Come from away: Non-native plant establishment within the boreal forest region of Newfoundland, Canada

by © Jennifer Sullivan

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Abstract

The early movement of Western Europeans to the island of Newfoundland has caused it to have one of the longest North American histories of continuous plant species introductions; one measured in centuries. Yet, we know little about the invasiveness of Newfoundland's non-native vascular flora, or of the ability of the island's boreal forests to resist their establishment. Anthropogenic linear disturbances (i.e., recreational trails and forest roads), pervasive throughout North American boreal forests, are particularly vulnerable to the establishment of non-native plant species and may act as corridors of invasion. We examined presence of non-native plants along anthropogenic linear disturbances within boreal forests of Newfoundland, and assessed the resistance of adjacent boreal forests to non-native plant establishment. We found that non-native plants occurred on the majority of observed linear disturbances. Yet, non-native species have not yet established within adjacent forest stands. Additionally, we examined presence of non-native plants on natural linear disturbances, i.e., stream banks, and found that non-native species are pervasive along these corridors. Under ongoing global change, our findings show that the stage is set for non-native plant establishment within disturbed areas of the boreal forest of Newfoundland, as the increased pressures from climate change, globalization, and natural and anthropogenic disturbances continue to influence the distribution and establishment of these species.

Keywords: non-native, invasive, boreal forest, introductions, riparian, disturbance, anthropogenic

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Co-Authorship Statement

All chapters of this thesis were co-authored with Dr. Carissa Brown. As the primary author, I have been the primary researcher of the study for the literature review, designing the research proposal, project planning and logistics, fieldwork, laboratory work, data analysis, and manuscript preparation. All parts of the project were done in collaboration with Carissa Brown who provided significant contributions to project design, data analysis, and manuscript preparations.

Chapter 1: Introduction and thesis overview

1.1 Introduction

1.1.1 Non-native species and global change

The distribution of species outside of their native range is a naturally occurring phenomenon (Vitousek et al., 1997), as species are introduced into new regions via long distance dispersal (e.g., via ocean currents or prevailing winds) or range expansion (Lockwood et al., 2007). However, the rate of non-native species introductions is on a rise globally (Seebens et al., 2015; McGeoch et al., 2010) and this increase is primarily due to human influence (Vitousek et al., 1997; Elton, 1958). The global trade of commodities, which has increased 30-fold since 1950 (Seebens et al., 2015), is the primary cause for these increased introductions (Latombe et al., 2017; van Kleunen et al., 2015; McGeoch et al., 2010; Meyerson & Mooney, 2007). Non-native species introductions and global trade are so interconnected that the number of non-native species invasions can be correlated with the import value of a country (Seebens et al., 2015). The current state of globalization and trade make prevention of non-native species establishment effectively impossible (Vitousek et al., 1997). Human transport of species, whether intentional or unintentional, is much more dynamic than natural methods of transport, and species are being moved much more quickly and at a wider scale than ever before (Lockwood et al, 2007).

Interactions with other features of global change, such as climate change, will stimulate non-native species to spread further (Ricciardi et al., 2017; Caplat et al., 2013).

The increase in temperature at high latitudes is predicted to allow the range expansion of species into regions previously inhospitable (Walther et al., 2009). The interacting effects of globalization and climate change on non-native species introduction and establishment may be profound, but comprehensive analyses are still lacking (Seebens et al., 2015). The projected increase of atmospheric carbon dioxide is expected to stimulate the growth of many plant species, and non-native plants are expected to respond with greater growth rates than native plants (Meyerson & Mooney, 2007). Because of this, there is a strong need to identify non-native species with potentially high environmental impacts for cost effective management (Kumschick et al., 2015).

The majority of non-native plant species may pose little threat to native species or ecosystems (Vitousek et al., 1997). This is explained by the Tens Rule, proposed by Williamson and Fitter (1996), which states that of all non-native species transported into a region, on average, only 10% become established, and of those that become established, only 10% will become invasive. Williamson and Fitter interpret a 10% estimate to actually be anywhere between 5% and 20%. However, there are researchers that argue that the dynamics of non-native species invasion are much more complicated and that the Tens Rule oversimplifies and underemphasizes the detrimental impacts that invasive species can have on natural ecosystems (Jaric & Cvijanovic, 2012). In fact, a study by Jeschke and others (2012) found that the Tens Rule did not fit the majority of studies that they assessed; in many cases they found that the probability that an introduced species would become invasive could be higher than the Tens Rule states, depending on the taxonomic group. There have also been arguments that the use of the Tens Rule may be concerning as it suggests that there is a small probability of an introduced species becoming invasive, which may deter the placement of effective policies and management for the control of these species (Jeschke et al., 2012; Jaric & Cvijanovic, 2012). Therefore, although we are unsure if the Tens Rule fits within non-native species research, we know that the percentage of introduced non-native species that become invasive can cause detrimental and irreversible ecological and economic impacts (Lockwood et al., 2007). What is concerning is the rate of invasion is expected to continue to increase globally, along with associated environmental and social costs (Seebens et al., 2015; McGeoch et al., 2010; Meyerson & Mooney, 2007).

1.1.2 A brief overview of non-native species terminology

Internationally recognized terminology is important in scientific research, as inconsistent use of scientific terms can distract from theoretical issues, cause poor comparisons between studies, as well as inhibit scientific development (Halifors et al., 2014; Colautti & MacIssac, 2004). Scientific terminology in invasion ecology has expanded and varied over time causing uncertainty and misuse regarding certain terms (Richardson et al., 2000). Many terms have been used interchangeably to describe both similar and dissimilar concepts (Colautti & MacIssac, 2004, Davis & Thompson, 2000). For instance, the terms ‘non-native’, ‘non-indigenous’, ‘exotic’, and ‘alien’ have all been used to describe species that have been introduced outside their native range due to human influence. The terms ‘naturalized’ and ‘invasive’ have been used interchangeably when their definitions are distinctly dissimilar (Richardson et al., 2000). The issue with using these two distinctly dissimilar terms interchangeably is that it combines two distinct phases in the invasion process.

Some of the terminology used in invasion ecology research has received attention over the years by individuals that are concerned with the use of xenophobic and militarized language (Comaroff, 2017; Larson, 2005; Simberloff, 2003). Terms such as ‘alien’, ‘exotic’, ‘weeds’, ‘pests’, and even ‘invasion’ have been considered emotive, negative, and unsatisfactory. This has caused introduced species to be commonly viewed as ‘bad’ and native species ‘good’, a preference based on geographic origin rather than individual merit (Slobodkin, 2001; Gould, 1998). In this study, we have chosen currently accepted terminology based on the previous discussions above. We use neutral terms, such as ‘introduced’, wherever possible and provide definitions for other terms that are used for clarification (Table 1.1). We have avoided the terms ‘exotic’ and ‘alien’ due to negative xenophobic connotations that are attached to these terms.

Table 1.1 Invasion ecology terminology

Term	**Definition**
Introduced/Non-native	*Species that were either intentionally or accidentally introduced into a novel environment as a result of human activity (Langor et al., 2014; Halifors et al., 2014; Richardson et al., 2000)*
Casual	*A non-native plant that does not persist for more than a couple years without continuous new introductions (Kloot, 1987)*
Naturalized/established	*Non-native species that have reproduced consistently and maintain a population of many life cycles without direct intervention by humans (Richardson et al., 2000; Kloot, 1987)*
Invasive	*Naturalized non-native or native species that produce large numbers of offspring, have a larger dispersal range, and cause economic or environmental harm to their novel environment (Colautti & MacIssac, 2004; Davis & Thompson, 2000)*

1.1.3 Impacts of non-native species

Invasive species are considered introduced species that have spread and become established in a natural or semi-natural area and that have caused some damaging or destructive impact on the local environment or economics (Simberloff et al., 2012; Kohli, 2009; Lockwood et al., 2007; Meyerson & Mooney, 2007; Williamson, 1996). These species are a significant component of global environmental change (Vitousek et al., 1997) and are considered one of the leading threats to global biodiversity (Simberloff et al., 2013; McGeoch et al., 2010; D'Antonio, 1997). For instance, in Canada non-native vascular plants have been identified as a major factor in the risk status of 45 Canadian species, including plants, birds, amphibians, reptiles, and insects (Canadian Food Inspection Agency, 2011). Although invasive species can be both native or non-native there are generally more than six times as many non-native invading species than native (Simberloff et al., 2012). These invasions are obscuring the distinctiveness of Earth's biota by breaking down the biogeographical barriers that encompass the major floral and faunal regions of Earth (Pysek et al., 2012; Vitousek et al., 1997; Elton, 1958).

Non-native plant invasions, in particular, are considered a primary threat to ecosystem function as well as endangered species (Blossey et al., 2001). Many invasive plants transform ecosystems both above and below ground by altering nitrogen fixing, nutrient cycles, etc. (Simberloff et al., 2013). Plant species invasions are more frequent than both fish and avian species invasions (Vitousek et al., 1997). At least 3.9% of all vascular plants that are currently known on Earth have become naturalized outside of their native ranges as a direct result of human activity; this is the equivalent of about 13,000 naturalized vascular plant species (van Kleunen et al., 2015). For example, *Cirsium arvense* (Canada thistle) alters soil microbial community, reducing the performance of some native plant species as well as aiding its own invasion and the invasion of other species (Verbeek & Kotanen, 2019).

The presence of these species can lead to alterations in nutrient cycling, disturbance regimes, hydrology, energy budgets, as well as altering native species abundance and survival (Latombe et al., 2017; Sanderson et al., 2012; McGeoch et al., 2010; Kohli, 2009; Lockwood et al., 2007; Vitousek et al., 1997). Some species invasions can even degrade human health and wealth directly (Vitousek et al., 1997; Elton, 1958). In Canada, invasive plants, such as *Cirsium arvense* (Canada thistle) and *Leucanthemum vulgare* (oxeye daisy), impact agriculture directly by causing yield losses as well as increasing herbicide use. These invasive plants can also reduce the carrying capacity of rangelands for livestock (CFIA, 2011). *Heracleum mantegazzianum* (Giant hogweed), now prevalent in British Columbia and Ontario, is an escaped garden ornamental with sap that causes severe skin inflammation in ultraviolet light (CFIA, 2011).

Disturbance regimes can be directly altered by the invasion of novel species (Vitousek et al., 1997). Some species of invasive grasses, for instance, are able to establish in a disturbed area and cause changes to the natural landscape by altering disturbance regimes. In western North America invasive grass *Bromus tectorum* (cheatgrass) has caused a severe increase in the fire return interval (D’Antonio & Vitousek, 1992). This species invaded the region, which historically had been comprised of perennial grasses, due to degradation of the landscape by livestock. This species is a winter annual that dies and dries out by spring, causing widespread fires as well as suppressing the growth of native plant species (D’Antonio & Vitousek, 1992).Estimates for shrubland in Idaho state an increase from a 60- to 110-year fire return interval before *B. tectorum* to a 3- to 5-year fire return interval after the introduction of *B. tectorum* (Whisenant, 1990). The litter of this species also enhances the seed germination of several other novel species due to an increase in water availability associated with it. Invasive perennial grasses *Agropyron desertorum* and *Dactylis glomerata* alter ecosystems in the Southwestern United States by suppressing pine reestablishment after wildfires (Elliott & White, 1987).

Non-native plant species can be classified as ‘good’ or ‘bad’ (i.e., invasive) depending on their ecological, economic, and sociological effects. This is heavily dependent on the perspective of the individual or organization doing the labelling (Simberloff et al., 2013). For example, nitrogen fixing can enhance an ecosystem’s nitrogen input, soil fertility, and productivity. This may be seen as a positive but in oligotrophic systems, this would be an issue for native species adapted to a low nitrogen environment (Simberloff et al., 2013). Another example is the invasion of Pinaceae in the southern hemisphere. These species impair decomposition, reduce litter quality, deplete the water table, and deplete many soil species but they are a fast-growing tree that supports the timber industry. In contrast, some non-native species are beneficial to humanity (Vitousek et al., 1997) and can aid in the conservation of certain species (Simberloff et al., 2013). Studies have shown that non-native species can provide food and shelter for native species, as well as serve as substitutes for extinct species and catalysts for the restoration of native species (Schlaepfer et al., 2011). For instance, in Puerto Rico, previously used pastures that have experienced soil erosion and contain sparse vegetation are not readily colonized by native tree species. Non-native plantation trees are able to survive in these areas, attracting seed dispersers and establishing microclimates which allow for native species to re-establish (Lugo, 1997).

It is hard to get a clear image of what causes the success or failure of certain species because of the effects of multiple extrinsic forces (Lockwood et al, 2007). It has been determined that ecosystem community, environmental conditions, and species traits play a role in whether or not a non-native species will successfully establish (Latombe et al., 2017; Kumschick et al., 2015; Lockwood et al, 2007). The impacts that are caused by invasive species vary both in location as well as in duration or frequency because they are under the influence of local abiotic and biotic variables (Kumschick et al., 2015). Anthropogenic disturbance, natural disturbance, environmental conditions, or current vegetation cover can affect non-native species invasion (Meyerson & Mooney, 2007).

Although there has been much debate among scientists regarding the scale of impacts caused by invasive species, as well as the extent to which intervention and management is warranted (Kumschick et al., 2015), there is a need for mitigation of impacts. Mitigation is required not only where non-native species are present, but also where they are expected to invade in the future (Kumschick et al., 2015). The full range of ecological, economic, and sociological consequences should be considered when an invasion impact is evaluated (Simberloff et al., 2013). Local factors (i.e., species interactions, physiochemical conditions, and coinciding stressors) can cause challenges for risk assessment and may misguide management decisions (Kumschick et al., 2015). Since an individual species' invasion impact can vary between locations, these local factors can cause difficultly in determining whether a non-native species will become invasive in a specific area. Sundaram et al. (2015) found that populations of the invasive plant *Lantana camara* are impacted by local propagule pressure and fire frequency in South India. Colonization probability was found to increase with proximity to already established populations, while density of populations was found to be constrained by increased fire frequency. The authors note that other local factors, such as soil, topography, and biotic interactions, were not investigated in their study and may also factor in to *Lantana camara's* colonization.

1.1.4 When do non-native species become invasive?

There are four main factors that influence the success of non-native plant species: competitive release, resource availability, propagule pressure, and disturbance (Kohli, 2009). Competitive release occurs when a species expands its range due to the removal of another species that competed for the same ecological niche (Allaby, 2010). Resource availability is an important factor in species invasions; without sufficient and appropriate resources it is difficult for a species to establish itself. Propagule pressure involves the number of individuals of a species that are introduced to a new system and the number of introduction events that occur (Meyerson & Mooney, 2007; Lockwood et al., 2005); this is increased when the same species is continually introduced to an area (Meyerson & Mooney, 2007). Generally, the larger the propagule pressure the more likely that species will be able to establish itself (Lockwood et al., 2005).

Disturbance can potentially cause shifts in species composition as well as resource availability, opening niche space and allowing the establishment of non-native species (Rendekova et al., 2019; Trip & Wiersma, 2015). It occurs when a natural or anthropogenic influence causes a pronounced change in an ecosystem. Disturbance generally results in a reduction of competitive pressure from other plant species, stimulation of germination, as well as alteration of resource levels (Hierro et al., 2006). Invasions often interact with disturbance regimes directly and human activities have increased the frequency of disturbances as well as created types of disturbances that are unknown in the evolutionary history of many species (Vitousek et al., 1997). Ruderal species, plant species that are adapted to thrive where there is disturbance (Hill et al., 2002), are a common life history type observed among non-native species (Hierro et al., 2006; Hasen and Clevenger, 2005). Since the majority of non-native plant species are agricultural weeds or garden escapees, they are generally adapted to grow in open-canopy habitats (Charbonneau and Fahrig, 2004). Disturbed areas create canopy openings in natural landscapes and, therefore, generally contain a higher proportion of non-native species (Rendekova et al., 2019). Ruderal species tend to have high seed production and high relative growth gate, similar to many established non-native species (Rendekova et al., 2019); although, non-native species may be able to outcompete native ruderal species. In a study by Rendekova et al. (2019) non-native, invasive species were found to have a negative effect on the biodiversity of ruderal plant communities in disturbed areas, at a local scale.

Local and global scale of disturbance are important factors in facilitating species invasions (Meyerson & Mooney, 2007). At the local scale, disturbances to the soil layer have been found to contribute to an increased proportion of non-native species (Von Holle & Motzkin, 2007). Globally, studies on climate change project an increase in atmospheric carbon dioxide (CO_2); this is predicted to increase the growth rate of many plant species and invasive species are expected to have higher growth rates than non-native and native species (Meyerson & Mooney, 2007). Disturbance may operate in association with other proposed mechanisms for non-native plant success, such as enemy release, competitive release, and allelopathy. These can cause non-native plant species to become dominant in foreign regions (Hierro et al., 2006). Positive interactions occur between propagule pressure and disturbance: the more disturbed an area is, the easier it is to invade (Meyerson & Mooney, 2007).

1.1.5 Non-native species and the boreal forest

The boreal forest is an important contributor to global carbon storage and biodiversity, containing about one third of all remaining global forests (Sanderson et al., 2012; Bradshaw et al., 2009). It is an important reservoir of biogenic stored carbon, containing about 32% of global terrestrial carbon, which is comparable to, if not greater than, tropical regions (Gauthier et al., 2015). It is because of this that the fate of these forests should be a global concern.

Non-native plant species, generally adapted for quick growth and propagation, tend to not have the adaptations required for the low light and limited nutrient availability of the boreal forest region (Sanderson et al, 2012). Because of this, the boreal forest is viewed as being an inhospitable, or resistant, environment to non-native species. Yet, its natural resistance does not stop some species from being able to penetrate the forest and establish populations (Sanderson et al, 2012) and non-native plant species have been steadily increasing in the boreal forest, due to a steady increase in human activity (Price et al., 2013; Sanderson et al., 2012; Williamson, 1996).

Arguably, the main factor influencing non-native species' success in the boreal forest is disturbance (Kohli, 2009). In fact, non-native species are more likely to penetrate a region by means of disturbance than any other means (Langor et al., 2014; Rose & Hermanutz, 2004). The boreal forest encompasses a wide array of both natural and anthropogenic disturbances that include wildfire, insect damage, weather events, as well as logging, agriculture, and urban development. Anthropogenic disturbances have increased in recent years due to increased development and economic activity (Gauthier et al., 2015; Langor et al., 2014; Bradshaw et al., 2009). Many of the natural disturbances that occur in the boreal forest are fundamental processes which increase biodiversity, therefore not allowing the ecosystem to become too homogenous, as well as increasing canopy openings in the forest which result in an increase in resource availability (Bonan, 1992). These disturbances, along with added anthropogenic disturbances could open up corridors for invasive species to establish within the forest ecosystem. Because disturbance is a necessary process within the boreal forest, caused naturally by fire and windfalls, and is also a corridor to non-native plant species, we can assume that disturbed areas of the boreal forest will contain a higher proportion of non-native plants (Rose & Hermanutz, 2004). Disturbances cause intermittent peaks of resource availability, which can have long term impacts on outcome of invasion, particularly if fluctuations coincide with availability and arrival of suitable propagules (Kohli, 2009). In this way, disturbance lowers the boreal forest's natural resistance to non-native plant species establishment.

Research tends to be scarce in the field of non-native plant establishment in the boreal forest, compared to some other ecosystems, and the ecological effects of invasion are poorly understood in this system (Sanderson et al., 2012). Non-native plant establishment has been considered to be relatively small in the boreal forest in comparison to other biomes (Sanderson et al., 2012), but a major concern in regards to this topic is climate change. Recent studies have found that, in Alaska, the number of non-native plants has increased by 46% between 1941 and 2006. This is partly due to climate change increasing the vulnerability of the boreal forest to non-native species introduction (Spellman et al., 2014). Climate change increases the rate of non-native species establishment in the boreal forest in direct (temperature and growing season) and indirect (altering disturbance regimes) ways. For example, the increase of wildfires in

Alaska due to climate change is expected to cause the boreal forest's natural resistance to invasion to decrease when the organic layer is removed, therefore, facilitating non-native plant establishment (Walker et al., 2017; Spellman et al., 2014). The warming temperatures observed across the Northern Hemisphere continue to rise, resulting in alleviated climate barriers to species that would not normally have been able to expand their range or grow their population into the boreal forest (Gauthier et al., 2015).

