUP, in addition to unmarked sources of fragmentation. Subsequent studies may target specific edge types and design an approach that better captures the environmental changes occurring as one moves into the interior. It is recommended that they expand into a two-sided edge effect study to create a predicative model of the complex drivers behind current edge conditions. Building a deeper level of understanding of edge effects will enable land managers, like Parks Canada, to adapt its ecological restoration plans to a dynamic urban environment. It will also enable it to improve habitat patch connectivity more effectively, not only within the park but across the broader landscape as well.

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6.0 Appendix

6.1 Pilot Study 2021 Data

6.1.1 RStudio Code

```
>install.packages("vegan")
>library(vegan)

#analysis of Fragment 1
>F1data<-read.csv("F1 Data Entire.csv")
>F1a<-F1data[-1]
>F1div<-diversity(F1a,index="shannon")

#Find standard deviation
>F1sd<-sd(F1div)

#create new csv file for diversity
>F1aly<-read.csv("F1 Diversity.csv")

#calculate required sample size for F1
# Testing Ho: mu1 = mu2
#       Ha: mu1 <> mu2
>power.t.test(power=.95, delta=1, sd=.735, sig.level = .05, type =  "two.sample")

#repeat for analysis of Fragment 2
>F2data<-read.csv("F2 Data Entire.csv")
>F2a<-F2data[-1]
>F2div<-diversity(F2a,index="shannon")

#Find standard deviation
>F2sd<-sd(F2div)
#create new csv file for diversity
>F2aly<-read.csv("F2 Diversity.csv")

#calculate required sample size for F2
# Testing Ho: mu1 = mu2
#       Ha: mu1 <> mu2
>power.t.test(power=.95, delta=1, sd=.528, sig.level = .05, type =  "two.sample")
```

6.1.2 Species Diversity

Plot	Species Diversity (H’)
F1T1_0m	1.742863
F1T1_60m	1.543029
F1T1_150m	0
F1T1_300m	0
F1T1_600m	0.165331
F1T2_0m	1.43422
F1T2_60m	1.921336
F1T2_150m	0
F1T2_300m	1.750159
F1T2_600m	0.562335
F1T3_0m	0
F1T3_60m	0
F1T3_150m	0
F1T3_300m	1.089501
F1T3_600m	1.087075
F1T4_0m	0
F1T4_60m	0
F1T4_150m	0
F1T4_315m	0.917792
F1T4_600m	1.475076
F2T1_0m	1.560711
F2T1 _50m	1.256384
F2T1 _125m	1.115291
F2T1 _250m	1.199519
F2T1_500m	0.932152
F2T2_0m	0.207653
F2T2_50m	0.6035619
F2T2_125m	N/A
F2T2_250m	N/A
F2T2_500m	1.2728692
F2T3_0m	0.379535
F2T3_50m	0.1702
F2T3_125m	0.989868
F2T3_250m	1.131393
F2T3_500m	0.274937

6.1.3 List of Species Observed

Common Name	Latin Name	Native	Naturalize	Invasive
Alt-Leaved Dogwood	*Cornus alternifolia*	1	0	0
American Beech	*Fagus grandifolia*	1	0	0
American Germander	*Teucrium canadense*	1	0	0
Ash sp.	*Fraxinus sp.*	1	0	0
Aspen sp.	*Populus sp.*	1	0	0
Autumn Olive	*Elaeagnus umbellata*	0	0	1
Basswood	*Tilia americana*	1	0	0
Bitternut Hickory	*Carya cordiformis*	1	0	0
Black Cherry	*Prunus serotina*	1	0	0
Black Raspberry	*Rubus occidentalis*	1	0	0
Black Walnut	*Juglans nigra*	1	0	0
Blue Stemmed Goldenrod	*Solidago caesia*	1	0	0
Broad Leaved Goldenrod	*Solidago flexicaulis*	1	0	0
Canada Anemone	*Anemonastrum canadense*	1	0	0
Common Buckthorn	*Rhamnus cathartica*		0	1
Common Lilac	*Syringa vulgaris*	0	1	0
Common Soapwort	*Saponaria officinalis*	0	0	1
Dog Strangling Vine	*Vincetoxicum rossicum/nigrum*	0	0	1
Dwarf Raspberry	*Rubus pubescens*	1	0	0
Eastern Hemlock	*Tsuga canadensis*	1	0	0
Eastern White Cedar	*Thuja occidentalis*	1	0	0
Eastern White Pine	*Pinus strobus*	1	0	0
False Solomon Seal	*Maianthemum racemosum*	1	0	0
Fern sp.		1	0	0
Goldenrod sp.	*Solidago sp.*	1	0	0
Green Ash	*Fraxinus pennsylvanica*	1	0	0
Helianthus sp.	*Helianthus sp.*	1	0	0
Honey Suckle (Goldie)	*Lonicera tataric/maackii/morrowii/bella*	0	0	1
Iron Wood	*Ostrya virginiana*	1	0	0
Japanese Barberry	*Berberis thunbergii*	0	0	1
Large Tootherd Aspen	*Populus grandidentata*	1	0	0
Manitoba Maple	*Acer negundo*	1	0	0
Maple Leaf Viburnum	*Viburnum acerifolium*	1	0	0
Mountain Maple	*Acer spicatum*	1	0	0
Northern Oak Fern	*Gymnocarpium dryopteris*	1	0	0
Obedient Plant*	*Physostegia virginiana*	1		
Paper Birch	*Betula papyrifera*	1	0	0
Poison Ivy	*Toxicodendron radicans*	1	0	0