1.1.6 Linear disturbances in forest ecosystems

Natural disturbance regimes can be categorized as having different levels of intensity (i.e., the amount of energy released by the physical process of disturbance) and severity (i.e., the amount of plant mortality that occurs due to a disturbance); however, a shared characteristic between these regimes is their rotation between disturbance and regrowth of plant species (Frelich et al., 2002). Severity and intensity are generally correlated, but that correlation varies based on the disturbance mechanism. For instance, windstorms have a highly correlated intensity versus severity; higher velocity winds generally result in more downed trees. However, herbivory has a low intensity but severity can be very high, resulting in large-scale mortality (Frelich et al., 2002). Anthropogenic linear types of disturbance, i.e., trails and roadways, are considered low intensity, chronic disturbances. These anthropogenic corridors fragment forest ecosystems, and are one of the most profound causes of non-native species dispersal, possibly accelerating non-native plant species introductions and establishment (Arevalo et al., 2010; Cole, 2004). The chronic aspect of these disturbances results from constant maintenance of trails and roads for human use, therefore constraining the ability for these areas to transition back to a forested landscape, as would occur within natural disturbance regimes. Consistent movement along these disturbances via foot traffic and vehicles creates a constant influx of propagule pressure (Allen et al., 2009). Not surprisingly, it follows that high occurrences of non-native species have been found along roadways (Spellman et al., 2014; Villano & Mulder, 2008). Once non-native plants are established on anthropogenic corridors, they may spread further along the corridors due to human traffic, wind, animals, and water (Arevalo et al., 2005). Propagule pressure, i.e., the number of individuals released into an area (Johnston et al., 2009), represents an important factor that may influence the success of species invasions. Roadways are considered to represent a reservoir of propagules of non-native species that might colonize adjacent natural areas where ecological requirements are met (Arevalo et al. 2005).

Natural linear types of disturbances, i.e., riparian areas and wildlife trails, are also considered low intensity, chronic disturbances. Although there is less human traffic along these disturbances, propagules can still be introduced via anthropogenic disturbances, animals, and abiotic conditions (i.e., wind or stream currents). Non-native propagules that are introduced along these disturbances may become established and persistent. Riparian areas in particular tend to be vulnerable to non-native species establishment (Stohlgren et al., 1999; Pysek & Prach, 1993). In fact, they have been found to contain a greater species richness and abundance of non-native species relative to non-riparian areas (Brown & Peet, 2003). The natural disturbance regime of riparian systems may allow the introduction and establishment of non-native plant species within these areas.

Anthropogenic linear disturbances that are connected to riparian areas can act as corridors for the dispersal and establishment of non-native plant species within these natural areas (Catford & Jansson, 2014).

1.1.7 Non-native species establishment in Newfoundland's boreal forests

The island of Newfoundland (herein: Newfoundland) is an ideal area to observe the process of non-native plant species establishment in boreal forests. It is an island that is covered by expansive stands of boreal forest, which are owned by the provincial government and leased to various industrial corporations, the largest of which belong to forestry operators. This relationship, in conjunction with a long history of human trade and settlement, has been critical in forming the island's unique flora. Islands have proven to be more vulnerable to non-native species establishment than continents (Arteaga et al., 2009; Lonsdale, 1999; Simberloff, 1995; Wilson et al., 1992). Unfilled niche space that is present on islands, due to decreased area and increased isolation in comparison to the mainland (MacArthur and Wilson, 1967), contributes to this vulnerability, allowing damaging species to become established more readily (van Kleunen et al., 2015; Langor et al., 2014; Vitousek et al., 1997). In fact, on islands the majority of extinctions are due to species invasions (Vitousek et al, 1997). On the island of Newfoundland, 28.4 % of vascular plant species are not native to Canada (Susan Meades, personal communication, Feb 8, 2020), this is slightly higher than the average for Canada, which is 24.2% of vascular plant species (CFIA, 2011). Newfoundland is a critical point of entry for non-native species, as it is one of the few areas where the boreal forest meets a capital city port that is used for global transport of commodities (Langor et al., 2014). The island has a long history of non-native plant introductions that coincides with the arrival of Western Europeans who travelled to the island for fishing purposes several centuries ago, and soon after settled on the island and adapted the landscape (Cooper, 1981).

In Newfoundland, there are a number of, primarily coastal, communities that are considered to be hot spots for non-native species introduction and establishment. These communities are connected by a matrix of transportation routes which tend to spread throughout neighbouring forests, fragmenting these natural areas. These pathways between towns could act as corridors for non-native plant invasion into disturbed boreal forest areas. Increases in non-native species, particularly invasive ones, may result in degradation of native flora and fauna communities that are located within the boreal forest region by outcompeting native plant species and altering nutrient cycles (Sanderson et al., 2012; Kohli, 2009). Natural protected areas that employ research and management techniques could assist in the mitigation, and possibly the circumvention, of non-native species establishment within natural areas. However, these areas are sparse in Newfoundland, which has one of the lowest percentages of protected landscape and freshwater (6.7%) in Canada (WERAC, 2020). This is far from Canada’s commitment to protect 17% of land and freshwater by 2020 (Parks Canada, 2016).

Negative impacts of non-native species have been observed on balsam fir regeneration in the boreal forest of Newfoundland. *Cirsium arvense* (Canada thistle) negatively affects the emergence and early survival of balsam fir (Humber & Hermanutz, 2011), while non-native mammals negatively affect many of the species’ life stages. Balsam fir’s female cones are heavily predated on by non-native *Tamiasciurus* *hudsonicus* (red squirrel), their seeds and seedlings are predated upon by non-native rodents and slugs, and the taller saplings experience heavy browsing by non-native *Alces alces* (moose) (Gosse et al., 2011). These interactions have effects on the long-term removal of understory balsam fir by shifting the understory from feathermoss to competitive grass species and non-native plants (Humber & Hermanutz, 2011). Within Gros Morne National Park, Newfoundland, Rose and Hermanutz (2004) found that boreal ecosystems are also susceptible to non-native plant invasion through vectors of disturbance. *Tussilago farfara* (coltsfoot) is one of the non-native species present in Gros Morne National Park and is primarily found in areas of disturbance which have altered abiotic elements, such as increased soil pH (Hendrickson et al., 2005). Non-native invertebrates have also been observed on the island, from the northern house mosquito which carries West Nile virus (Chaulk et al., 2016), to non-native slugs which can be damaging to ecosystems (Moss & Hermanutz, 2010).

Whether non-native species are passengers or drivers of change is difficult to resolve by observation alone (Kumschick et al., 2015). Research is important to understanding why some species are more damaging than others, some ecosystems are more susceptible than others, and also to determine how limited management resources should be allocated (Kumschick et al., 2015). Policy and cooperation across ecological and political borders are essential to slowing the rate of non-native species introductions (Latombe et al., 2017). As an island, Newfoundland's primary ecological barrier involves the ocean surrounding it. Currently, non-native plant propagules are able to be transported into Newfoundland via human and product transportation, as there are few strategies for management and mitigation of these species on the island. Governments, landowners and managers, as well as the general public could benefit from outcomes of studies to create preventative measures (Kumschick et al., 2015).

We have some understanding of non-native species' distributions and impacts on the island of Newfoundland, but further research is needed. Currently the number of non-native plant species in the boreal zone is low relative to other regions of Canada; 303 of the 1229 non-native species observed in Canada are found within this zone (Langor et al., 2014). This number is even lower in the undisturbed forested area within Canada's boreal zone (Langor et al., 2014). This provides a unique opportunity to prevent the spread of non-native plant species in these areas. With this study, we plan to gain insight on the conditions which may allow non-native species to establish within natural boreal forest areas of Newfoundland and gain an understanding of the role that anthropogenic linear disturbances play in this establishment.

1.1.8 Provincial, federal, and international strategies for dealing with invasive species

Internationally, there are two agreements that include Canada as a member. Firstly, the *International Plant Protection Convention* (IPPC) includes 166 different signing parties and is focused on general plant health. This agreement was signed by Canada in 1951 and 1953 (CFIA, 2011). The IPPC focuses on the 'pests' of plants and plant products, preventing the introduction and spread these 'pests' with appropriate measures of control. In this agreement, 'pests' are defined as any organism (species, strain, or biotype) that is damaging to plants and plant products. Since this is applied to wildlife flora, it includes any plant species that are being negatively impacted by a non-native species (CFIA, 2011). However, ‘pests’ are predominantly defined by their economic importance. Secondly, the *Convention on Biological Diversity* (CBD) includes 190 different signing parties and is focused on promoting biological diversity as well as sustainable use of natural resources (CFIA, 2011). Canada signed an agreement with the CBD in 1992. This agreement promotes preventing significant reduction or loss of biological diversity and mentions non-native species specifically; it states that each party will prevent the introduction, or control the spread, of introduced species that threaten native species or ecosystems (Langor et al., 2014; CFIA, 2011).

At the Federal level in Canada, a number of legislations have been implemented that may provide some non-native species strategies. The National Invasive Species Strategy for Canada involves four goals, which include 1) prevention of the introduction of invasive species, 2) early detection of new invaders upon entry into the country, 3) rapid response to invasive species after detection, and 4) management of invasive species that are established and spreading (Langor et al., 2014). Prevention is agreed to be the most effective and least expensive strategy for dealing with invasive plant species. Although the majority of these strategies are not implemented for the boreal region specifically, they are directed towards all ecoregions in Canada (Langor et al., 2014). The *Wild Animal and Plant Protection and Regulation of International and Inter-provincial Trade Act and Regulations* (S.C. 1992, C.52, WAPPRIITA) targets the movement of invasive plant species, preventing these species from being imported into Canada, as well as being transported between provinces. The *Canadian National Parks Act* (S.C. 2000, c.32) provides direction in the management of national parks. Invasive plants management may be deemed necessary if these species are threatening the ecological integrity of a park (CFIA, 2011). The *Species at Risk Act* (S.C. 2002, c.29) was created to prevent Canadian species from becoming extirpated or extinct as well as to aid with the recovery of threatened or endangered species. If an invasive species threatens the survival of a listed wildlife species, directly or indirectly, a recovery strategy must be prepared to address these threats (CFIA, 2011).

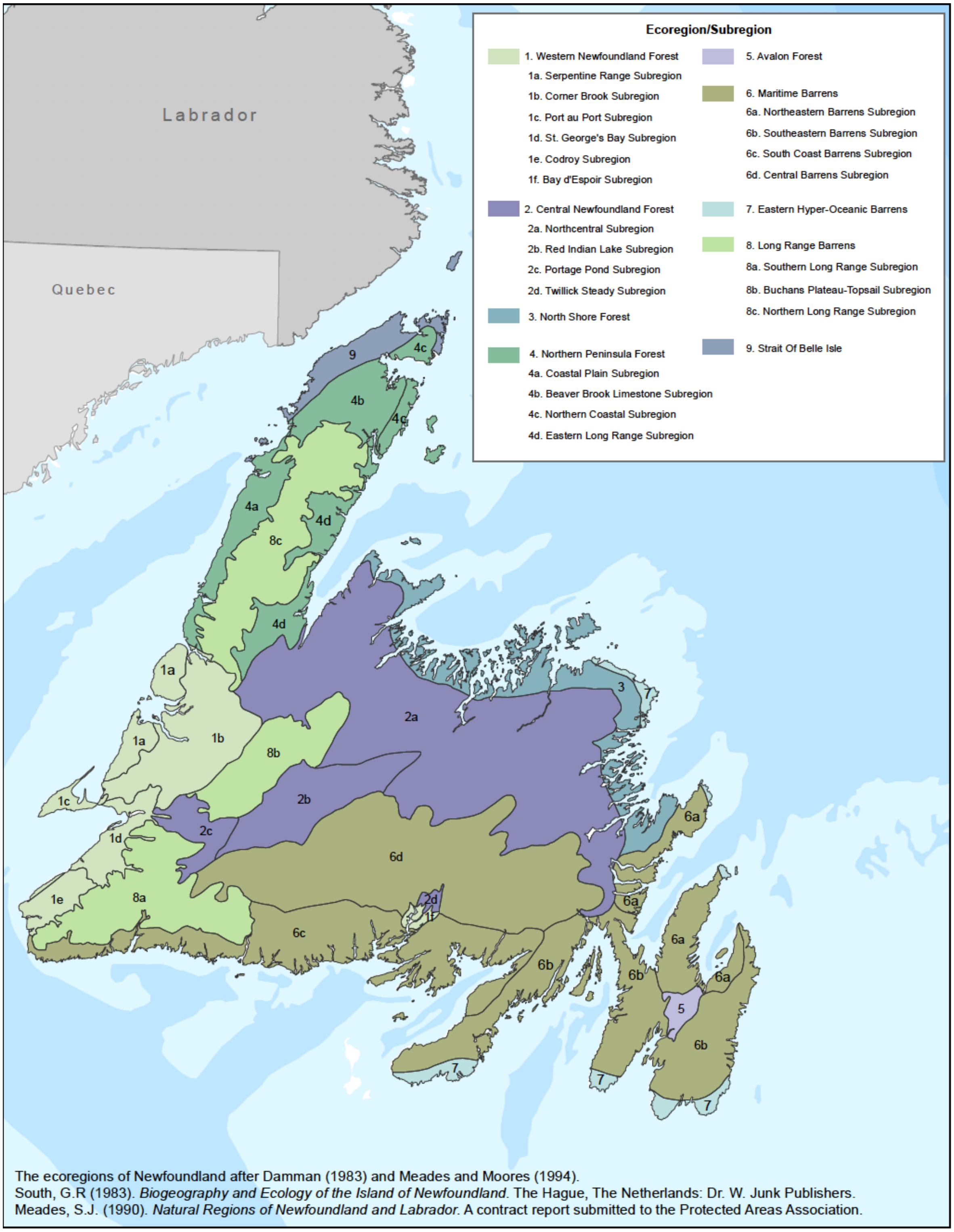
Provincially, Newfoundland and Labrador has little in the form of invasive plant management strategies. The provincial *Plant Protection Act* (S.N. 1978, c.49) is focused only on 'pests' of agricultural plants, or products/by-products of these plants and regulates the movement of these 'pests' into or within the province. Therefore, any non-native plant that is damaging outside of these realms would not by subject to this Act (Lewis, 2006). Regulations of this Act are administered by provincial agricultural departments and are not enforced or administered by the local government.

1.2 Study site

This study took place throughout the Avalon Peninsula of the island of Newfoundland, Canada, within boreal forests containing linear disturbances. Newfoundland's varied climate and geology underlie regional differences in landscape structure and vegetation (Damman, 1983). To remain consistent in predicted species composition within forested plots as well as local climatic conditions, we included only the North- and Southeastern Maritime Barren Ecoregion (MBE) as well as the Avalon Forest Ecoregion (AFE) in this study (Figure 1.1). These areas are characterized by a strong maritime influence, causing large amounts of precipitation (1200-1700 mm annually), mild winters with intermittent snow cover, and cool summers (Banfield, 1983). The AFE is more sheltered and has higher fog frequency than the surrounding MBE, but otherwise have many similarities in climatic conditions.

The Avalon Forest Ecoregion is a relatively small area (~500 km^2) occupying the sheltered interior of the Avalon Peninsula. Despite its small size, the region contains a vegetation community that is different enough from the surrounding area to be separated into its own ecoregion (Damman, 1983). It is the most productive forest on the peninsula, and contains most of the commercially forested areas on the Avalon; in fact, 45% of the forest in this ecoregion has been harvested (Arsenault et al., 2016). Forested areas are dominated by *Abies balsamea* (L.) Mill, *Picea mariana* (Mill.) B.S.P., and *Betula papyrifera* (Marshall) while the understory contains Bryophyte species as well as some vascular plants (*Cornus canadensis* L., *Vaccinium angustifolium* (Aiton), *Kalmia angustifolia* L., etc.). The region is scattered with a large number of small lakes and bogs. The Maritime Barren Ecoregion (~37,000 km^2) is composed of extensive barren areas and forested valleys. The dominant tree species in forested areas of the MBE is *Abies balsamea*, which have stunted growth in stands that are very dense. Common shrub species throughout this region include *Viburnum nudum* var. *cassinoides* and *Ilex mucronata*. The climate of this area is influenced by its extensive barrens which cause high wind velocity (Damman, 1983).

The climate of Newfoundland is changing, and is projected to continue to change throughout the next 20 to 50 years, due to the impacts of climate change. Mean daily temperatures are projected to increase throughout the island. The Avalon Peninsula may experience fewer snow storms that are higher in intensity, as well as more frequent and heavier rainfall during the winter months, due to its mean temperature being close to zero degrees (Finnis & Daraio, 2018). There is projected to be an increase in growing degree days (GDD), the measure of available energy for plant growth during the growing season, across the island (Finnis & Daraio, 2018; Finnis, 2013). This suggests that there will be an increase in plant growth as well as earlier plant maturation on the island. Increases are expected to be substantial in the summer as well as slightly lower, but still significant, in the fall (Finnis & Daraio, 2018; Finnis, 2013). Increases of ~200 GDD are projected in the summer and ~100 GDD in the fall are projected in the vicinity of St. John’s.



The ecoregions of Newfoundland after Damman (1983) and Meades and Moores (1994).
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Figure 1.1 Ecoregion map of Newfoundland, Canada. (source: Fisheries and Land Resources, Government of Newfoundland and Labrador)

1.3 Thesis objectives

Research questions

1. Are anthropogenic linear disturbances associated with the establishment of non-native plant species within adjacent boreal forest stands?

   We aimed to assess the effects of i) distance from linear disturbance (i.e., walking trails, all-terrain vehicle trails, and forest roads), ii) magnitude (width) of linear disturbance, and iii) stand characteristics (e.g., density, composition, soil attributes) on non-native plant establishment.

   <u>*Predictions:*</u> We predicted that non-native plant species richness would be greatest i) in the stands adjacent to linear disturbances that have a higher magnitude of disturbance (width measurement), ii) in plots closest to the linear disturbance, and iii) in forest sites with lower stand density and higher canopy openness (i.e., greater light availability).

2. Do anthropogenic linear disturbances affect the establishment of non-native plants along the natural linear disturbances (i.e., streams) that they intersect, within forested areas of the boreal zone?

   We aimed to assess the effects of i) distance from linear disturbance and ii) area of establishment along stream bank on non-native plant establishment.

   <u>*Predictions:*</u> We predicted that non-native plant species richness would be greatest in stream bank areas that are closest to the linear disturbance and that non-native

species richness would decline as the distance from the anthropogenic linear disturbance increased. Non-native species richness would decline as the stream bank transitioned into the adjacent natural forest ecosystem.

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Chapter 2: Come from away – Non-native plant establishment within the boreal forest region of Newfoundland

2.1 Introduction

The rate of non-native species introductions is on the rise globally, primarily due to the global trade of commodities (Latombe et al., 2017; van Kleunen et al., 2015; McGeoch et al., 2010; Meyerson & Mooney, 2007). The human transport of species, whether intentional or unintentional, is much more dynamic than naturally occurring methods of dispersal; therefore, these species are being moved much more quickly and at a wider geological scale than ever before (Lockwood et al, 2007).

Although some non-native species have few impacts on local ecosystems (Vitousek et al., 1997), certain species can become invasive, i.e., causing damage to natural ecosystems and/or local trade and economy. Invasive species can change how an ecosystem functions by altering nutrient cycling, fire regimes, hydrology, and directly competing with native species (Latombe et al., 2017; Sanderson et al., 2012; McGeoch et al., 2010; Kohli, 2009; Lockwood et al., 2007; Vitousek et al., 1997). Such species are a significant component of global environmental change (Vitousek et al., 1997) and are considered one of the leading threats to global biodiversity (Simberloff et al., 2013; McGeoch et al., 2010; D'Antonio, 1997). Invasive species have caused more extinctions than the effects of human-induced climate change, and are the second leading cause of extinction after habitat loss (Kohli, 2009), although some researchers challenge this (see: Davis et al., 2011; Sax et al., 2007; Gurevitch and Padilla, 2004; Sax et al., 2002). It is hard to get a clear image of what causes success or failure in the establishment of certain non-native species because of the effects of multiple extrinsic forces (Lockwood et al., 2007). Ecosystem community, environmental conditions, and species traits play a role in whether or not a non-native species will be successful in introduced ecosystems (Latombe et al., 2017; Kumschick et al., 2015; Lockwood et al., 2007). In order to mitigate ongoing impacts of these species as well as prevent future invasions, a better understanding of non-native species presence and response in various natural environments is needed.

The island of Newfoundland (herein referred to as Newfoundland) is an ideal area to observe non-native plant species establishment in boreal forest ecosystems, as it is an oceanic island that is covered with a large expanse of boreal forest, although much has been harvested. Islands are more vulnerable to non-native species establishment than continents due to potentially unfilled niche space that usually occurs, and a lower number of individual species that inhabit it (van Kleunen et al., 2015; Langor et al., 2014; Vitousek et al., 1997). In fact, on islands the majority of extinctions are due to species invasions (Vitousek et al., 1997). In Newfoundland, 28.4% of vascular plant species are not native to the island (Susan Meades, personal communication, Feb. 8, 2020), compared to the 24.2% of vascular species throughout Canada (CFIA, 2011). This creates a unique mixture of primarily Eurasian and native species. This has been largely influenced by the island's long history with Europe, whose inhabitants traveled to Newfoundland in the early 16th century for fishing and whaling, and settled on the island shortly after (Cooper, 1981). Newfoundland is also a critical point of entry for non-native species as it is one of the few areas where the boreal forest meets a capital city port that is used for global transport of commodities (Langor et al., 2014). The island has many non-native plant species due to the island's colonial history, heavy ferry traffic, and travel tourism, as well as relatively low non-native species research (but see Trip & Wiersma, 2015; Humber & Hermanutz, 2011; Hendrickson et al., 2005; Rose & Hermanutz 2004).

Non-native plants are generally agricultural weeds or garden escapees that are considered ruderal species, i.e., species that are adapted to grow in disturbed, open canopy, areas (Hierro et al., 2006; Hasen & Clevenger, 2005). Ruderal species tend to be fast-growing, quick propagators and, therefore, may not be well-adapted to the low light, nutrient availability, and low pH of the boreal forest region (Sanderson et al, 2012). Because of this, the boreal forest is viewed as being an inhospitable, or resistant, environment to non-native plant species; but, its natural resistance does not prevent some species from establishing a population within the forest (Sanderson et al, 2012). Arguably, the main factor influencing invasive species' success in the boreal forest is disturbance (Kohli, 2009). In fact, non-native species are more likely to penetrate a region by means of disturbance than any other means (Langor et al., 2014; Hendrickson et al., 2005; Rose & Hermanutz, 2004; Williamson, 1996). Disturbance is an important process within the boreal forest, it is crucial for forest renewal and maintenance of forest heterogeneity and is caused naturally by fire, insects, and windfalls (Bonan, 1992). Since disturbance is also a conduit for non-native plant species introduction and establishment into natural areas, we can predict that disturbed areas of the boreal forest may contain a higher proportion of non-native plants (Rose & Hermanutz, 2004). This occurs because these disturbances lower the boreal forest's natural resistance to non-native species establishment by causing intermittent bouts of resource availability, such as exposed soil beds and increased canopy openings, as well as creating empty niche space (Rendekova et al., 2019). Increases in anthropogenic disturbances, i.e., logging, roadways, oil extraction, is a concern as it increases opportunities for non-native species introductions into these areas. Ongoing climate changes may influence the vulnerability of the boreal forest to non-native plant establishment (Spellman et al., 2014). The direct effects of warmer temperatures and a longer growing season might facilitate non-native species survival. Indirectly, climate change can alter disturbance regimes, increasing areas that are more easily invaded by these species (Spellman et al., 2014).

Anthropogenic linear types of disturbance, i.e., trails and roads, are considered low intensity, chronic disturbances. The chronic quality of these disturbances results from constant maintenance of trails and roads for human use, therefore constraining the ability for these areas to transition into a more natural landscape. Consistent movement along these disturbances via foot traffic and vehicles create a constant influx of propagule pressure (Allen et al., 2009). Previous studies have found high occurrences of non-native species along roadways (Spellman et al., 2014; Villano & Mulder, 2008). Propagule pressure, i.e., the number of individuals released into an area to which they are not native (Johnston et al., 2009), represents an important factor that may influence the success of species invasions.

Natural linear types of disturbances, i.e., riparian areas and moose trails, are also considered low intensity, chronic disturbances. For instance, natural stream banks experience seasonal freezing and flooding disturbance annually, exposing the soil bed of bank zones. Although there is less human traffic along these disturbances, propagules can still be introduced via anthropogenic disturbances, animals, and abiotic conditions (i.e., wind or stream currents). Non-native propagules that are introduced along these disturbances may become established and persistent. Riparian areas in particular tend to be vulnerable to non-native species establishment (Stohlgren et al, 1999; Pysek & Prach, 1993); in fact, they have been found to contain a greater species richness and abundance of non-native species relative to non-riparian areas (Brown & Peet, 2003). Anthropogenic linear disturbances that are connected to riparian areas, i.e., roadways and logging areas, can act as corridors for the dispersal and establishment of non-native plant species within these natural areas (Catford & Jansson, 2014).

Here, we assess whether anthropogenic linear disturbances affect the establishment of non-native plant species i) along natural linear disturbances; and ii) in adjacent forest stands within the Avalon Peninsula region of Newfoundland. Canada is currently experiencing issues with invasive non-native plant species that are spreading quickly across the country and disrupting natural ecosystems, i.e., *Cirsium arvense* (Canada thistle), *Euphorbia esula* (leafy spurge), and *Lythrium salicaria* (purple loosestrife) (Lewis, 2006). Since damage from invasive species has already occurred in natural areas of Canada and is ongoing, it is important to further research on non-native species in proximity to the boreal forest, so as to prevent or mitigate any damage that may occur from future invasions.

2.2 Methods

Our study was conducted on the Avalon Peninsula of the island of Newfoundland, Canada, within boreal forests containing anthropogenic linear disturbances, such as walking trails and forest roads (Figure 2.1). Newfoundland’s varied climate and geology underlie regional differences in landscape structure and vegetation (Damman, 1983). To remain consistent in predicted species composition within forested plots as well as local climatic conditions, we included only the North- and Southeastern Maritime Barren Ecoregion (MBE) as well as the Avalon Forest Ecoregion (AFE) in this study. These areas are characterized by a strong maritime influence, causing large amounts of precipitation (1200-1700 mm annually), mild winters with intermittent snow cover, and cool summers (Banfield, 1983). The AFE is more sheltered and has higher fog frequency than the surrounding MBE, but otherwise have many similarities in climatic conditions.

The Avalon Forest Ecoregion is a relatively small area (500 km^2) occupying the sheltered interior of the Avalon Peninsula. Despite its small size, the region contains a vegetation community that is different enough from the surrounding area to be separated into its own ecoregion (Damman, 1983). It is the most productive forest on the peninsula, and contains most of the commercially forested areas on the Avalon. Forested areas are dominated by *Abies balsamea* (L.) Mill, *Picea mariana* (Mill.) B.S.P., and *Betula papyrifera* (Marshall) while the understory contains Bryophyte species as well as some vascular plants (*Cornus canadensis* L., *Vaccinium angustifolium* (Aiton), *Kalmia angustifolia* L., etc.). The Maritime Barren Ecoregion is much larger in size than the AFE, it includes the majority of the island’s eastern peninsulas, as well as the central barrens and the narrow coastal area that extends west towards Port aux Basques, Newfoundland. It is composed of extensive barren areas and forested valleys. Forested areas of the MBE, dominated by dense growth of *Abies balsamea*, as well as *Viburnum nudum* var. *cassinoides* and *Ilex mucronata*. Although sampling was conducted in both ecoregions, there were no detectable differences in total native and non-native species richness, therefore ecoregion was ignored in subsequent analyses (Appendix II, Table AII).

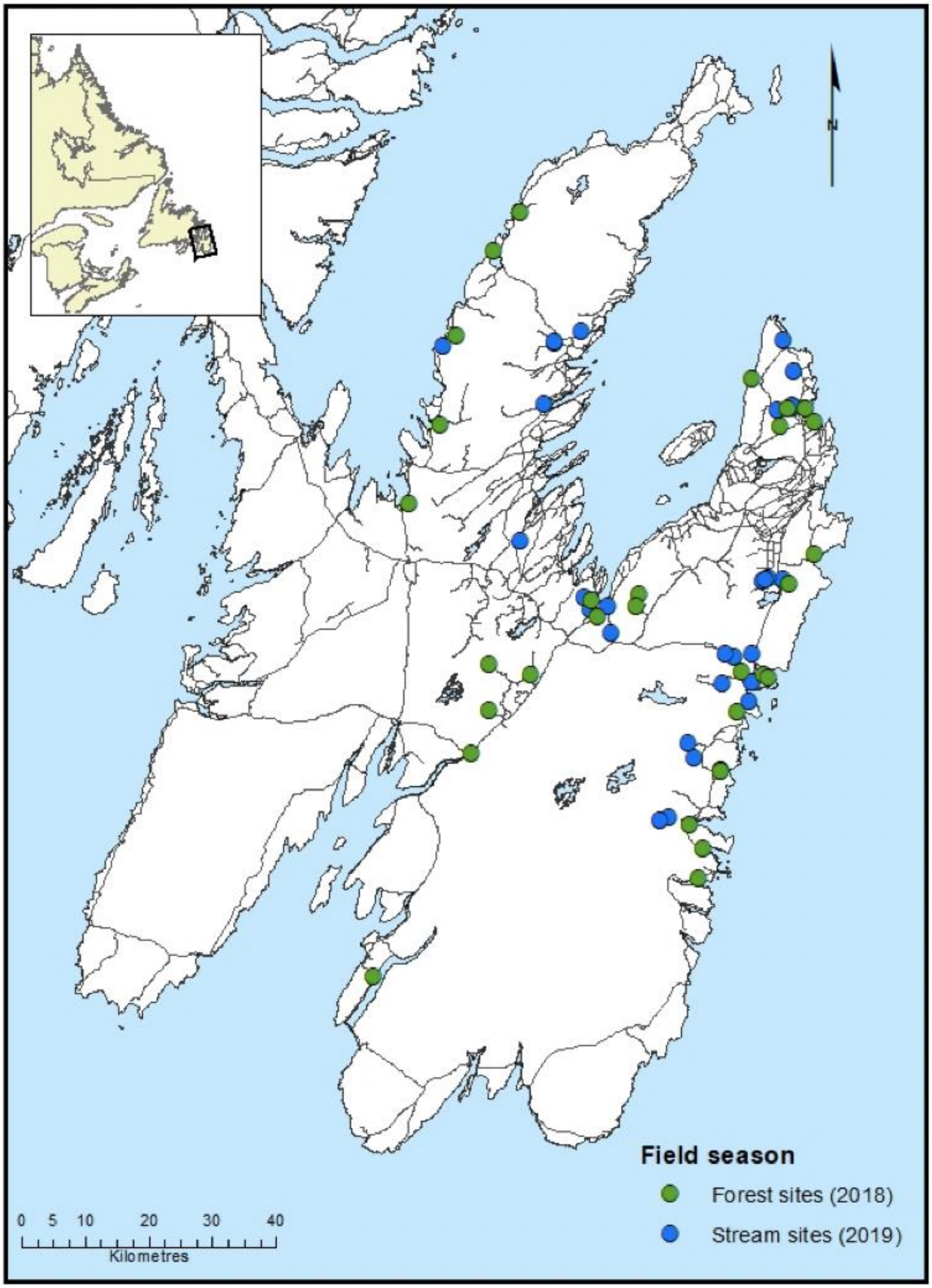

Figure 2.1 Forest and stream study site locations, Avalon Peninsula, Newfoundland and Labrador, Canada

2.2.1 Forest survey

Site Selection

For the purposes of this study we assessed three different types of anthropogenic linear disturbance: walking trails, all-terrain-vehicle (ATV) trails, and forest roads, i.e., unpaved, gravel roads that are within natural forested areas (Figure 2.2). These three types were chosen to assess whether size and usage of a disturbance influenced non-native species present along the disturbance as well as within adjacent forest stands. We first identified study sites based on two criteria: 1) sites must be located within 5 km of a community, and 2) linear disturbances must be adjacent to intact forest (at least 100 m^2). Sites were first located via Google Earth, and then assessed by groundtruthing to determine if the criteria were met as well as the disturbance type classification, as follows. Disturbance types were categorized based on width and access; walking trails were generally categorized as such by local organizations (East Coast Trail Association) and were limited to lower magnitudes of disturbance, i.e., foot traffic. ATV trails and forest roads were differentiated by the trail pattern and condition; generally, if the path lead to specific location, was in decent travelling condition, and was wide enough to allow transport of a vehicle, it was categorized as a forest road. Forest roads were considered to be our highest magnitude of disturbance due to repetitive maintenance of the road as well as use by pedestrians and vehicles regularly. If the path was not able to be traversed by a car and was winding in nature it was considered an ATV trail. The average width of all walking trail, ATV trail, and forest road sites ranged from 0.43m-1.64m, 1.4m-2.26m, and 2.22m-7.02m, respectively. Due to the location of the walking trail system within the study area being primarily coastal (most trails were part of the East

Coast Trail Association network), there may be a more extreme coastal influence, altering climate conditions in comparison to interior areas of the Avalon Peninsula.

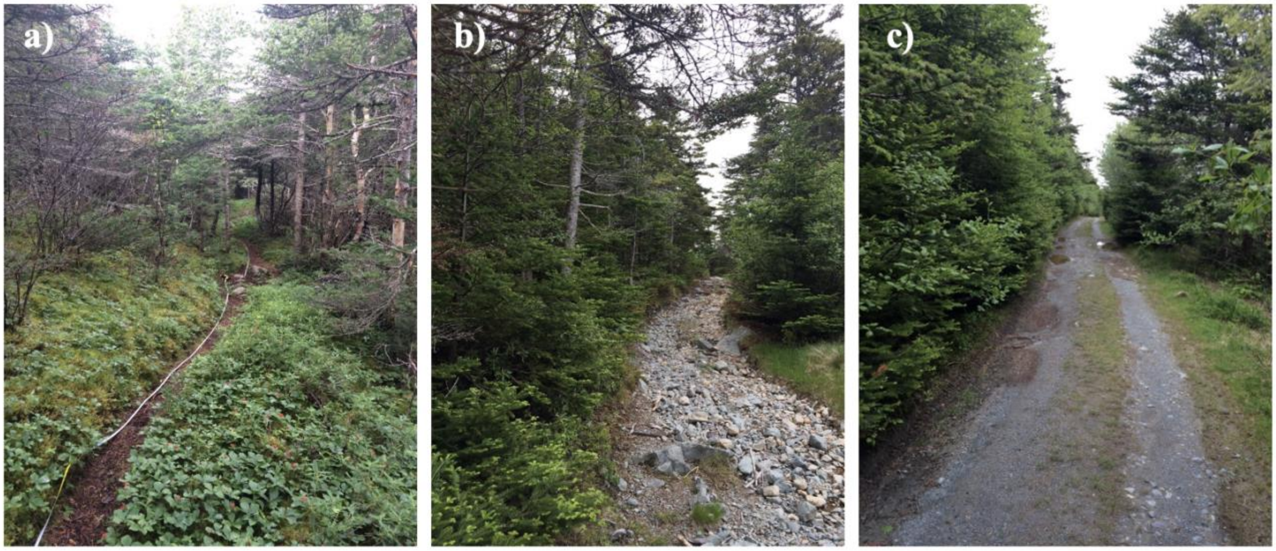

Figure 2.1 Linear disturbances selected during the 2018 field season were classified as either a) walking trails, b) ATV trails, c) forest roads, based on accessibility of the trail to either foot, ATV, or vehicular traffic. Avalon Peninsula, Newfoundland and Labrador, Canada.

Sampling design

At each of 30 sites located around the Avalon Peninsula (10 per linear disturbance type), we assessed non-native and native plant species diversity via two types of transects: 1) linear disturbance transects and 2) forest transects. We placed a 50 metre transect along the length of the linear disturbance within areas that contained at least 100 m^2 of intact forest (Figure 2.2a) and identified all non-native plant species in plant communities between the disturbance and adjacent forests (typically within a 1 m belt), thereby identifying the non-native species pool with potential to invade adjacent forest stands based on proximity. We measured the linear disturbance width at five locations spaced 10 m apart along the linear disturbance transect, allowing us to calculate an average width for each disturbance. We placed five forest transects perpendicular to the linear disturbance and continuing into the forest for 50 m. The forest transects were spaced ten metres apart, starting at five metres along the linear disturbance transect and ending at 45 metres. Forest transects were run perpendicular to the linear disturbance to assess potential changes in plant community composition (including non-native species establishment) with increasing distance into the forest stand.

Forest vegetation surveys

At five-metre intervals along each forest transect, we visually quantified the native and non-native species richness and percent cover within a 1 m x 1 m quadrat (fifty total quadrats per site) (Figure 2.3). Percent cover was quantified by a single individual throughout all sites to within +/- 5% precision. Quadrats were placed with the lower left corner at the appropriate metre division. All vascular plant species were identified to the species level, while Bryophytes were categorized as being either acrocarpous, pleurocarpous type, or sphagnum. Any additional observations of non-native species along the transect, but aside the quadrats, were noted. Identification of native and non-native species was completed using the guides listed in Appendix I.

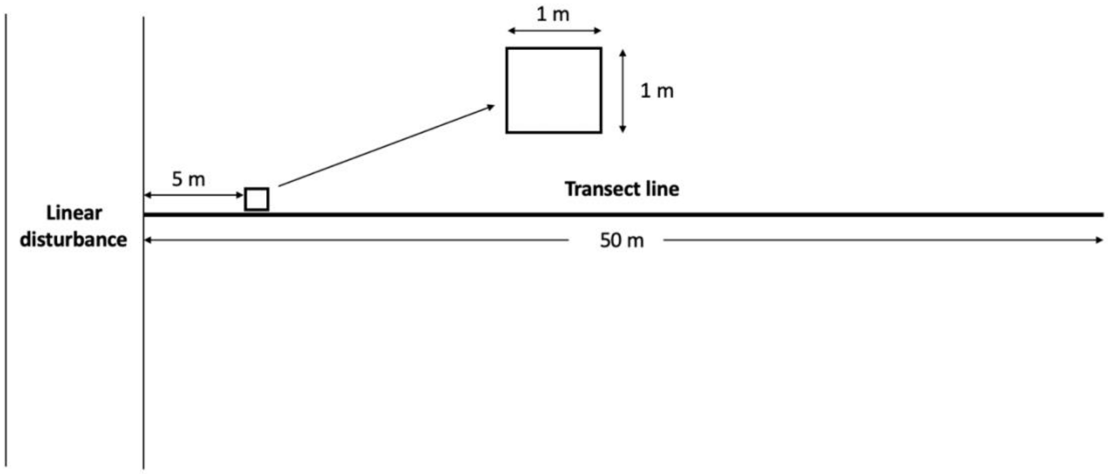

Figure 2.3 Forest vegetation surveys were completed during the 2018 field season using a 50 metre transect line, running perpendicular from a linear disturbance into the adjacent intact forest. Avalon Peninsula, Newfoundland and Labrador, Canada.

Along each forest transect, we also quantified tree composition using a 2 m wide belt transect. All trees were identified to species, and tree density was measured by counting the number of individuals present per belt transect. One individual of the dominant tree species per belt transect was cored to estimate the average age of the forest stand. The individual chosen was considered an example of an average tree for the forest stand being measured. For each non-native tree species surveyed, we recorded its distance from the linear disturbance to assess the penetration of non-native trees into the boreal forest stands.