Purple Crown Vetch	*Securigera varia*	0	0	1
Purple Stemmed Angelica	Angelica atropurpurea	1	0	0
Raspberry sp.	Rubus sp.	1	0	0
Red Maple	*Acer rubrum*	1	0	0
Red Oak	*Quercus rubra*	1	0	0
Red Osier Dogwood	*Cornus sericea*	1	0	0
Riverbank Grapevine	*Vitis riparia*	1	0	0
Rough Horsetail	*Equisetum hyemale*	1		
Slippery Elm	*Ulmus rubra*	1		
Staghorn Sumac	*Rhus typhina*	1	0	0
Sugar Maple	*Acer saccharum*	1	0	0
Trembling Aspen	*Populus tremuloides*	1	0	0
Trillium	*Trillium grandiflorum*	1	0	0
Virginia Creeper	*Parthenocissus quinquefolia*	1	0	0
White Ash	*Fraxinus americana*	1	0	0
White Baneberry	*Actaea pachypoda*	1	0	0
White Elm*	*Ulmus americana*	1	0	0
White Snakeroot	*Ageratina altissima*	1	0	0
Wild Sarsaparilla	*Aralia nudicaulis*	1	0	0
Willow sp.	Salicaceae sp.	1	0	0
Witch Hazel	*Hamamelis virginiana*	1	0	0
Yellow Birch	*Betula alleghaniensis*	1	0	0
Total		52	1	7