Environmental characteristics

Within each of the fifty 1 m^2 quadrats along forest transects, we collected measurements to assess the extrinsic conditions that might allow for larger amounts of non-native species establishment (Appendix II, Table A11.2). We estimated canopy cover to assess the light availability along the transect line using a Forestry Suppliers Spherical Crown Densiometer, Convex Model A. Depth of the soil organic layer (from mineral soil surface to the base of living vegetation) was measured within each quadrat and ~250 g samples of mineral soil were taken from a subset of quadrats (totalling nine samples per site) to assess the pH and texture of the mineral layer. The organic soil layer was replaced after the sample was taken for minimal disturbance. We measured percent slope along each transect using a clinometer at the entrance of the forest stand at a distance of 20 m, these totals were then averaged to calculate a site-level slope.

Soil pH was analyzed following Robertson et al. (1999) as follows; from each undried soil sample (fresh-frozen and thawed for analysis), two 15 g subsamples were weighed and separated. Thirty mL of deionized water was added to each soil subsample. The mixture was then stirred well and allowed to stand for 30 minutes, after which the mixture was gently stirred while the pH reading was taken with a Fieldscout pH 400 meter. We calibrated the pH meter using a 4.0 pH and 7.0 pH solution after every 10 subsamples and rinsed the pH electrode with deionized water between each sample.

We analyzed percent of sand in mineral soil by drying soil samples in a convection oven at 60 degrees Celsius for 24 hours. The dried samples were weighed and then shaken through 2 mm and 0.05 mm sieves, which were used to separate sand from silt and clay soil types. Separated sand was then weighed and divided by the total soil sample weight to get a percentage of sand. We then compared these percentages to assess if stands with a higher percentage of sand, and therefore higher ground drainage, showed any variation in species composition as well as other environmental conditions.

A measurement of total kilometres of road was taken within a 2 km^2 area surrounding each linear disturbance. This was completed by measuring area distance on Google Earth and quantifying the length of all roadways (paved and gravel) within this area.

2.2.2 Stream survey

Sampling design

Stream survey study sites were selected based on the presence of a riparian area that was crossed by a gravel road within boreal forest stands. Forest road-stream crossings were first identified on Google Earth, and then visited in person to verify their suitability for the study. At each of 30 stream-road crossing sites located around the Avalon Peninsula, we assessed non-native plant species diversity via two types of transects: 1) linear disturbance transects and 2) stream transects. Following the methods described for linear disturbance transects, we placed a 25 metre transect along the length of the linear disturbance on either side of the adjacent stream (i.e., perpendicular to the stream; for a total of 50 m of survey transect), and identified all non-native plant species in plant communities present along the disturbance, thereby identifying the non-native species pool with potential to establish within adjacent riparian zone. Three measurements were taken (at 0 m, 25 m, and 50 m along the forest road) to calculate an average width of each forest road.

Stream vegetation surveys

We placed a 50 m stream transect both upstream and downstream of the linear disturbance crossing. Along the entire length of each stream transect we quantified non-native plant species richness along both banks of the riparian zone, from the water edge to the adjacent boreal forest community. The area of the bank varied between sites, but was generally within one to several metres wide. For each non-native species observation, we recorded three spatial attributes: 1) Distance from road was measured to assess the degree to which non-native species have established from potential population sources (the road crossing); 2) The patch size of observed populations was measured to assess the extent of their establishment along our selected riparian zones; and 3) Area of establishment of each observed species within three different bank zones was measured to assess whether streams are acting as a vector for non-native species to colonize adjacent boreal forest stands. The bank zone classification used for this study was: 1) submerged zone, where consistent stream flow was present; 2) flood zone, outside of regular stream flow where there were signs of flooding and freezing; and 3) upland zone, where the riparian area begins to transition into the adjacent boreal forest community. Non-native plant species abundance was measured by calculating the total measured distance each species was present along our stream banks.

Environmental characteristics

The width of each stream was measured at six locations (3 upstream and 3 downstream of the forest road crossing) to calculate an average stream width. We made generalized assessments of the stream substrate of each study stream to allow further classification of streams for analysis (e.g., gravel, cobblestone, etc.). Width measurements were taken of each bridge or culvert to estimate potential flow of the stream (i.e., we assumed that streams with larger culverts had the potential for greater seasonal flow). All measurements were taken in during summer 2019, between July 7th to July 23rd.

2.2.3 Statistical analyses

All statistical analyses were performed using R version 3.6.1 (R Core Team, 2019) via RStudio version 1.2.5001 (RStudio Inc., 2019). We used the packages “ggplot2” (Wickham, 2016), “jtools” (Long, 2020), and “interactions” (Long, 2019) for analysis of our generalized linear model and used the “vegan” (Oksanen et al., 2013), “ecodist” (Goslee & Urban, 2007), and “MASS” (Venables & Ripley, 2002) packages for our ordination analysis.

We used general linear models to test the relationships between response variables (non-native plant species richness, on linear disturbances and stream banks, as well as native species richness within forest stands) with various environmental characteristics (i.e., canopy cover, forest density, soil pH, stream width, etc.), for both forest and stream site data (Appendix II, Table AII.2 and Table AII.4, respectfully). A polynomial function was used to assess the non-linear relationship between non-native species richness on linear disturbances and linear disturbance width. One-way analysis of variance (ANOVA) was used to compare ecoregions with forest and trail species richness (Table AII.3), bank zones with non-native species abundance (Table AII.6), and stream sediment type was compared with stream non-native species richness and non-native species abundance

(Table AII.5). To consider any interactions, we tested the influence that disturbance width had on non-native species richness when total kilometres of road was varied, three disturbance widths were selected for our equations (1.0 m, 4.0 m, 7.0 m) based on the range of width observed throughout our study sites. A generalized linear model was used to test the response of non-native plant species present on linear disturbances with a negative binomial distribution. Negative binomial was found to have the best fit for our data after first attempting analysis with Poisson and Quasi-Poisson family models (Appendix VI).

Multivariate analysis

The forest vegetation and environmental characteristic data as well as stream vegetation data were analyzed using non-metric multidimensional scaling for the purpose of highlighting any variance in species community based on our chosen categorizations (NMDS, Vegan package). NMDS is an ordination technique that places ecological data within a chosen number of axes based on the similarity of the observed data variables. Our NMDS used Bray-Curtis coefficients as measures of dissimilarity. The best solutions were reached under two dimensions. For the entire dataset the percent cover of all species was averaged for each site and rounded to the nearest tenth decimal place. This caused the exclusion of some species from our ordination since they were observed in very small percentages, causing them to be averaged to zero percent. Out of a total of 81 species, 18 species were excluded from our ordination. The majority of these species were native, but the exclusion also contained seven non-native species (*Taraxacum officinale*, *Rosa* *cinnamomea*, *Rosa rubiginosa*, *Veronica serpyllifolia*, *Anthoxanthum adoratum*, *Hypericum perforatum*, and *Rumex acetosella*).

2.3 Results

2.3.1 Forest survey

Distribution of non-native plants

Linear disturbances

We detected a total of 38 non-native plant species (Appendix III) on linear disturbances within 25 of our 30 study sites (Figure 2.4). The five sites in which we did not observe non-native species were all within the walking trail category of linear disturbance. This category is the smallest magnitude of disturbance out of the three types, as the average width of the walking trails were smaller than both ATV trails and forest roads (Figure 2.5). Although there was some overlap of disturbance width between types (Figure 2.5), the three types remained distinct based on accessibility. Site locations were focused on central and northern regions of the Avalon Peninsula based on availability of sites within the AFE and MBE that contained the previously stated criteria. The majority of non-native species observed were categorized as forbs, i.e., herbaceous flowering plants, followed by graminoid species, although graminoids were some of the most common species observed throughout our sites (Figure 2.4). The most common non-native plant observed was *Hieracium vulgatum*, which was present at 64% of sites containing non-native species (16 sites, Figure 2.4). Other common non-native species were *Agrostis capillaris*, *A*, *canina*, *Veronica officinalis*, and *Anthoxanthum officinalis*.

The most common families observed were Asteraceae (26% of species), Poaceae (18% of species), and Plantaginaceae (11% of species). Some species were only observed once, such as *Betula pendula*, which was found on a trail that was in close proximity to a community. *Myosotis sylvatica and Potentilla simplex* are species that are common within communities on the Avalon Peninsula, but were each only observed at one of our sites. *Hypericum perforatum* was only found at two of our sites in spite of its widespread distribution throughout disturbed habitats of Newfoundland. The invasive species, *Centaurea nigra,* was present at 16% of sites containing non-native species (five sites), all of which were categorized as forest road disturbance sites, which generally had the largest measurement of disturbance width (Figure 2.5).

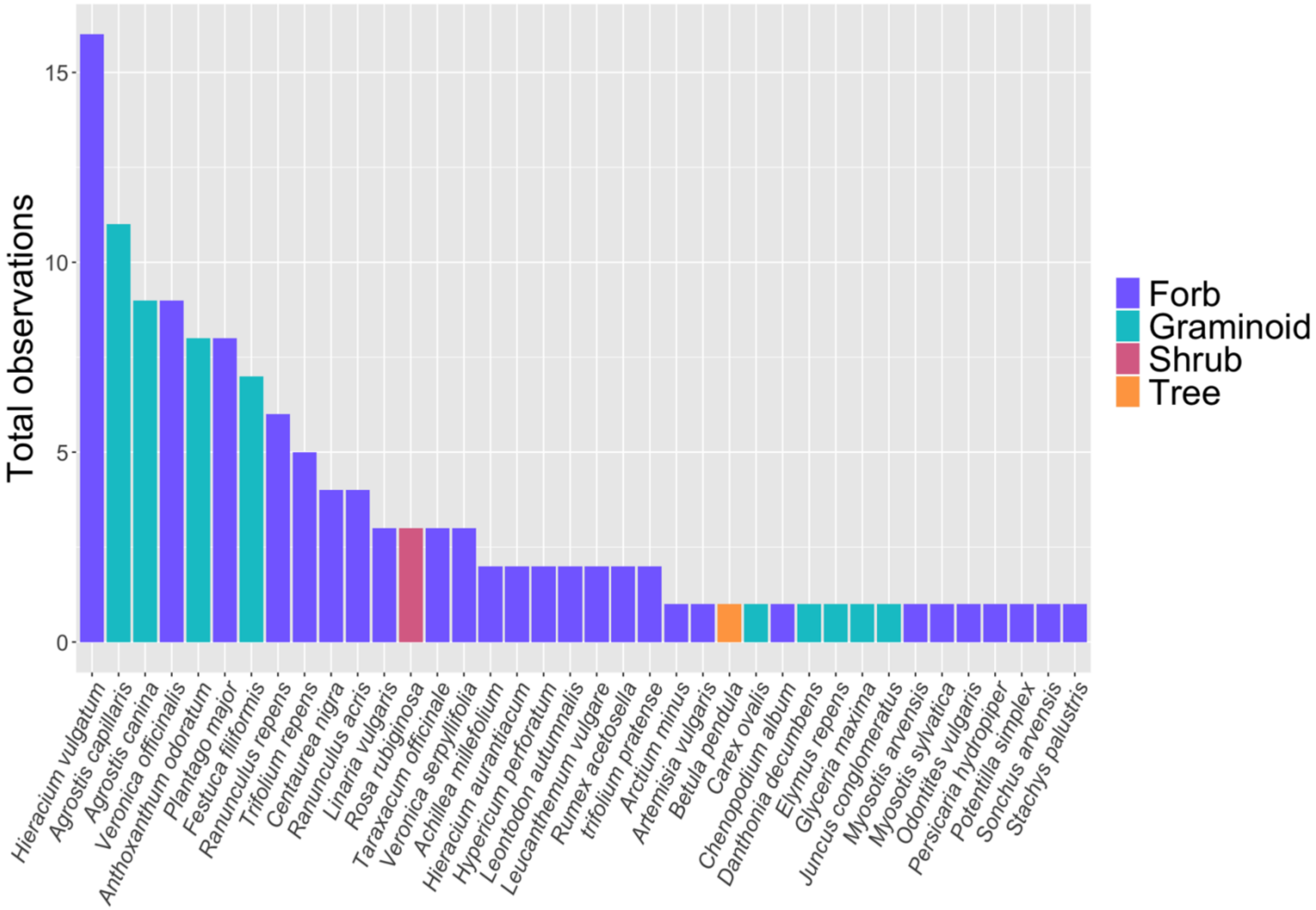

Figure 2.4 Total observations of non-native plant species present on 30 linear disturbances within the boreal region of the Avalon Peninsula of Newfoundland and Labrador, Canada. These species were categorized within four groups; forb, graminoid, woody shrub, and tree (Identification via guides listed in Appendix I).

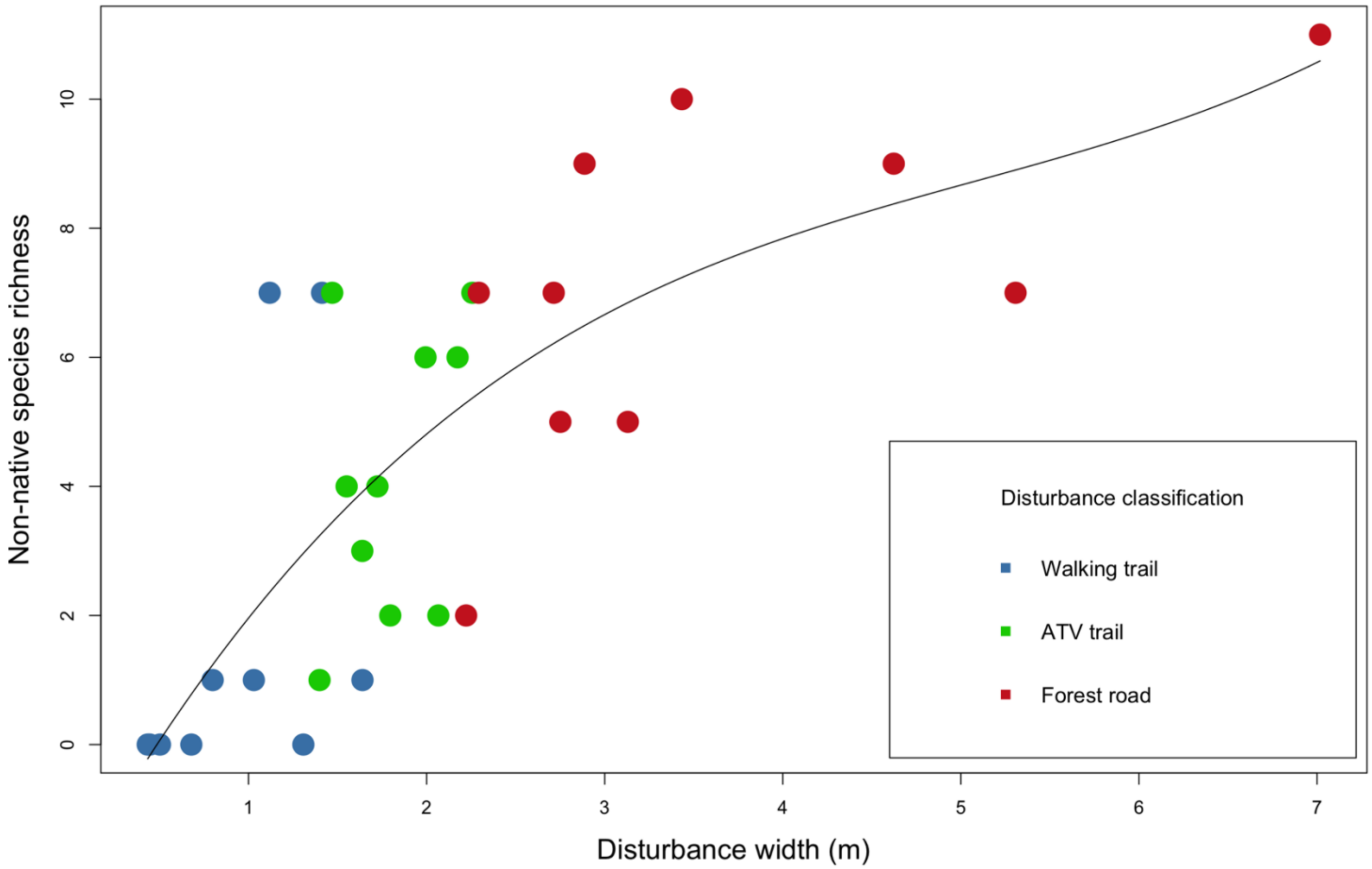

Figure 0.5 The relationship of non-native species richness and width of 30 linear disturbances located within the boreal forest of the Avalon Peninsula of Newfoundland and Labrador, Canada. Each linear disturbance site is categorized by type of disturbance.

When we assess all of our forest sites, our generalized linear model revealed that there is no apparent interaction between linear disturbance width and total kilometres of road surrounding each site in regards to non-native species richness on disturbances ($p<0.348$, z-value=-0.939). Our general linear models revealed that non-native plant species richness on linear disturbances was significantly associated with the width of the linear disturbance ($p<3.37x10^{-6}$, t-value = 5.774), with wider linear disturbances showing higher non-native species richness. Total kilometres of road within a 2 km^2 area surrounding the linear disturbance did not have a significant relationship with non-native species richness on linear disturbances ($p<0.12$, t-value = 1.404).

Forest vegetation and environmental characteristics

The presence of non-native plants within boreal forest stands was rare, despite the occurrence of non-native species on adjacent linear disturbance, as summarized above. A large proportion (47%) of non-native species observed within forest stands were excluded from our ordination due to their coverage being too low to analyze (i.e., averaging to zero percent cover per site). Therefore, due to these limited occurrences, an analysis on non-native species within the forest could not be completed. Fifteen non-native species were observed throughout all forest sites but only eight of these species were included in our ordination analysis; these included *Hieracium vulgatum*, *Ranunculus repens*, *Veronica officinalis*, *Festuca filiformis*, *Juncus conglomeratus*, *Agrostis canina*, *Sorbus aucuparia* and *Acer pseudoplatanus*. Of the 30 sites assessed, only five contained non-native species in quantities too small to analyze, these sites were in the ATV trail and forest road category of disturbance only.

General linear models demonstrated (Appendix II, Table AII.2) that native species richness within the forest stands adjacent to the studied linear disturbance had a positive association with average soil pH ($p<0.003$, t-value=3.289, Figure 2.6), where sites with higher pH measurements were associated with a higher native species richness. Also demonstrated in our models was a lack of relationship ($p>0.05$) between forest native species and all other environmental characteristics, such as, canopy cover, soil organic layer thickness, percent sand, stand density, and average stand age (Figure 2.6). All native species observed are listed in Appendix IV, Table AIV.1.

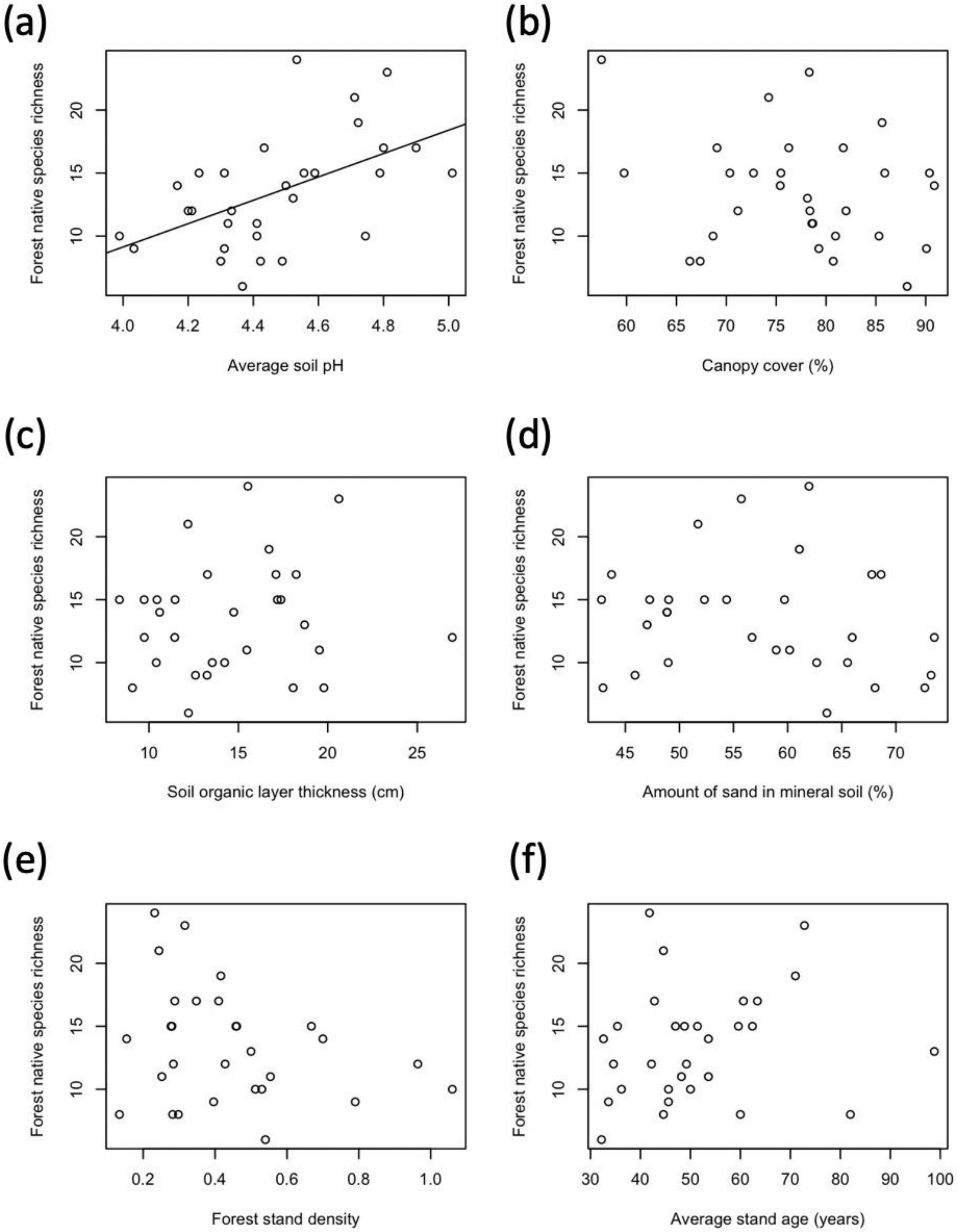

Figure 0.6 Influence of (a) average soil pH, (b) canopy cover, (c) soil organic layer thickness, (d) percent sand in mineral soil, (e) forest stand density, and (f) average stand age, on native species richness of forest stands adjacent to a linear disturbance. Linear lines are present only on plots that had a significant correlation ($p<0.5$). Avalon Peninsula, Newfoundland and Labrador, Canada.

Forest composition

Nonmetric Multidimensional Scaling (NMDS) ordination revealed that forest stands adjacent to walking trails, ATV trails, and forest roads exhibited similar species composition, evidenced by the overlap of the three disturbance type confidence ellipses (Figure 2.7). This pattern suggests that the classification of linear disturbance, as well as its magnitude of disturbance (i.e., width), does not influence the species composition observed within adjacent forests in the sites that we observed. The 95% confidence ellipses for all three of our disturbance classifications had a large overlap of area within our ordination. ATV trails seem to encompass the widest ordination area (0.5501), signifying the largest variation in plant species within sites. Forest roads and walking trails have a smaller ordination area (0.2993 and 0.2783, respectively), encompassing a smaller subset of forest species. The stress value of an NMDS ordination reflects how well the ordination summarizes the observed distance among samples, where the commonly accepted stress limit for NMDS representation is 0.2 and lower (Dexter et al., 2018). Our NMDS ordination has a stress of 0.147, which is within the previously stated limit, suggesting that our ordination is a good representation of our data in two-dimensional space

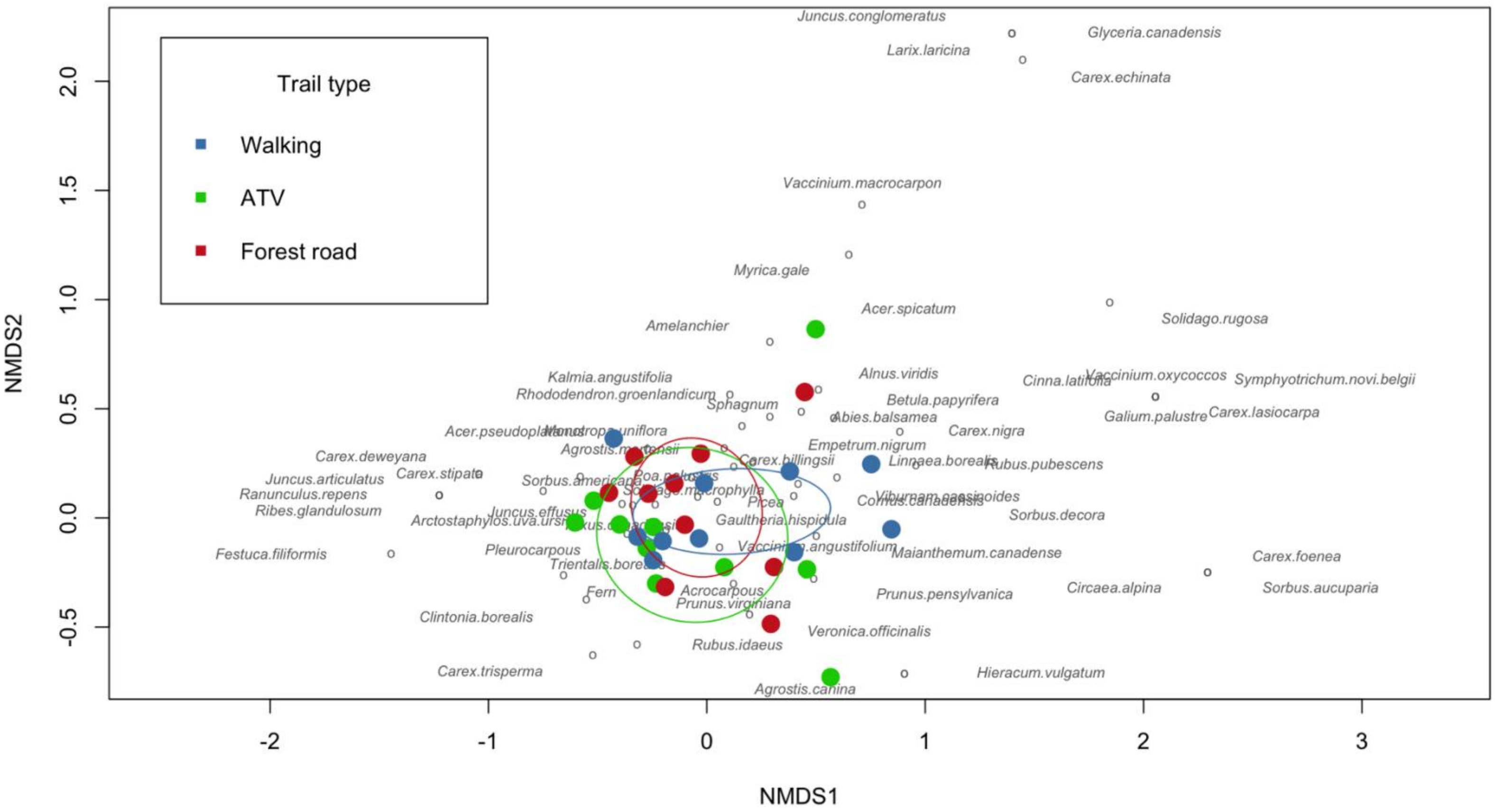

Figure 2.7 Nonmetric Multidimensional Scaling ordination of the forest understory plant community of 30 sites within the boreal forest in the Avalon Peninsula of Newfoundland and Labrador, Canada. Sites are categorized by type of linear disturbance.

Our overlay of environmental vectors on the NMDS suggests that increasing forest stand density was associated with shade-tolerant boreal species, such as *Clintonia borealis* (Figure 2.8, vectors included are all significant, with $p<0.05$). Forest stand density was also associated with decreased soil pH as well as decreased proportion of deciduous tree species. The increase of understory species richness was associated with species that are light-tolerant, such as *Kalmia angustifolia* and deciduous tree species *Amelanchier sp.* and *Acer sp*. Increasing soil pH was associated with graminoid species, such as *Carex echinata*, *Agrostis mertensii*, and *Juncus conglomeratus*, whereas soils with lower pH are associated with common boreal forest species, such as *Clintonia borealis* and *Trientalis borealis*. Increasing canopy cover was associated with shade-tolerant species similar to those observed with increasing forest stand density.

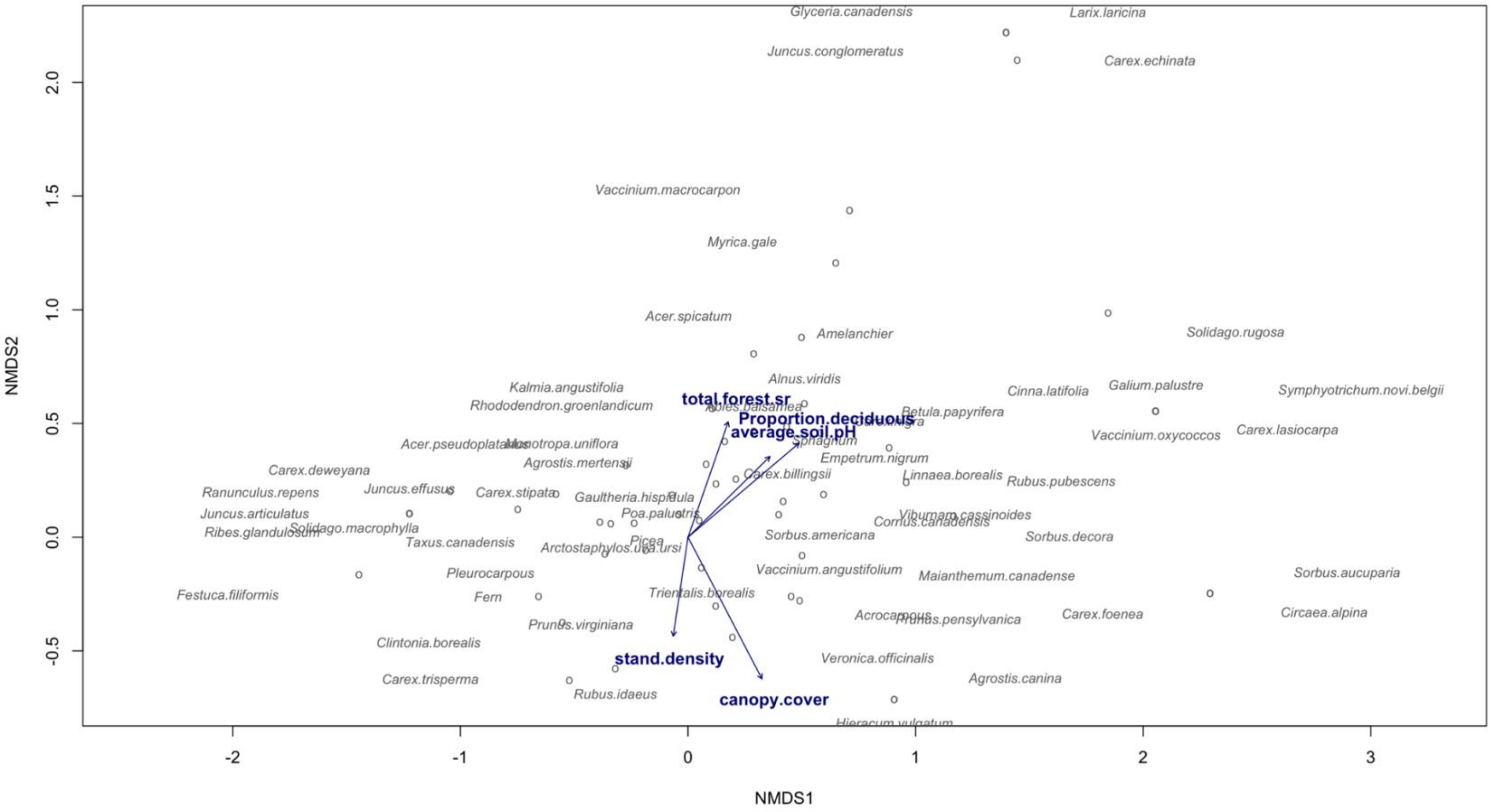

Figure 0.8 Nonmetric Multidimensional Scaling ordination of forest understory plant community at 30 sites located on the Avalon Peninsula of Newfoundland and Labrador, Canada. Fitted significant environmental characteristics ($p<0.05$) layered as vectors.

2.3.2 Stream survey

We detected a total of 47 non-native plant species (Appendix III) on linear disturbances and stream banks within 30 study sites. Non-native plant species were observed on all forest roads that were assessed and were present on every stream that the roads crossed. The majority of the non-native species were found on both stream banks and forest roads (53%), with some found only on forest roads (37%) or only along stream banks (10%) (Figure 2.9). The majority of non-native species observed were in the family Asteraceae (28% of species) and Poaceae was the second most common family (15% of species). The vast majority of species were categorized as forbs (81% of species), while graminoid (15% of species) and tree (4% of species) were also observed. The most common non-native plant observed was *Ranunculus repens*, which was present at 93% of sites (Figure 2.9). Other common non-native species were *Pilosella caespitosa* (90% of sites), *Hieracium vulgatum* (87% of sites), *Taraxacum officinale* (80% of sites). *Centaurea nigra*, a well-known invasive plant in the area, was present at 73% of sites (Figure 2.9).

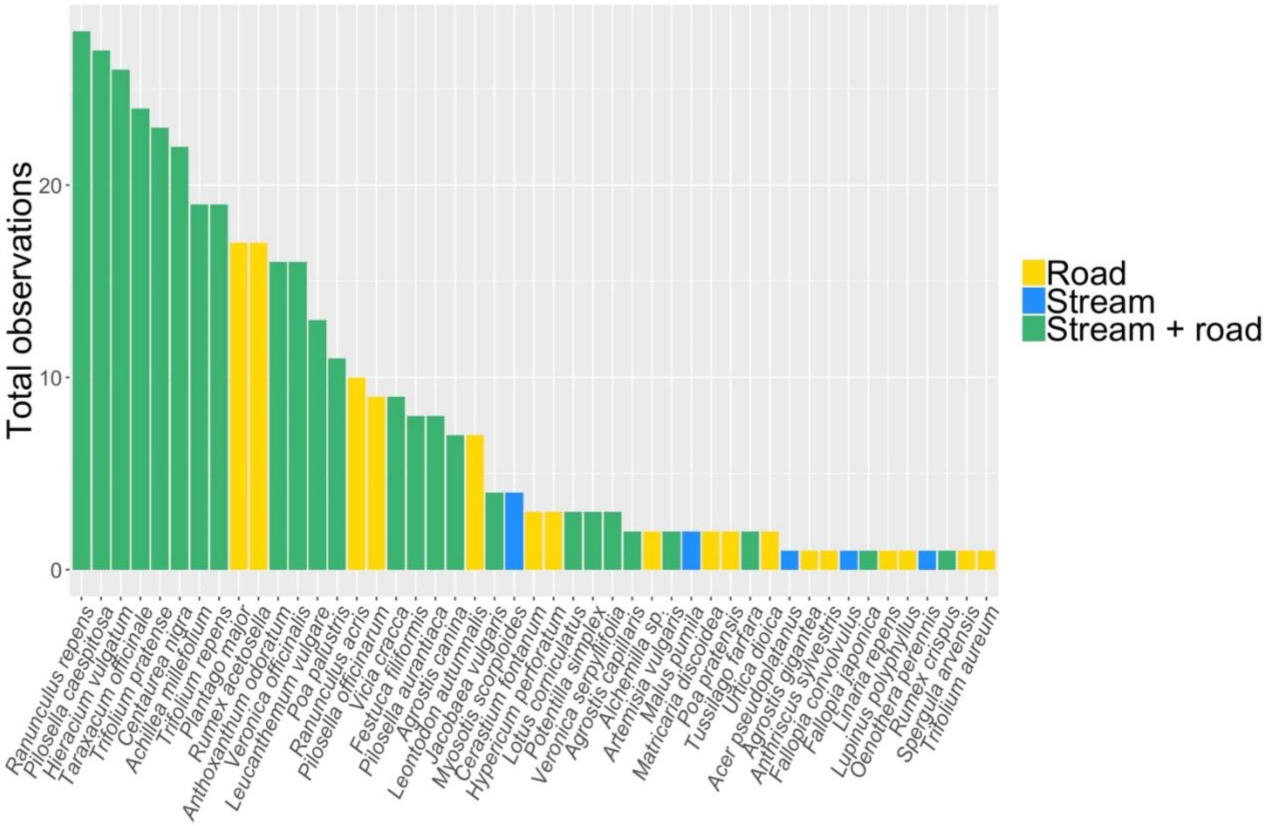

Figure 0.9 Total observations of non-native plant species present on 30 forest roads and stream crossings within the boreal region of the Avalon Peninsula, Newfoundland and Labrador, Canada. These species were categorized within three groups; i) species found only along streams, ii) species found only along roads, and iii) species found on both roads and streams.

Non-native plant species richness and abundance varied on stream banks, despite the occurrence of non-native species on adjacent linear disturbance. Although width of linear disturbance was significantly associated with non-native plant species richness in our forest site analysis (Figure 2.4), our general linear model revealed that this was not the case for forest roads at our stream sites (Appendix II, Table AII.4). General linear models demonstrated that non-native species richness on streams was not significantly associated with non-native species richness on roads (Figure 2.10). Total kilometres of road within a 2 km^2 area surrounding the linear disturbance had a positive significant association with non-native species richness of stream banks, although species richness shows much variation in our plot ($p<0.008$, t-value = 2.856, Figure 2.10c). Total kilometres of road was not significantly associated with non-native species richness of forest roads. The abundance of non-native species along stream banks was not significantly associated with total kilometres of road surrounding the site, but did have a positive significant relationship with linear disturbance width ($p<0.058$, t-value = 1.977), with a higher abundance of non-native species along stream bank associated with wider linear disturbances. Some streams had exceptionally high non-native species richness, causing some outliers in our plot (Figure 2.10f). Abundance had a positive significant relationship with the non-native species richness of streams ($p<1.53x10^{-5}$, t-value=5.217), streams with a higher non-native species richness tended to have a higher non-native species abundance. Our analysis of dominant sediment type of streams (Appendix II, Table AII.5) showed that gravel dominant stream beds had a higher non-native species richness and abundance among our sites (Figure 2.11).