6.2 Primary Study 2022 Data

6.2.1 RStudio Code

```
setwd("C:/Users/Redmond Naval/OneDrive - University of Waterloo/Thesis/Master Thesis RStudio WD")

#Calculating species diversity with Shannon-Weiner Diversity Index

install.packages("vegan")
library(vegan)
data2022<-read.csv("2022 Sampling Data.csv")
data2022a<-data2022[-1]
Hdiv<-diversity(data2022a,index="shannon")

#create csv for species diversity
df<-data.frame(Div = c(Hdiv))
write.csv(df,"C:\\Users\\Redmond Naval\\OneDrive - University of Waterloo\\Thesis\\Master Thesis
RStudio WD\\2022 Species Diversity.csv",row.names=FALSE)

#add column for distance in Excel
Div2022<-read.csv("2022 Species Diversity.csv")

#RTEI Testing for H'

div0<-read.csv("RTEI-0m.csv")
div50<-read.csv("RTEI-50m.csv")
div125<-read.csv("RTEI-125m.csv")
div250<-read.csv("RTEI-250m.csv")
div500<-read.csv("RTEI-500m.csv")

#calculate mean species diversity
mean_div0<-mean(div0$div_0m)
mean_div50<-mean(div50$div_50m)
mean_div125<-mean(div125$div_125m)
mean_div250<-mean(div250$div_250m)
mean_div500<-mean(div500$div_500m)


#create function for calculating MEI
MEI<-function(pair){
  e<-pair[1]
  r<-pair[2]
  result<-(e-r)/(e+r)
  return(result)
}

#alpha=0.01
#df=n-1=25
#critical value=2.787
```

```
#Calculate Observed MEI at 0m
#MEI=(e-r)/(e+r)
oMEI_0m<-(mean_div0-mean_div500)/(mean_div0+mean_div500)

#RTEI H' at 0m
rDiv_0m<-read.csv("rDiv_0m.csv")
RTEI_0m_results<-numeric(5000)

for(i in 1:5000){
  r0<-sample(rDiv_0m$ï..rDiv_0m,size=2,replace=FALSE)
  result<-MEI(r0)
  RTEI_0m_results[[i]]<-result
}

RTEI_0m_hist<-hist(RTEI_0m_results,xlab="Magnitude of Edge Influence")

#Compare observed H' to randomized distribution
SD_0m<-sd(RTEI_0m_results)
RTEI_0m_mu<-mean(RTEI_0m_results)
tval_0m<-(oMEI_0m-RTEI_0m_mu)/(SD_0m/sqrt(26))

#Calculate Observed MEI at 50m
#MEI=(e-r)/(e+r)
oMEI_50m<-(mean_div50-mean_div500)/(mean_div50+mean_div500)

#RTEI H' at 50m
rDiv_50m<-read.csv("rDiv_50m.csv")
RTEI_50m_results<-numeric(5000)

for(i in 1:5000){
  r50m<-sample(rDiv_50m$rDiv_50m,size=2,replace=FALSE)
  result<-MEI(r50m)
  RTEI_50m_results[[i]]<-result
}

RTEI_50m_hist<-hist(RTEI_50m_results,xlab="Magnitude of Edge Influence")

#Compare observed H' to randomized distribution
SD_50m<-sd(RTEI_50m_results)
RTEI_50m_mu<-mean(RTEI_50m_results)
tval_50m<-(oMEI_50m-RTEI_50m_mu)/(SD_50m/sqrt(26))

#Calculate Observed MEI at 125m
#MEI=(e-r)/(e+r)
oMEI_125m<-(mean_div125-mean_div500)/(mean_div125+mean_div500)

#RTEI H' at 125m
```

```
rDiv_125m<-read.csv("rDiv_125m.csv")
RTEI_125m_results<-numeric(5000)

for(i in 1:5000){
  r125m<-sample(rDiv_125m$rDiv_125m,size=2,replace=FALSE)
  result<-MEI(r125m)
  RTEI_125m_results[[i]]<-result
}

RTEI_125m_hist<-hist(RTEI_125m_results,xlab="Magnitude of Edge Influence")

#Compare observed H' to randomized distribution
SD_125m<-sd(RTEI_125m_results)
RTEI_125m_mu<-mean(RTEI_125m_results)
tval_125m<-(oMEI_125m-RTEI_125m_mu)/(SD_125m/sqrt(26))

#Calculate Observed MEI at 250m
#MEI=(e-r)/(e+r)
oMEI_250m<-(mean_div250-mean_div500)/(mean_div250+mean_div500)

#RTEI H' at 250m
rDiv_250m<-read.csv("rDiv_250m.csv")
RTEI_250m_results<-numeric(5000)

for(i in 1:5000){
  r250m<-sample(rDiv_250m$rDiv_250m,size=2,replace=FALSE)
  result<-MEI(r250m)
  RTEI_250m_results[[i]]<-result
}

RTEI_250m_hist<-hist(RTEI_250m_results,xlab="Magnitude of Edge Influence")

#Compare observed H' to randomized distribution
SD_250m<-sd(RTEI_250m_results)
RTEI_250m_mu<-mean(RTEI_250m_results)
tval_250m<-(oMEI_250m-RTEI_250m_mu)/(SD_250m/sqrt(26))

#RTEI Testing for Species Evenness

eve0<-read.csv("Evenness-0m.csv")
eve50<-read.csv("Evenness-50m.csv")
eve125<-read.csv("Evenness-125m.csv")
eve250<-read.csv("Evenness-250m.csv")

#calculate mean species evenness
mean_eve0<-mean(eve0$eve_0m)
mean_eve50<-mean(eve50$eve_50m)
mean_eve125<-mean(eve125$eve_125m)
```

```
mean_eve250<-mean(eve250$eve_250m)
mean_eve500<-mean(eve0$eve_500m)

#For J' Calculate Observed MEI at 0m
#MEI=(e-r)/(e+r)
oMEve_0m<-(mean_eve0-mean_eve500)/(mean_eve0+mean_eve500)

#RTEI J' at 0m
rEve_0m<-read.csv("rEve_0m.csv")
RTeve_0m_results<-numeric(5000)

for(i in 1:5000){
  rev0m<-sample(rEve_0m$rEve_0m,size=2,replace=FALSE)
  result<-MEI(rev0m)
  RTeve_0m_results[[i]]<-result
}

RTeve_0m_hist<-hist(RTeve_0m_results,xlab="Magnitude of Edge Influence")

#Compare observed H' to randomized distribution
SDeve_0m<-sd(RTeve_0m_results)
RTeve_0m_mu<-mean(RTeve_0m_results)
tvaleve_0m<-(oMEve_0m-RTeve_0m_mu)/(SDeve_0m/sqrt(26))

#For J' Calculate Observed MEI at 50m
#MEI=(e-r)/(e+r)
oMEve_50m<-(mean_eve50-mean_eve500)/(mean_eve50+mean_eve500)

#RTEI J' at 50m
rEve_50m<-read.csv("rEve_50m.csv")
RTeve_50m_results<-numeric(5000)

for(i in 1:5000){
  rev50m<-sample(rEve_50m$rEve_50m,size=2,replace=FALSE)
  result<-MEI(rev50m)
  RTeve_50m_results[[i]]<-result
}

RTeve_50m_hist<-hist(RTeve_50m_results,xlab="Magnitude of Edge Influence")

#Compare observed H' to randomized distribution
SDeve_50m<-sd(RTeve_50m_results)
RTeve_50m_mu<-mean(RTeve_50m_results)
tvaleve_50m<-(oMEve_50m-RTeve_50m_mu)/(SDeve_50m/sqrt(26))

#For J' Calculate Observed MEI at 125m
#MEI=(e-r)/(e+r)
oMEve_125m<-(mean_eve125-mean_eve500)/(mean_eve125+mean_eve500)
```

```
#RTEI J' at 125m
rEve_125m<-read.csv("rEve_125m.csv")
RTeve_125m_results<-numeric(5000)

for(i in 1:5000){
  rev125m<-sample(rEve_125m$rEve_125m,size=2,replace=FALSE)
  result<-MEI(rev125m)
  RTeve_125m_results[[i]]<-result
}

RTeve_125m_hist<-hist(RTeve_125m_results,xlab="Magnitude of Edge Influence")

#Compare observed H' to randomized distribution
SDeve_125m<-sd(RTeve_125m_results)
RTeve_125m_mu<-mean(RTeve_125m_results)
tvaleve_125m<-(oMEve_125m-RTeve_125m_mu)/(SDeve_125m/sqrt(26))

#For J' Calculate Observed MEI at 125m
#MEI=(e-r)/(e+r)
oMEve_250m<-(mean_eve250-mean_eve500)/(mean_eve250+mean_eve500)

#RTEI J' at 250m
rEve_250m<-read.csv("rEve_250m.csv")
RTeve_250m_results<-numeric(5000)

for(i in 1:5000){
  rev250m<-sample(rEve_250m$rEve_250m,size=2,replace=FALSE)
  result<-MEI(rev250m)
  RTeve_250m_results[[i]]<-result
}

RTeve_250m_hist<-hist(RTeve_250m_results,xlab="Magnitude of Edge Influence")

#Compare observed H' to randomized distribution
SDeve_250m<-sd(RTeve_250m_results)
RTeve_250m_mu<-mean(RTeve_250m_results)
tvaleve_250m<-(oMEve_250m-RTeve_250m_mu)/(SDeve_250m/sqrt(26))
```