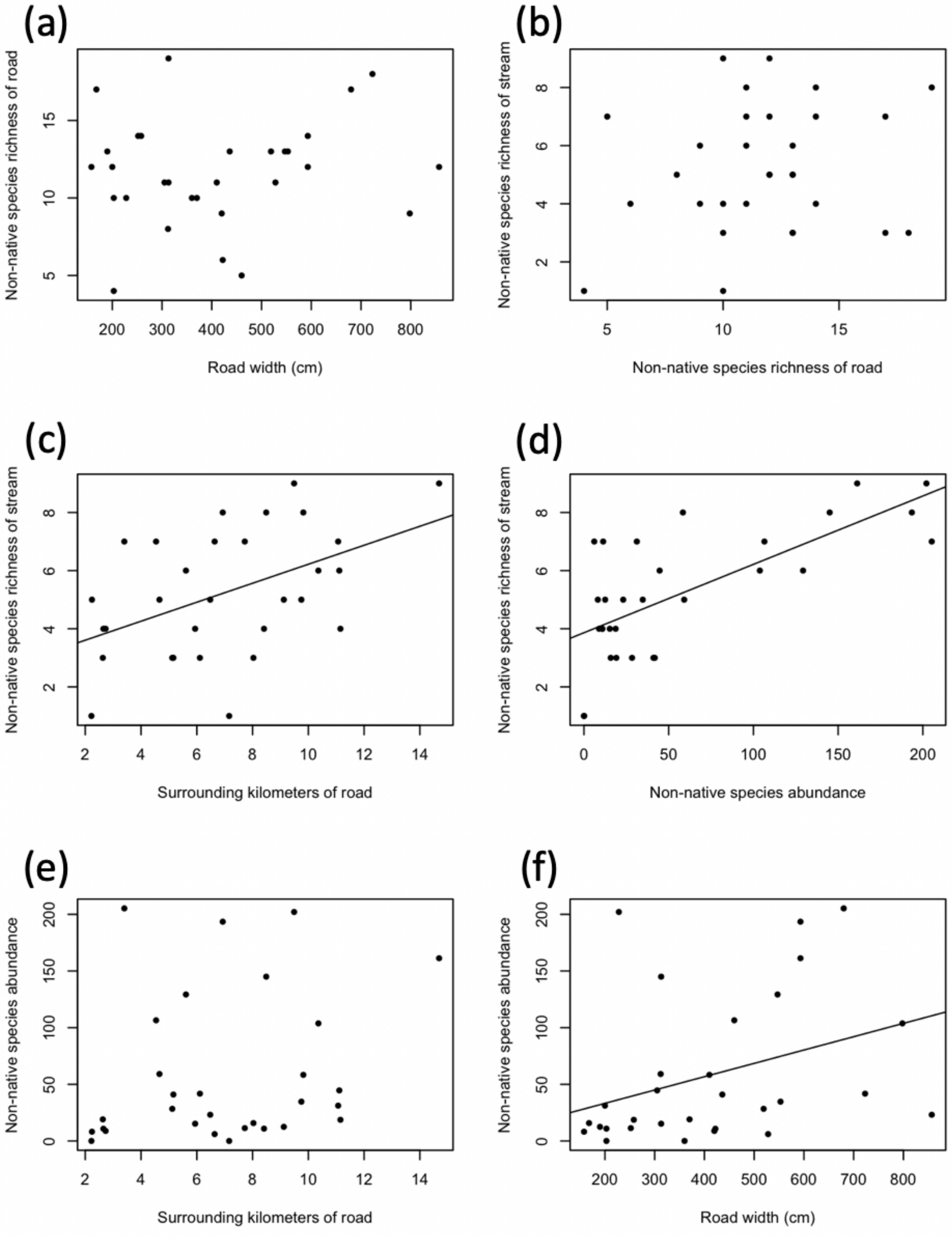

Figure 0.10 Interactions of non-native species richness on roads with (a) road width; non-native species richness of streams with (b) non-native species richness of roads, (c) surrounding total kilometres of road, and (d) non-native species abundance along stream banks; and non-native species abundance along stream banks with (e) surrounding total kilometres of road and (f) road width. Linear lines are present only on plots that had a significant correlation ($p<0.06$). Avalon Peninsula, Newfoundland and Labrador, Canada.

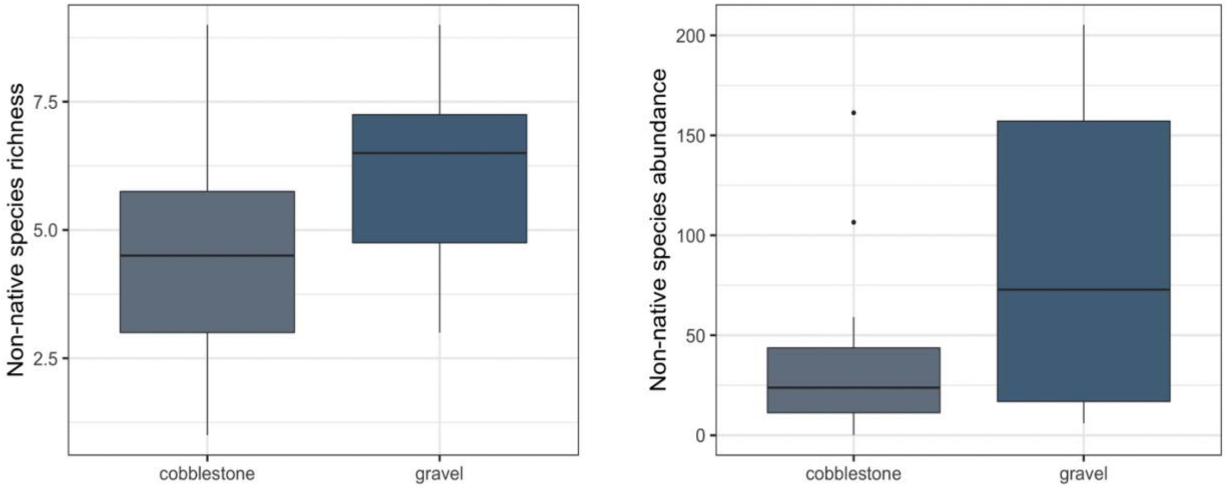

Figure 0.11 Comparison of non-native species richness and abundance along streambanks based on dominant sediment type of streams within the Avalon Peninsula, Newfoundland and Labrador, Canada. Sediment was categorized into one of two dominant sediment types; i) cobblestone or ii) gravel.

The non-native species richness along stream banks was at its highest in closer proximity to the forest road crossing and declined as the distance from the road increased (Figure 2.12) to about 30 m downstream where numbers stabilised. The majority of non-native species observed along the stream bank were within the flood-zone region of the bank (p<0.00069, f-value=7.929, Figure 2.13). Stream characteristics not mentioned in this chapter can be found in Appendix V, Table AV.1.

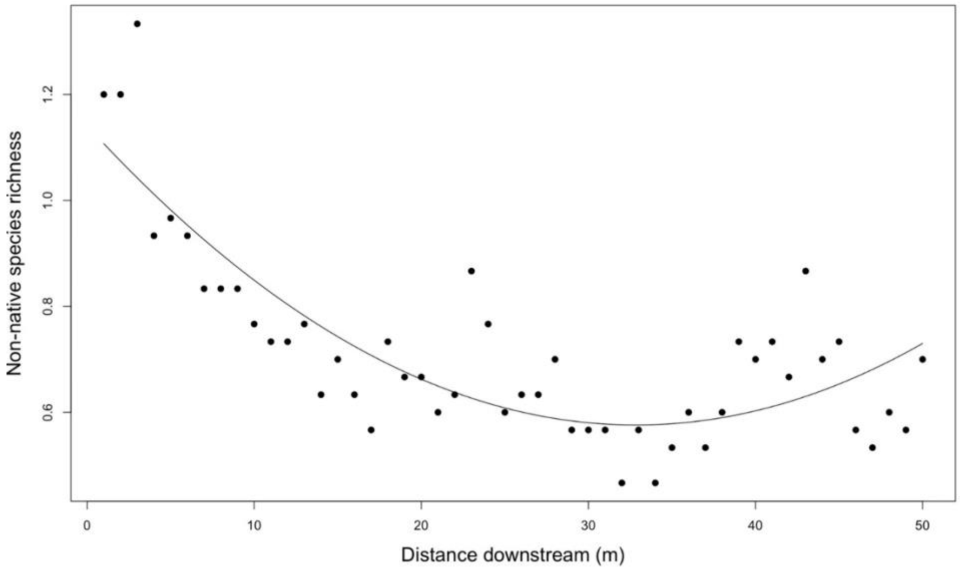

Figure 0.12 Non-native plant species richness along the downstream bank of 30 stream sites within the Avalon Peninsula, Newfoundland and Labrador, Canada. Measurements were taken along a distance of 50 metres from a forest road crossing.

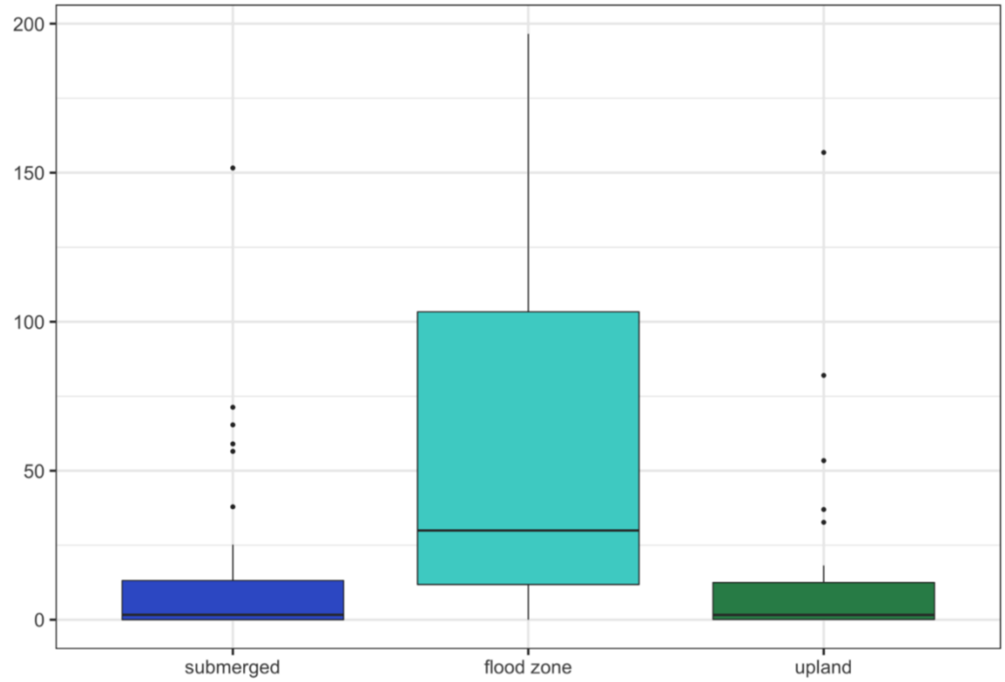

Figure 0.13 Abundance of non-native species observed within each of three bank zone categories along natural streams within the Avalon Peninsula, Newfoundland and Labrador, Canada.

Nonmetric Multidimensional Scaling (NMDS) ordination revealed that forest roads adjacent to streams exhibit a different species composition than both upstream and downstream sections of the streams. The 95% confidence ellipses for the two stream sections suggest that these sections of stream adjacent to forest road linear disturbance type exhibited similar species composition, evidenced by the overlap of the upstream and downstream disturbance type confidence ellipses (Figure 2.14). The 95% confidence ellipses for all three of our classifications suggest that forest road linear disturbance types show less similarity to upstream and downstream banks than the stream banks similarity to each other within our ordination. Forest road linear disturbances seem to encompass the narrowest ordination area (0.2233), signifying the smallest variation in non-native plant species between sites. Upstream and downstream regions have a wider ordination area (0.4898 and 0.4123, respectively), encompassing a larger variation of non-native species between sites. Our NMDS ordination has a stress of 0.145, which is within the previously stated limit, suggesting that our ordination is a good representation of our data in three-dimensional space.

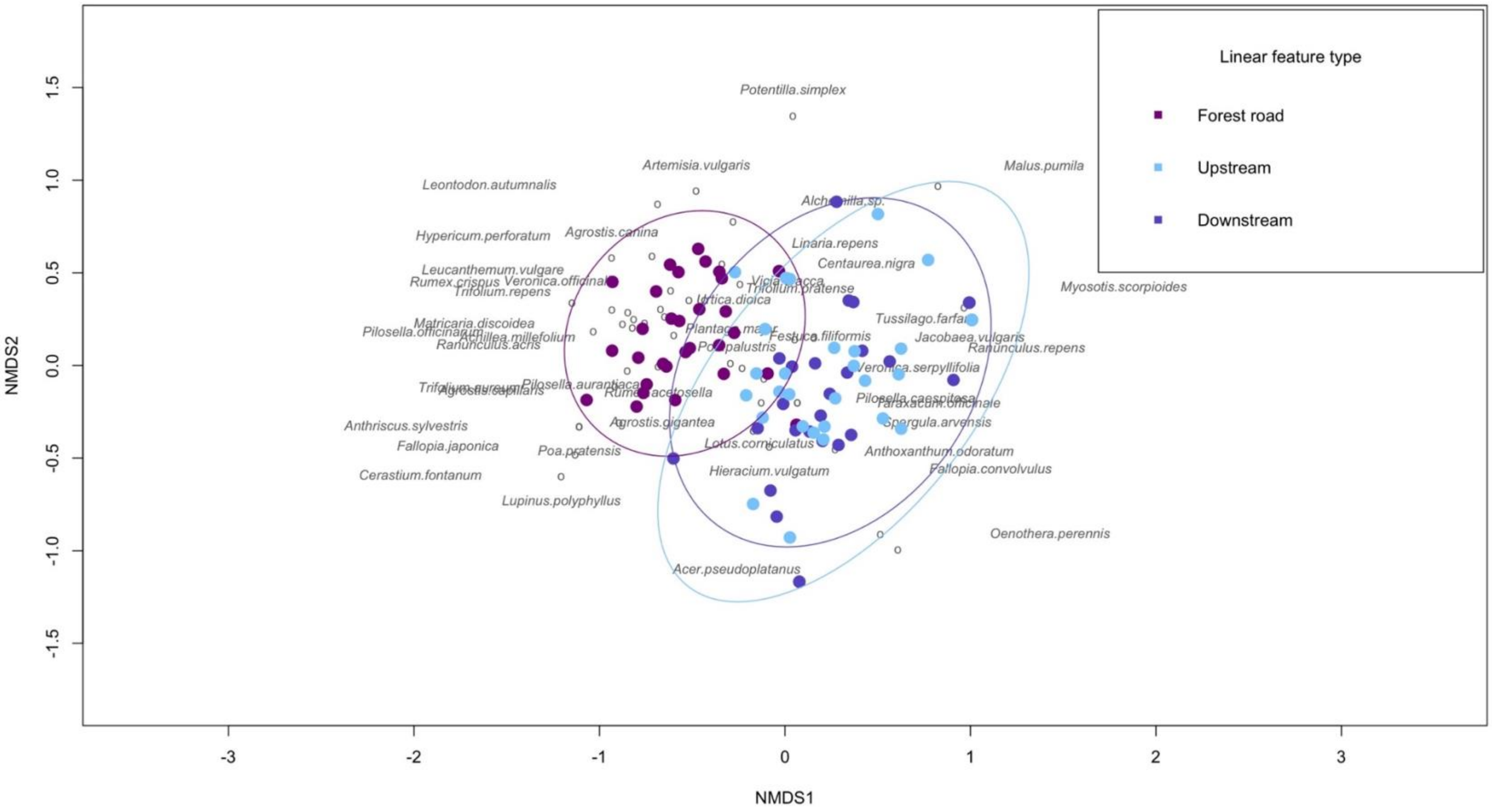

Figure 0.14 Nonmetric Multidimensional Scaling ordination of the non-native species composition of 30 sites located at forest road and stream crossings within the boreal forest region on the Avalon Peninsula, Newfoundland and Labrador, Canada. Sites are categorized by type of linear disturbance.

2.4 Discussion

We present evidence that non-native plant species are found extensively along anthropogenic and natural linear features surrounding eastern boreal forest stands in the Avalon Peninsula region of Newfoundland, Canada. Along anthropogenic linear features, the number of non-native species found increased with the magnitude of human disturbance, from hiking trails to forest roads. Non-native species were also found established along every natural stream crossed by a forest road in our study. Yet, despite the prevalence of non-native species along these features, small, relatively intact patches of boreal forest found within this fragmented landscape contained very few non-native species in our study sites on the Avalon Peninsula. We explore each of these findings in detail, below.

2.4.1 Corridors of colonization: anthropogenic linear features

Here, we demonstrate that the distribution and richness of non-native species on the Avalon Peninsula of Newfoundland is related to the intensity of human disturbance. Of the three types of anthropogenic linear disturbance assessed, only walking trails had sites with no non-native species present along them. These disturbances, being the smallest in magnitude, experience much less total disturbance than the other types of linear disturbance we assessed. Since the width (0.43 m to 1.64 m) of our walking trails was relatively small, the canopy cover of the trail was generally higher along these disturbances, contributing to the lowered light availability, and, therefore, the lower number of non-native ruderal species observed. Our walking trails also had many areas where the organic soil layer was compacted from foot traffic, but still intact; this can influence the species observed due to the difficulty that non-native plants might have germinating and establishing in the thick organic soil layer of the boreal forest understory.

The width of linear disturbance was significantly associated with non-native species richness at our forest sites but not at our stream sites, where we focused only on the largest magnitude of linear disturbance, forest roads. The range of width for both forest and stream sites were similar, forest sites had a range of 0.43 m to 7.02 m (6.69 m variance), while our stream sites had a range of 1.59 m to 8.57 m (6.98 m variance). This suggests that the use of linear disturbance has a large impact on the presence of non-native species, as it influences the trampling disturbance in the area as well as the propagule pressure. When considering our walking trail sites for our forest survey, we notice that half of the sites had a trail width lower than one metre, of these sites only one had non-native species present (20%), whereas the other half of our forest sites were over one metre in width and four of the five site contained non-native species (80%). The influence of light availability is apparent here, as trails that are less than one metre wide would have little, if any, canopy openings that are large enough to support ruderal species growth. Charbonneau and Fahrig (2004) found that as canopy cover decreased within their forest sites, the proportion of non-native plants species increased, suggesting that decreased canopy cover facilitates the establishment of non-native species.

Any variation between forest stands was influenced by environmental conditions rather than the adjacent linear disturbance. Our ordination suggested that higher canopy cover and stand density generally resulted in a plant community that was composed of species that prefer low-light conditions, such as *Cornus canadensis* and *Maianthemum* *canadense*. However, based on our models, these environmental conditions (including stand age and soil organic layer thickness) did not influence the total native species richness of our forest stands. This lack of association may be due to the length of our forest transects; 50 metres may not have been long enough to accurately characterize forest stands based on adjacent linear disturbances or environmental conditions.

One environmental factor that did influence our native species richness was the average soil pH measured. As pH increased, becoming more neutral, more species were observed. The conditions that are commonly observed within boreal forest stands can influence the presence of non-native species. Conifer-dominant forests, the focus of our study, generally have challenging understory conditions due to their denser canopy limiting light availability and altering the microclimate of the forest understory (Chavez & Macdonald, 2009; Macdonald & Fenniak, 2007), and are associated with a lower soil pH due to the shedding of acidic leaf litter. The plants that are commonly found in denser, low light portion of the boreal forest are adapted to these low-light conditions (e.g., *Maianthemum canadense*). Gaps in canopy increase light availability and can change forest floor characteristics, which facilitate the establishment of shade-intolerant understory plant species (Cole, 2004; Chavez & Macdonald, 2002). Our ordination also suggested that a higher proportion of deciduous tree species and higher mineral soil pH were also associated with a higher total species richness. A higher proportion of deciduous tree species can influence the micro climate, possibly allowing more canopy openings and increasing the pH of the soil (Ovington, 1954). Sites with these conditions had a combination of shade-tolerant and shade-intolerant species, such as *Vaccinium* spp. The three linear disturbance types, each varying in magnitude and width range, did not influence the native plant community of our sites, based on our ordination. All three disturbance types had very similar overlap in community. This may also have been influenced by the length of our forest transects, a longer transect line may show this relationship in the future.

2.4.2 Linear disturbances as propagule sources into natural linear features

Non-native species were observed within each riparian study area in our stream survey, but species richness varied between sites, independent of the non-native species richness of the adjacent forest road. The total surrounding kilometres of road within a 2 km^2 area had a positive significant correlation with non-native species richness along stream banks, suggesting a strong influence from the surrounding landscape. Although species richness and abundance along stream banks did not have a strong correlation with species richness along forest roads, the non-native species richness observed along stream banks was highest near the road crossing and tapered off as the distance from the crossing increased. Anthropogenic linear disturbance crossings over natural streams influence the non-native plants species observed along stream banks and can be considered hot spots for these species' introduction and establishment in naturally disturbed areas. Riparian areas have been found to have a greater species richness as well as species cover of non-native species relative to upland/non-riparian areas (Brown and Peet, 2003). We found that non-native species abundance was influenced by non-native species richness along stream banks as well as the width of forest roads. As the width of our forest roads increased, which we can correlate with increased usage and therefore magnitude of disturbance, the abundance of non-native species along stream banks also increased. Common species that were high in abundance along stream banks were *Pilosella* sp. as well as *Ranunculus repens*. *R. repens* has been known to form large clonal patches in both forested and riparian areas of Newfoundland (Rose and Hermanutz, 2004) and is a species that we observed in low quantities within our forest sites as well. *Pilosella caespitosa* has a highly adapted method of dispersal, spreading propagules via large numbers of wind dispersed seeds as well as reproducing vegetatively via stolons, rhizomes, and adventitious roots that can be spread by water (Grosskopf & Cortat, 2016).

Although boreal forest regions have comparatively low proportions of non-native species to other regions, riparian areas are frequently invaded (our data; Hendrickson et al., 2005; Rose & Hermanutz, 2004; Pysek & Prach, 1993). Riparian zones are efficient conduits for the dispersal of invading species across landscapes (Nilsson et al., 2013), and their widespread nature within the boreal region can cause them to act as corridors of non-native species introduction into natural areas (Pysek & Prach, 1993). In fact, in much of the temperate region, non-native species already make up a considerable proportion of riparian flora (Nilsson et al., 2013).

Our ordination suggests that the forest roads assessed had a distinctly different non-native species composition than adjacent stream banks. Many of the species that were observed on roads were species adapted to drier substrate (e.g., *Plantago major*), which are not water tolerant. Species that are not water adapted would not be commonly found along moist stream banks due to the inability to successfully establish. Species that were observed on roads which have adaptations for wet conditions, such as *P. caespitosa* (Bishop & Davy, 1994) and *R. repens* (Harper, 1957), would be expected to invade stream banks. This was observed within our stream sites, as both *R. repens* and *P. caespitosa* were observed on 73% and 87% of our forest roads, respectively, as well as along 90% and 70% of stream banks, respectively. The narrow area seen in our ordination signifies a smaller variation in species composition on roadways, which may be influenced by the higher species richness observed along all road sites. Although the species richness was high along most roads, the species observed there were similar among sites. Both upstream and downstream disturbances had wider ordination areas, signifying more variation in species composition between sites. This may have been influenced by the variation in species richness that was observed between sites, where some sites had only one non-native species along its bank, with others having many species present.

The flood zone region of the stream bank had a higher total number of non-native species observations than either the submerged or upland bank zones. The influence of disturbance plays a large role in the establishment of flora within the flood zone, as it experiences the most disturbance due to seasonal freezing and flooding. Plants that are established in this region would generally have similar traits to many invasive species; high fecundity, spatial growth, and resource use ability (Catford & Jansson, 2014). These species, which are able to reproduce and grow quickly, can quickly take advantage of resources that become available after disturbance, dominating the flood zone region of stream banks (i.e., *R. repens*). The low level of non-native species established within the upland zone and adjacent forest area, relative to the flood zone area may be influenced by a lower propagule pressure in combination with a lower disturbance frequency in these areas (Brown & Peet, 2003). Propagules travelling along the stream current may be deposited along banks by flooding and generally may not reach the upland zone (Catford & Jansson, 2014). The submerged bank zone can support only species that are water-adapted, therefore, limiting the variety of species that is observed. Many of the species that we observed disperse via wind (e.g., *Pilosella caespitosa* or *Taraxicum officinalis*) or by vegetative reproduction and clonal growth (e.g., *Ranunculus repens*). These methods of dispersal are effective along stream corridors, which may act as channels for wind currents along with water.

2.4.3 Boreal forest dynamics in relation to non-native species establishment

Our forest sites showed very few occurrences of non-native species within intact forest stands adjacent to linear disturbances. The boreal forest is a system that has a natural resistance to species that are not adapted to its conditions due to limiting factors such as light availability, soil pH, and soil organic layer thickness. Although, when resources are not limiting in natural areas non-native species are able to establish, i.e., when soil pH is favourable, canopy is opened by natural disturbances, or when mineral soil is exposed (Rose and Hermanutz, 2004). This is a contrast to anthropogenic disturbances within the boreal forest, where there is generally a steep increase in resource availability that invasive, non-native plants may be more adapted to exploit (Leffler et al., 2014; Burke & Grime, 1996). The association of non-native species with potentially higher resource sites may be because many invasive, non-native plant species tend to generally share similar character traits, such as high fecundity, spatial growth, and resource use (van Kleunen et al., 2010), i.e., *Centaurea nigra* and *Fallopia japonica*. In

Alaska, two non-native *Melilotus* sp. (sweet clover) were found to have established in natural flood plain areas from roadside populations due to favourable soil pH conditions (Conn et al., 2008). There are native ruderal species on the island able to colonize disturbed ground rapidly, i.e., *Solidago* spp., but they may compete with introduced species for available resources (Cooper, 1981). Although we found few non-native species within forest stands adjacent to linear disturbances on the Avalon Peninsula, this was not the case in Gros Morne National Park, on the west coast of Newfoundland (Rose and Hermanutz, 2004), where *Ranunculus repens*, *Tussilago farfara*, *Taraxicum officinale*, *Hieracium* spp., *Cirsium arvense*, and *Digitalis purpurea* were found to be established. Humber and Hermanutz (2011) observed *Cirsium arvense* growing in natural and anthropogenic forest gaps many kilometres from the nearest road. They found that *C. arvense* can negatively impact regenerative stages of native *Abies balsamea* (L.) due to changes that the plant may cause to soil properties and microsite conditions. These impacts were magnified when grouped with impacts from non-native ungulate, *Alces alces* L.

Although few non-native species were found to be established in intact forest stands of eastern Newfoundland, these stands are fragmented by a matrix of linear disturbances where we have found that non-native species establishment is pervasive. Non-native species are known to establish and propagate rapidly along forest margins as well as within forest gaps (Charbonneau & Fahrig, 2004; Geldenhuys, 2004) and fragmentation of the boreal forest continues to increase, due to anthropogenic activities. Forest landscapes that are surrounded by a large amount of open area receive a larger number of non-native propagules, due to higher numbers of non-native species in the surrounding area (Charbonneau &Fahrig, 2004). Irregular shape of fragmented areas, along with smaller forest fragment size, can result in a higher proportion of non-native species that readily establish in these areas (Ewers and Didham, 2007). Impacts that change the natural composition of the boreal forest may lower the resistance of these forest patches in the future. Warming caused by climate change could benefit non-native species that are within temperate climate zones (Nilsson et al., 2013). Natural and anthropogenic disturbances cause an increase in resource availability that non-native species may be able to use more efficiently than some native species (Leffler et al., 2014).

2.4.4 Conclusions

Newfoundland has a long history of non-native species introduction and establishment. European colonization on the island, increased farming in the 19th century, the construction of the Newfoundland Railway, and acceleration of suburbanization during the latter half of the 20th century have all played a part in driving the pattern of non-native plant establishment that we see on the island today (Cooper, 1981). We have shown that non-native species are pervasive along the open canopy (i.e., high-light availability) anthropogenic and natural linear disturbances that surround intact patches of boreal forest; however, there are few examples of invasion into the adjacent intact forest ecosystem within this study. Fluctuations in resource availability in forested areas due to natural or anthropogenic disturbances and ongoing propagule pressure from non-native species in open canopy areas surrounding these forest sites are expected, thus, non-native species introduction and establishment within newly disturbed areas is anticipated.

Therefore, the stage is set for future non-native species establishment within forest stands. A disturbance that changes intact patch conditions (e.g., increased light availability, exposed mineral soil) will weaken the boreal forest's natural resistance to non-native species establishment, and may allow non-native species to maintain populations within these areas, as has been observed in other studies on the island of Newfoundland (Humber and Hermanutz, 2011; Rose and Hermanutz, 2004).

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Chapter 3: Summary and conclusions

3.1 Summary and discussion

3.1.1 Summary of findings

Increased globalization, i.e., the global movement of goods and people, has a great influence on the spread and introduction of non-native plant species. As the rate of transport continues to increase, so will the introduction of novel species into natural landscapes (Seebens et al., 2015; McGeoch et al., 2010). In the boreal forest, it is clear that there is a connection between establishment of non-native plant species and the magnitude of disturbance of an area (Langor et al., 2014; Kohli, 2009; Rose & Hermanutz, 2004; Haeussler et al., 2002). Although the boreal forest has a natural resistance to novel species, there are many examples of non-native plants establishing within these areas by means of disturbance. In many cases, non-native species are able to impact these natural areas by altering nutrient cycling, fire regimes, and hydrology as well as outcompeting native species (Latombe et al., 2017; Sanderson et al., 2012; McGeoch et al., 2010; Kohli, 2009; Lockwood et al., 2007; Vitousek et al., 1997). This is why boreal forest landscapes form an ideal system for research on these introduced species.

This thesis examines anthropogenic linear disturbances as corridors of invasion for non-native plant species. Anthropogenic corridors not only fragment natural landscapes but are also a profound cause of non-native species dispersal into these natural areas (Arevalo et al., 2010). Human traffic along these corridors as well as natural means of dispersal of these species, i.e., via wind, water, animals, etc., may allow them to spread into adjacent natural areas, accelerating non-native species establishment (Arevalo et al., 2010; Allen et al., 2009). Natural linear disturbances, i.e., riparian areas, have a higher vulnerability to non-native species establishment due to the presence of a natural disturbance regime (Stohlgren et al., 1999; Pysek & Prach, 1993). Anthropogenic linear disturbances that are adjacent to natural disturbances increase the dispersal and establishment of non-native plant species along these natural areas (Catford & Jansson, 2014).

Within the boreal forest region, non-native plant species introduction and establishment is influenced by means of disturbance more than any other means (Langor et al., 2014; Kohli, 2009; Rose and Hermanutz, 2004; Williamson, 1996). A larger magnitude of disturbance is the result of disturbances that have a higher severity. These areas tend to more readily provide the resources that non-native plants species may be more adapted to exploit (Leffler et al., 2014; Burke and Grime, 1996). Openings in the forest canopy due to disturbances, such as trails and roads, increase the light availability in dense, low-light boreal forests (Cole, 2004). Higher magnitudes of disturbance also tend to have more exposed soil bed, reduced species competition, as well as more space for propagules to be introduced and establish a population (Cole, 2004). The usage of anthropogenic linear disturbances also contributes to their disturbance magnitude. Increase travel along these corridors influence the propagule pressure of non-native species (Pickering & Hill, 2007; Rooney, 2005). Linear disturbances that are used for various forms of transportation, such as vehicular transport and foot traffic, may be exposed to a higher number of non-native plant propagules. These propagules increase the probability of non-native species establishment. We found that, of the linear disturbances we examined, forest roads contained the highest non-native species richness and experienced the largest representation of various traffic types. Areas that are used for foot traffic alone, i.e., walking trails, will tend to have a lower propagule pressure and ground layer trampling than areas that are used for multi-vehicular traffic, i.e., forest roads. This was observed throughout our 2018 forest survey sites, where low magnitude linear disturbances tended to have higher canopy cover and lower amounts of exposed mineral soil and space, and, therefore, lower non-native species presence. Since higher magnitudes of disturbance are more vulnerable to non-native plant species, these areas may have an increase likeliness of invasive species presence. For instance, *Centaurea nigra* (black knapweed) was only observed within five of our thirty 2018 sites, all five of which were forest roads sites.

Anthropogenic linear disturbances allow for non-native species to disperse and establish into natural areas. Propagules that are introduced to a novel area are more likely to have dispersed there from nearby populations (Lockwood et al., 2005). Naturally disturbed areas are especially vulnerable to establishment because of the increased resource availability provided by natural disturbance regimes. Along stream banks, non-native plants were observed at a higher abundance within the flood zone of the bank. This zone has the highest magnitude of disturbance compared to both the submerged and upland bank zones due to seasonal flooding and freezing. While non-native species observations within relatively undisturbed forest adjacent to linear disturbances were low, these species were present there. Although, in general these areas were dominated by native species adapted to the resource-limited conditions of the boreal forest (Daehler, 2003).

There have been numerous studies showing non-native plants tend to establish within anthropogenically and naturally disturbed areas of the boreal region on Newfoundland (Charron and Hermanutz, 2015; Trip and Wiersma, 2015; Humber and Hermanutz, 2011; Hendrickson et al., 2005; Rose and Hermanutz, 2004). This study further supports this hypothesis as well as supporting the idea these disturbances can act as corridors for the establishment of non-native species within natural areas of the boreal forest. This pattern is a cause for concern, as the quantity and intensity of anthropogenic disturbances is continuing to increase within the boreal zone of Canada (Gauthier et al., 2015; Langor et al., 2014; Bradshaw et al., 2009).

3.1.2 Vascular plant species and a changing climate

Temperature and atmospheric carbon dioxide (CO_2) will increase within the boreal zone due to global climate change; these alterations will impact the diversity, population size, and distribution of both native and non-native plants (Langor et al., 2014; Smith et al., 2012; Buckland et al., 2001). Plant distributions as well as functional groups are both influenced by the annual minimum temperature of an area (Woodward & Williams, 1987). In fact, with the changing climate there is potential for both native and non-native species to establish beyond their current range of distribution and dominate areas, even those areas that are undisturbed (Ricciardi et al., 2017; Caplat et al., 2013; Walther et al., 2009; Buckland et al., 2001). In Newfoundland, the daily minimum and maximum temperatures are projected to increase by 3 – 4 °C and 1 – 3 °C, respectively, with the greatest changes projected in winter. A projected increase in growing degree days for the island will result in a greater potential for plant growth (Finnis, 2013). Milder winters and warmer, longer summers are likely to increase the survival of annual weeds, i.e., quick growing and propagating species (Peters et al., 2014), and higher average temperatures may accelerate their movement into higher latitudes (Patterson, 1995). Nilsson et al. (2013) state that boreal riparian areas are especially vulnerable to non-native species establishment due to climate change. They predict that the species richness and the abundance of non-native species will increase in riparian areas due to the longer growing season and shorter winters expected from climate change. These areas are conduits of dispersal of species across landscapes, therefore, it is likely that non-native species that are found in riparian zones will disperse into the surrounding landscape (Nilsson et al., 2013). Variations in precipitation patterns as well as aridity caused by climate change could also alter the distribution and impact of non-native plant species (Ramesh et al., 2017). Non-native species will continue to respond favourably to climate change, likely exacerbating the ecological and economic problems they cause (Willis et al., 2010).

In addition to affecting temperature and growing seasons directly, climate change may also influence non-native species in the boreal forest by indirectly affecting disturbance regimes (Spellman et al., 2014). Rising temperatures, aridity, and changing weather systems are causing an increase in fire severity and frequency in the boreal forest that is expected to continue (Spellman et al, 2014). Wildfire, specifically severe cases, can increase the susceptibility of a natural area to non-native plant invasions by increasing resource availability, i.e., exposing mineral soil, opening canopy, and reducing competition (Walker et al., 2017; Spellman et al., 2014). Available propagules must be present in these post-fire areas in order for non-native species to take advantage of these resources; this has been observed in forested areas of Alaska (Walker et al., 2017; Spellman et al., 2014). Climate change is also altering insect disturbance regimes, causing range expansion and increases in outbreak severity in North America (Pureswaran et al., 2015). This can cause large-scale changes to the boreal forests across Newfoundland, as black spruce forests can be replaced with more productive mixed-wood forests or less productive ericaceous shrub growth due to changes in regeneration patterns and nutrient cycling caused by alterations in insect disturbance regimes (Pureswaran et al., 2015).

The concentration of carbon dioxide (CO_2) in the air will directly affect plants (Kirkham, 2011), and these effects will be different based on the type of photosynthesis that plants use. In all plants, photosynthesis involves the C_3 process that converts CO_2 into phosphoglyceric acid, a three-carbon molecule compound. But, in some species, a C_4 process occurs first where CO_2 is converted into oxaloacetic acid, a four-carbon molecule compound (Kirkham, 2011). It is difficult to predict whether a species will benefit from elevated CO_2 without studying the species and community it inhabits directly (Dukes, 2000). In general, most C_3 plants respond favourably to increased atmospheric CO_2 (Ramesh et al., 2017; Patterson, 1995), while C_4 plants are not as predictable. There are two categories of C_3 species which respond to elevated atmospheric CO_2 more strongly, those are, i) fast-growing species and ii) nitrogen fixing species (Dukes, 2000). Non-native plants are expected to respond with greater growth rates than native plants

(Meyerson & Mooney, 2007); in fact, in-lab experiments have shown that many invasive plant species have a positive response to increased CO_2 (Duke & Mooney, 1999). Although interactions in natural ecosystems may be more complicated, making it difficult to predict which individual species will flourish under elevated CO_2 conditions.

The associated effects of projected temperature changes and elevated CO_2 due to climate change display a strong need to flag non-native species with potentially high environmental impacts for cost effective management in the future (Kumschick et al., 2015). Not only will climate change affect plant growth and distribution directly, but it will also affect it indirectly by altering ecosystem processes, soil nutrients, and moisture. Range changes will alter competition, predation, as well as other critical species interactions (Dukes et al., 2009). Effects most likely will occur at different levels, causing a lag response between biotic interactions of above and below ground herbivores, pathogens, symbiotic mutualists and decomposer organisms and abiotic interactions with nutrients and moisture (Van der Putten et al., 2010).

3.2 Study considerations and limitations

While this research provides important findings on the association of anthropogenic disturbances and the establishment of non-native species, there are some considerations to be discussed. The main limitation of this research was temporal, as individual sites were measured during a single growing season. Some non-native species are quick growing, annual species that can go through their life-cycle in a short amount of time, such as *Matricaria discoidea* (pineappleweed) or *Fallopia convolvulus* (black bindweed). Many species are perennials which may have variable blooming times based on the plant species life cycle as well as the climatic conditions of the area. For instance, *Linaria vulgaris* (butter and eggs) and *Hypericum perforatum* (common St. John's-wort) only bloom between the months of July and August, while *Anthriscus sylvestris* (wild chervil) has an early blooming season from late May to early July. This may have limited the species observed within each site, as no more than 2 days were spent at each site location and plant surveys lasted from mid-July to late September, specifically in the 2018 field season. Many non-native species no longer flower as late as September, such as *Potentilla simplex* (old field cinquefoil) and *pilosella caespitosa* (meadow hawkweed). Long term monitoring of sites would be interesting in these cases to assess annual variation of non-native species present, as well as any increase or decrease in species richness over time, as annual climate variations can influence the non-native species observed in certain areas. Also, we did not consider the timing of the establishment of anthropogenic linear disturbances in this study. It is likely that many of these areas were developed at different times, influencing the establishment of non-native species populations present along them. The actual usage of each anthropogenic linear disturbance was also not quantified in this study, although magnitude of disturbance was measured based on width and type of disturbance, this may not give insight to the overall traffic usage of these disturbances. For example, perhaps some of the disturbances experience a higher seasonal level of traffic than others, impacting the magnitude of disturbance as well as the overall propagule pressure of these areas.

Although we measured non-native plant species along linear disturbances, there is no way for us to know for certain that propagules originated from these areas, although we can infer that close areas would supply the most pressure within natural environments. Many of the locations that we surveyed were in close proximity to a human population, where non-native species are generally commonly found. These areas may have influenced the non-native species richness we observed in more ways than the total kilometres of surrounding road that we measured.

We could not consider the effects of forest characteristics on non-native plant establishment due to the lack on non-native species that were observed within forested areas away from disturbance. While the lack of non-native species in forest stands is a positive observation, it meant we could not achieve one of the original intentions of this study; although, the variation of forests could influence the presence of non-native species in other areas. The lack of non-native species in forest stands may be influenced by the presence of alternative disturbances and dispersal vectors, e.g., *Alces alces* (moose) trails throughout the forest and browsing disturbance. These were observed by Humber and Hermanutz (2011) and Rose and Hermanutz (2004) and may explain why non-native species were found within forested areas on the west coast of the island, but not in forested areas of our study on the east coast. Future studies should include a larger sample of areas across the island to allow for different influences of ecoregions as well as the differences between boreal forest communities.