6.2.2 Species Diversity & Species Evenness

Plot	Species Diversity (H’)	Species Abundance	Species Evenness (J’)
F1T1_0m	1.367702	13	0.533228
F1T1_50m	1.352547	11	0.564056
F1T1_125m	1.761669	10	0.765083
F1T1_250m	1.385221	5	0.860686
F1T1_500m	0.937416	5	0.582449
F1T2_0m	1.91109	11	0.796987
F1T2_50m	1.236053	5	0.768003
F1T2_125m	1.323787	7	0.680292
F1T2_250m	1.692491	7	0.869768
F1T2_500m	0.976888	8	0.469784
F1T3_0m	1.609438	6	0.898244
F1T3_50m	1.706578	12	0.686777
F1T3_125m	1.149111	4	0.828908
F1T3_250m	1.664852	6	0.929172
F1T3_500m	1.299617	10	0.564416
F1T4_0m	0.144815	3	0.131816
F1T4_50m	0.031282	3	0.028474
F1T4_125m	1.35763	7	0.697684
F1T4_250m	1.133393	4	0.81757
F1T4_500m	1.537015	9	0.699526
F1T5_0m	1.07777	9	0.490514
F1T5_50m	0.981856	11	0.409466
F1T5_125m	1.978252	13	0.771263
F1T5_250m	1.852562	10	0.804557
F1T5_500m	1.464816	5	0.910142
F1T6_0m	2.131584	19	0.723935
F1T6_50m	2.029025	13	0.791058
F1T6_125m	0.693147	2	1
F1T6_250m	0.876663	5	0.544701
F1T6_500m	1.791759	5	1.113283
F1T7_0m	2.485332	18	0.859866
F1T7_50m	2.261226	15	0.835002
F1T7_125m	0.547571	5	0.340225
F1T7_250m	1.315544	9	0.59873
F1T7_500m	1.30641	5	0.811718
F1T8_0m	1.409722	9	0.641592
F1T8_50m	0	1	N/A
F1T8_125m	0.848686	3	0.772507
F1T8_250m	1.707341	9	0.777044
F1T8_500m	0.842698	8	0.405252
F1T9_0m	0.96033	8	0.461821
F1T9_50m	0.917993	7	0.471755
F1T9_125m	1.483107	8	0.713223

F1T9_250m	1.325809	7	0.681331
F1T9_500m	1.354506	8	0.65138
F2T1_0m	1.298211	8	0.624308
F2T1_50m	1.638027	13	0.63862
F2T1_125m	0.930435	8	0.447445
F2T1_250m	0.10875	3	0.098988
F2T1_500m	1.971384	12	0.793343
F2T2_0m	2.040391	9	0.928622
F2T2_50m	1.494966	8	0.718927
F2T2_125m	0.562335	2	0.811278
F2T2_250m	0.090876	2	0.131107
F2T2_500m	1.508357	13	0.588065
F2T3_0m	1.016369	9	0.462569
F2T3_50m	2.309319	14	0.875054
F2T3_125m	1.959222	15	0.723481
F2T3_250m	1.642367	9	0.747473
F2T3_500m	2.079885	12	0.837007
F2T4_0m	1.189374	11	0.496008
F2T4_50m	0.168501	4	0.121548
F2T4_125m	0.37677	2	0.543564
F2T4_250m	1.599843	9	0.72812
F2T4_500m	1.098612	3	1

6.2.3 List of Species Observed

Common Name	Latin Name	Native	Naturalized	Invasive
Alternate-leaved Dogwood	*Cornus alternifolia*	1	0	0
American Beech	*Fagus grandifolia*	1	0	0
American Hog Peanut	*Amphicarpaea bracteata*	1	0	0
American Lopseed	*Phryma leptostachya*	1	0	0
American Witch Hazel	*Hamamelis virginiana*	1	0	0
Ash sp.	*Fraxinus sp.*	1	0	0
Basswood	*Tilia americana*	1	0	0
Bittersweet Nightshade	*Solanum dulcamara*	0	1	0
Black Cherry	*Prunus serotina*	1	0	0
Black Walnut	*Juglans nigra*	1	0	0
Bladdernut	*Staphylea trifolia*	1	0	0
Bloodroot	*Sanguinaria canadensis*	1	0	0
Blue Cohosh	*Caulophyllum thalictroides*	1	0	0
Blue-stem Goldenrod	*Solidago caesia*	1	0	0
Broadleaf Enchanter's Nightshade	*Circaea canadensis*	1	0	0
Bulblet Fern	*Cystopteris bulbifera*	1	0	0
Bull Thistle	*Cirsium vulgare*	0	1	0
Burdock	*Arctium minus*	0	0	1
Canada Goldenrod	*Solidago canadensis*	1	0	0
Choke Cherry	*Prunus virginiana*	1	0	0
Christmas Fern	*Polystichum acrostichoides*	1	0	0
Colt's Foot	*Tussilago farfara*	0	0	1
Common Bracken	*Pteridium aquilinum*	1	0	0
Common Buckthorn	*Rhamnus cathartica*	0	0	1
Common Mullein	*Verbascum thapsus*	0	1	0
Dame's Rocket	*Hesperis matronalis*	0	0	1
Deptford Pink	*Dianthus armeria*	0	1	0
Devil's Beggartick	*Bidens frondosa*	1	0	0
Dog Strangling Vine	*Vincetoxicum rossicum/nigrum*	0	0	1
Early Meadow Rue	*Thalictrum dioicum*	1	0	0
Eastern Hemlock	*Tsuga canadensis*	1	0	0
Eastern White Cedar	*Thuja occidentalis*	1	0	0
Eastern White Pine	*Pinus strobus*	1	0	0
European Water Horehound	*Lycopus europaeus*	0	1	0
False Nettle	*Boehmeria cylindrica*	1	0	0
False Solomon Seal	*Maianthemum racemosum*	1	0	0
Fleabane	*Erigeron annuus*	1	0	0
Garlic Mustard	*Alliaria petiolata*	0	0	1
Ground Ivy	*Glechoma hederacea*	0	0	1

Guelder Rose	*Viburnum opulus*	0	0	1
Hairy Sweet Cicely	*Osmorhiza claytonii*	1	0	0
Heath Speedwell	*Veronica officinalis*	0	1	0
Herb Robert	*Geranium robertianum*	1	0	0
Honey Suckle	*Lonicera tataric/maackii/morrowii/bella*	0	0	1
Intermediate Wood Fern	*Dryopteris intermedia*	1	0	0
Ironwood	*Ostrya virginiana*	1	0	0
Jack-in-the-pulpit	*Arisaema triphyllum*	1	0	0
Large-toothed Aspen	*Populus grandidentata*	1	0	0
Manitoba Maple	*Acer negundo*	1	0	0
Mapleleaf Viburnum	*Viburnum acerifolium*	1	0	0
Marginal Wood Fern	*Dryopteris marginalis*	1	0	0
Mayapple	*Podophyllum peltatum*	1	0	0
Meadow Horsetail	*Equisetum pratense*	1	0	0
Nannyberry	*Viburnum lentago*	1	0	0
Nipplewort	*Lapsana communis*	0	1	0
Norway Maple	*Acer platanoides*	0	0	1
Ostrich Fern	*Matteuccia struthiopteris*	1	0	0
Paper Birch	*Betula papyrifera*	1	0	0
Poison Ivy	*Toxicodendron radicans*	1	0	0
Prickly Gooseberry	*Ribes cynosbati*	1	0	0
Purple Flowering Raspberry	*Rubus odoratus*	1	0	0
Queen Anne's Lace	*Daucus carota*	0	1	0
Rattlesnake Root (Tall)	*Nabalus altissimus*	1	0	0
Red Maple	*Acer rubrum*	1	0	0
Red Oak	*Quercus rubra*	1	0	0
Red Raspberry	*Rubus idaeus*	1	0	0
Riverbank Grape	*Vitis riparia*	1	0	0
Robin's Plantain	*Erigeron pulchellus*	1	0	0
Round-leaved Dogwood	*Cornus rugosa*	1	0	0
Sensitive Fern	*Onoclea sensibilis*	1	0	0
Sharp-lobed Hepatica	*Hepatica acutiloba*	1	0	0
Shatsha Daisy	*Leucanthemum × superbum*	0	1	0
Small-flowered Buttercup	*Ranunculus parviflorus*	0	1	0
Smaller Forget-me-not	*Myosotis laxa*	1	0	0
Smooth Serviceberry	*Amelanchier laevis*	1	0	0
Spinulose Wood Fern	*Dryopteris carthusiana*	1	0	0
Spreading Dogbane	*Apocynum androsaemifolium*	1	0	0
Staghorn Sumac	*Rhus typhina*	1	0	0
Stately Maiden Fern	*Thelypteris kunthii*	1	0	0
Stickseed	*Hackelia virginiana*	1	0	0

Stinging Nettle	*Cnidoscolus stimulosus*	1	0	0
Sugar Maple	*Acer saccharum*	1	0	0
Tall Thimbleweed	*Anemone virginiana*	1	0	0
Trembling Aspen	*Populus tremuloides*	1	0	0
Trillium	*Trillium grandiflorum*	1	0	0
Virgina Waterleaf	*Hydrophyllum virginianum*	1	0	0
Virginia Creeper	*Parthenocissus quinquefolia*	1	0	0
White Ash	*Fraxinus americana*	1	0	0
White Baneberry	*Actaea pachypoda*	1	0	0
White Oak	*Quercus alba*	1	0	0
White Snakeroot	*Ageratina altissima*	1	0	0
White Spruce	*Picea glauca*	1	0	0
White Vervain	*Verbena urticifolia*	1	0	0
Wild Bergamot	*Monarda fistulosa*	1	0	0
Wild Geranium	*Geranium maculatum*	1	0	0
Wild Sarsaparilla	*Aralia nudicaulis*	1	0	0
Wood Anenome	*Anemonoides quinquefolia*	1	0	0
Wood Aven	*Geum urbanum*	0	1	0
Wood Nettle	*Laportea canadensis*	1	0	0
Woolly Foxglove	*Digitalis lanata*	0	1	0
Zigzag Goldenrod	*Solidago flexicaulis*	1	0	0
Total		79	12	10