3.3 Recommendations for the management of non-native species on the island of Newfoundland

The boreal forest is a key economic resource, with estimated value of CAD $37.5 billion across all products extracted annually in Canada (Sanderson et al., 2012), and generates an estimated 40% of Canada's timber harvesting (Langor et al., 2014). The boreal forest also provides many critical ecosystem services, such as global carbon storage and biodiversity (Langor et al., 2014; Krawchuk et al., 2012; Sanderson et al., 2012). Invasion of the boreal forest by non-native species could cause detrimental effects on the Canadian economy; i.e., wood supply, and the environment; i.e., forest degradation, native species loss, alteration of disturbance regimes, and loss of conservation value of protected areas (Langor et al., 2014). Since the boreal forest region is expected to be particularly vulnerable to invasion under climate change, the assessment and prediction of non-native species impacts is essential for the allocation of resources by policy-makers who decide on investments for the management of these species (Smith et al., 2012).

Canada is obligated to prevent introduction of, control or eradicate invasive species based on being a signatory to the Convention on Biodiversity (Langor et al., 2014). Screening of potentially invasive species is a necessary first step in the prevention of invasion. Preventing the entry of non-native species is generally considered the most effective way to minimize the risk of invasive species (Wagner et al., 2017; Sheley & Smith, 2012; McGeoch et al., 2010), this can be accomplished by quarantining them, as many invasions begin with a small number of individuals (Mack et al., 2000). For instance, in Australia non-native species are stopped at the border by means of the

Australian Weed Risk Assessment system which was implemented to reduce the high economic costs and environmental damage associated with invasive plant species (Weber et al., 2009); this system has also been adapted for use by other countries, such as Canada (Auld, 2012). The cost of this would be much lower than the cost and effort that is needed to attempt to control non-native species after establishment. Creating quarantine zones for invasive species was ineffective on the island of Newfoundland, where one was created in 1980 to slow the spread of *Gremmeniella abietina* (European scleroderris canker), but due to lack of sufficient enforcement and public knowledge, it now threatens native red pine populations (Langor et al., 2014).

Public education is an important tool in slowing the colonization of non-native species (Langor et al., 2014), but it is a challenge to communicate knowledge gained from research so that it can be efficiently influence policy and management (Sheley et al., 2010). Issues that may be common within academia may not have the same exposure beyond it. In many cases, the information gained from research in academia is often diverse and conflicting, causing a significant delay between research and public response (Smith et al., 2012). Many provinces in Canada thus have an invasive species council to aid with communicating this information between the scientific community and members of the public. In Newfoundland and Labrador, the Botanical Garden of Memorial University of Newfoundland currently disseminates information regarding invasive species.

When invasive species are introduced, early detection can be useful if species are found before establishment or very soon after establishment and rapid response is important to mitigate impacts and minimize the risk of spreading (Sheley & Smith, 2012;

McGeoch et al., 2010). The issue with this method is that it relies on the widespread monitoring of natural areas, which is not always possible. In many cases, non-native species are ignored until they are widespread and invasive and therefore deemed to be worth the expense of eradication, but at this point eradication is most likely not an option (Blossey et al., 2001). This may be because of the time lag between species introductions and invasion, in many cases there is a long period of time between these two events, creating an issue with rallying people in support of non-native species control (Mack et al., 2000). In order to successfully eradicate a species', there needs to be sufficient resources over time, widespread support from agencies and the public, as well as some biological aspect of the species that can be targeted (Mack et al., 2000).

If a species cannot be eradicated, then restoration and maintenance is the logical next step (Sheley & Smith, 2012), with three main methods of maintenance: chemical, mechanical, and biological (Wagner et al., 2017; Langor et al., 2014; Mack et al., 2000). Chemical maintenance is a common method of management that involves the use of herbicides to reduce or eliminate invasive plants (Curtis & Bidart, 2017). The use of chemical herbicides can cause health hazards to humans and can have negative impacts on native species and ecosystems (Curtis & Bidart, 2017; Mack et al., 2000). In Canada, herbicides are commonly used as a control method for non-native invasive species within public wildland areas, although data is not tracked by agencies so little is known regarding the magnitude of use, effectiveness, and financial costs (Wagner et al., 2017). More data is needed to narrow the knowledge gap of herbicide usage in Canada and to determine if herbicides are an effective management tool (Wagner et al., 2017). Mechanical maintenance is the physical removal and destruction of non-native species

(Langor et al., 2014); it can be successful in some cases, such as setting hunting regulations for invasive mammals (Mack et al., 2000). Biological control involves the introduction of an invasive species' natural enemy to control invasive species populations (Ehler, 1998); optimally these two populations will control each other so that neither is able to cause damage, but occasionally these introduced species can have negative impacts on native species populations (Blossey et al., 2001). Because of this, the biological method requires extensive testing before implementation (Blossey et al., 2001).

Management, control, and prevention of invasive species should be considered from a long-term, large-scale, system management approach, rather than the approach of focussing on individual species invasions (Sheley et al., 2010; Mack et al., 2000). Forest management needs to be considered. Charron and Hermanutz (2015) discuss the importance of active restoration of disturbed boreal forest areas of Terra Nova National Park, Newfoundland, for the preservation of ecosystem services. This may also mitigate invasive species establishment within these areas, since areas that that have lower disturbance tend to be less invaded than areas that have higher levels of disturbance (Rose & Hermanutz, 2004; Daehler, 2003; Haeussler et al., 2002).

3.4 Conclusion

Non-native species establishment within boreal forest regions of eastern Newfoundland is linked to anthropogenic disturbances within these areas. We found that non-native species are pervasive along both anthropogenic linear disturbances as well as adjacent natural linear disturbances within the Avalon Peninsula of the island. The closed canopy and low resource availability that is observed within boreal forest stands is likely a major limiting factor to the establishment of non-native species. However, instances of disturbance within these regions can increase resource availability and expose the forest to non-native species introduction and establishment. Within the Avalon Peninsula of Newfoundland, a matrix of linear disturbances fragment forest stands, resulting in a plethora of non-native propagules that are capable of dispersing into natural areas. If these propagules are introduced in newly disturbed areas of the forest they may be well adapted to take advantage of these increased resources. This may be further exacerbated by the influence of climate change in this region, which is not only projected to increase growth of plant species, but could also alter the relationship that native and naturalized non-natives have in present local ecosystems. This could cause plants that are not currently invasive to become so in the future.

3.5 References

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Appendix I: Plant identification references

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Appendix II: Statistical models

Table AII.1 Correlation table of forest data variables collected from intact forest sites adjacent to anthropogenic linear disturbances during 2018. Avalon Peninsula, Newfoundland and Labrador, Canada (*'road' variable is a measure of total kilometres of road in a 2 km area surrounding each site).

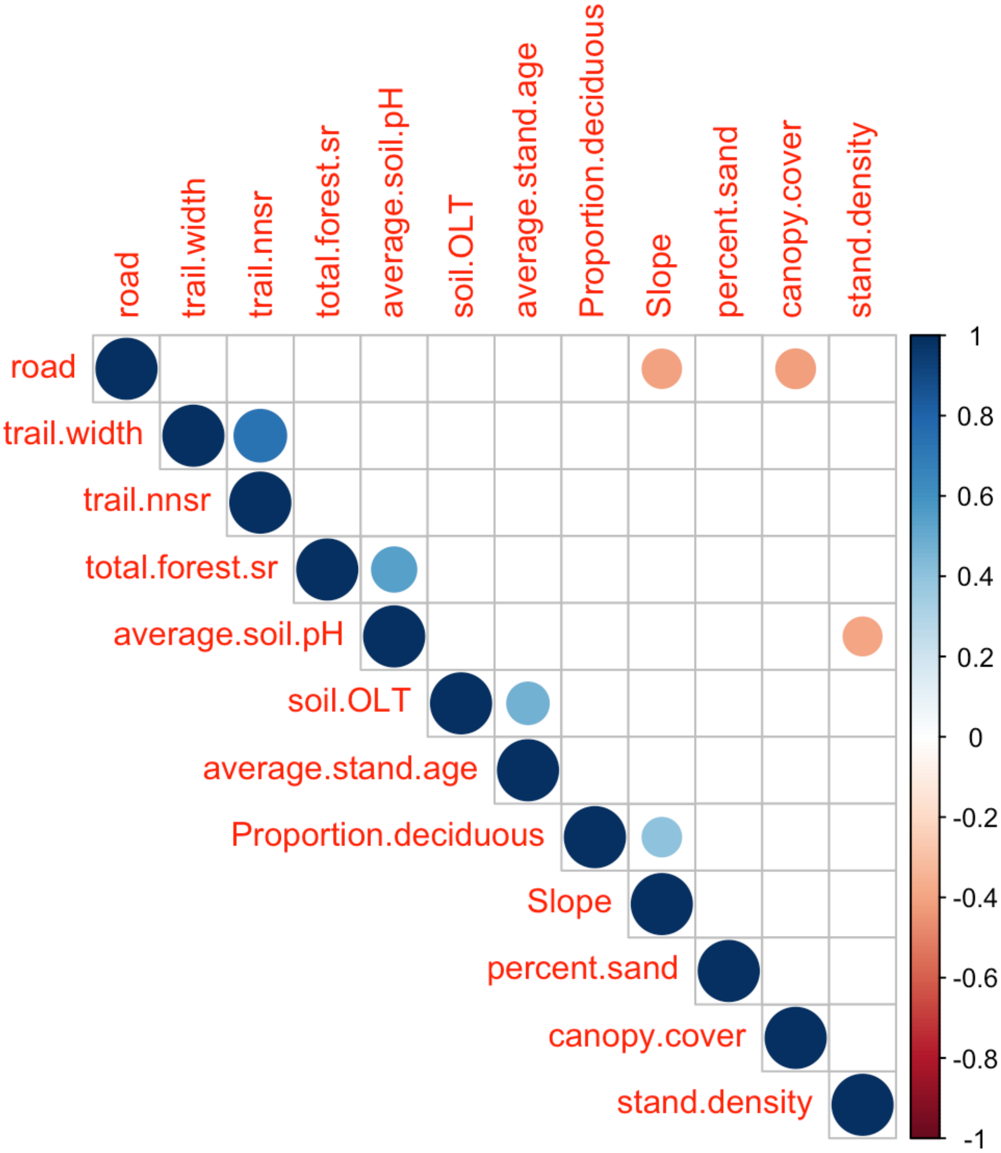


Table AII.2 General linear models for non-native species richness on anthropogenic linear disturbances and native species richness of adjacent intact forest. Text bolded for models with significant (p>0.05) values. Collected in 2018 on the Avalon Peninsula, Newfoundland and Labrador, Canada.

Variable	**Trail non-native species richness**		**Forest native species richness**	
	p	*t-ratio*	*p*	*t-ratio*
Trail width (m)	**3.37×10^{-6}**	**5.774**	0.637	0.478
Total kilometres of road (km)	0.0933	1.738	0.0836	1.794
Canopy cover	-	-	0.195	-1.328
Soil organic layer thickness (cm)	-	-	0.513	0.663
Mineral soil sand composition (%)	-	-	0.354	-0.942
Forest stand density	-	-	0.129	-1.563
Proportion of deciduous tree species	-	-	0.159	1.448
Average stand age (years)	-	-	0.337	0.978
Average soil pH	-	-	**0.00271**	**3.289**
Slope (%)	-	-	0.469	-0.733

Table AII.3 ANOVA comparing the ecoregion and region* of 2018 site locations to non-native species richness of anthropogenic linear disturbances and native species richness of adjacent intact forest. Avalon Peninsula, Newfoundland and Labrador, Canada. (*Region category groups southern and northern maritime barrens ecoregions into one region)

Variable	**Trail non-native species richness**		**Forest native species richness**	
	p	*f-ratio*	*p*	*f-ratio*
Ecoregion	0.704	0.356	0.556	0.355
Region*	0.228	1.562	0.631	0.236

Table AII.4 General linear models for non-native species richness of forest roads and stream banks as well as non-native species abundance of stream banks. Data collected from stream survey sites in 2019 on the Avalon Peninsula, Newfoundland and Labrador, Canada. Text bolded for significant (p>0.06) models.

Variables	**Non-native species richness on roads**		**Stream non-native species richness**		**Stream non-native species abundance**	
	p	*t-ratio*	*p*	*t-ratio*	*p*	*t-ratio*
Stream width (cm)	-	-	0.326	0.999	0.536	0.026
Flow space (cm)	-	-	0.231	1.226	0.955	0.057
Road width (cm)	0.454	0.760	0.402	0.851	**0.058**	**1.977**
Total kilometres of road (km)	0.131	1.555	**0.00799**	**2.856**	0.240	1.201
Stream non-native species abundance	0.19	1.342	**$1.53\text{x}10^{-5}$**	**5.217**	-	-
Non-native species richness on roads	-	-	0.3229	1.006	0.190	1.342

Table AII.5 ANOVA comparing dominant stream sediment type to non-native species richness and abundance on stream banks. Data collected from stream sites in 2019 on the Avalon Peninsula, Newfoundland and Labrador, Canada.

	Stream non-native species richness		**Stream non-native species abundance**	
	p	*f-ratio*	*p*	*f-ratio*
Dominant sediment type (gravel/cobble)	**0.0534**	**4.067**	**0.0216**	**5.925**

Table AII.6 ANOVA comparing non-native species abundance on stream banks to the area of the bank in which they were observed. Data collected in 2019 from stream survey sites on the Avalon Peninsula, Newfoundland and Labrador, Canada.

	Stream non-native species abundance	
	p	*f-ratio*
Bank zone	0.00069	7.929

Appendix III: Non-native species observed

Table AIII.1 A compiled list of all non-native plant species observed at both forest (2018) and stream (2019) survey sites, including the number and location of each species observation. Avalon Peninsula, Newfoundland and Labrador, Canada.

Non-native plant species		***Forest site observations***				***Stream site observations***			**Total observations across sites**
Latin name	**Common name**	**Walking trail**	**ATV trail**	**Forest road**	**Total observations**	**Forest road**	**Stream bank**	**Total observations**	
Acer pseudoplatanus	Sycamore maple	-	-	-	**0**	-	1	**1**	**1**
Achillea millefolium	Common yarrow	-	-	2	**2**	9	3	**22**	**24**
Agrostis canina	Velvety bentgrass	4	1	4	**9**	7	-	**7**	**16**
Agrostis capillaris	Common bentgrass	3	6	2	**11**	2	-	**2**	**13**
Agrostis gigantica	Redtop bentgrass	-	-	-	**-**	1	-	**1**	**1**
Alchemilla sp.	Lady's mantle	-	-	-	**-**	2	1	**3**	**3**
Anthoxanthum odoratum	Sweet vernal grass	1	3	4	**8**	4	1	**25**	**33**
Anthriscus sylvestris	Cow parsley	-	-	-	**-**	1	-	**1**	**1**
Arctium minus	Common burdock	-	-	1	**1**	-	-	**-**	**1**
Artemisia vulgaris	Common mugwort	-	-	1	**1**	2	1	**3**	**4**
Betula pendula	Weeping birch	-	-	1	**1**	-	-	**-**	**1**
Carex ovalis	Oval sedge	-	1	-	**1**	-	-	**-**	**1**
Centaurea nigra	Black Knapweed	-	-	4	**4**	2	2	**34**	**38**
Cerastium fontanum	Common mouse-ear chickweed	-	-	-	**-**	3	1	**4**	**4**
Chenopodium album	Pigweed	1	-	-	**1**	-	-	**-**	**1**
Danthonia decumbens	Heath grass	-	-	1	**1**	-	-	**-**	**1**
Elymus repens	Couch grass	-	-	1	**1**	-	-	**-**	**1**
Fallopia convolvulus	Black bindweed	-	-	-	**-**	-	1	**1**	**1**
Fallopia japonica	Japanese knotweed	-	-	-	**-**	1	-	**1**	**1**
Festuca filiformis	Fine leaf sheep's fescue	1	2	4	**7**	8	1	**9**	**16**
Glyceria maxima	Great manna grass	-	1	-	**1**	-	-	**-**	**1**
Hieracium lachenalii	Common hawkweed	2	5	9	**16**	1	9	**40**	**56**

Hypericum perforatum	St. John's wort	-	-	2	**2**	3	-	**3**	**5**
Jacobaea vulgaris	Common ragweed	-	-	-	**-**	2	3	**5**	**5**
Juncus conglomeratus	Compact rush	1	-	-	**1**	-	-	**-**	**1**
Non-native plant species		***Forest site observations***				***Stream site observations***			**Total observation across sites**
Latin name	**Common name**	**Walking trail**	**ATV Trail**	**Forest road**	**Total observations**	**Forest road**	**Stream bank**	**Total observations**	
Leontodon autumnalis	Fall dandelion	1	-	1	**2**	7	-	**7**	**9**
Leucanthemum vulgaris	Oxeye daisy	-	-	2	**2**	3	4	**17**	**19**
Linaria repens	Pale toadflax	-	-	-	**-**	1	-	**1**	**1**
Linaria vulgaris	Yellow toadflax	-	-	3	**3**	-	-	**-**	**3**
Lotus corniculatus	Bird's-foot trefoil	-	-	-	**-**	3	1	**4**	**4**
Lupinus polyphyllus	Large- leaved lupin	-	-	-	**-**	1	-	**1**	**1**
Malus pumila	Common apple	-	-	-	**-**	-	2	**2**	**2**
Matricaria discoidea	Pineappleweed	-	-	-	**-**	2	-	**2**	**2**
Myosotis arvensis	Rough forget-me-not	-	1	-	**1**	-	-	**-**	**1**
Myosotis scorpioides	European forget-me-not	-	-	-	**-**	-	4	**4**	**4**
Myosotis sylvatica	Woodland forget-me-not	-	1	-	**1**	-	-	**-**	**1**
Odontites vulgaris	Red bartsia	-	-	1	**1**	-	-	**-**	**1**
Oenothera perennis	Small evening primrose	-	-	-	**-**	-	1	**1**	**1**
Persicaria hydropiper	Marshpepper smartweed	-	1	-	**1**	-	-	**-**	**1**
Pilosella aurantica	Orange hawkweed	-	-	2	**2**	8	4	**12**	**14**
Pilosella caespitosa	Meadow hawkweed	-	-	-	**-**	6	1	**47**	**47**
Pilosella officinarum	Mouse-ear hawkweed	-	-	-	**-**	9	-	**9**	**9**
Plantago major	Common plantain	-	3	5	**8**	7	-	**17**	**25**
Poa palustris	Fowl blue grass	-	-	-	**-**	1	1	**12**	**12**
Poa pratensis	Kentucky bluegrass	-	-	-	**-**	1	-	**1**	**1**
Potentilla simplex	Common cinquefoil	1	-	-	**1**	2	1	**3**	**4**
Ranunculus acris	Tall buttercup	-	3	1	**4**	0	-	**10**	**14**
Ranunculus repens	Creeping buttercup	-	4	2	**6**	2	7	**49**	**55**
Rosa rubiginosa	Sweetbrier rose	-	3	-	**3**	-	-	**-**	**3**
Rumex acetosella	Sheep's sorrel	-	-	2	**2**	6	-	**16**	**18**
Rumex crispus	Curly dock	-	-	-	**-**	1	2	**3**	**3**
Sonchus arvensis	Field milk thistle	-	1	-	**1**	-	-	**-**	**1**
Spergula arvensis	Corn spurrey	-	-	-	**-**	1	-	**1**	**1**
Stachys palustris	Marsh hedge nettle	1	-	-	**1**	-	-	**-**	**1**
Taraxacum officinale	Common dandelion	-	-	3	**3**	3	7	**40**	**43**

Trifolium aureum	Yellow clover	-	-	-	**-**	1	-	**1**	**1**
Trifolium pratense	Red clover	-	-	2	**2**	3	7	**30**	**32**
Trifolium repens	White clover	-	1	4	**5**	9	2	**21**	**26**
Non-native plant species		***Forest site observations***				***Stream site observations***			**Total observations across sites**
Latin name	**Common name**	**Walking trail**	**ATV trail**	**Forest road**	**Total observations**	**Forest road**	**Stream bank**	**Total observations**	
Tussilago farfara	Coltsfoot	-	-	-	**-**	1	1	**2**	**2**
Urtica dioica	Common nettle	-	-	-	**-**	2	-	**2**	**2**
Veronica officinalis	Common speedwell	-	3	6	**9**	4	4	**18**	**27**
Veronica serpyllifolia	Thyme-leaved speedwell	1	1	1	**3**	2	1	**3**	**6**
Vicia cracca	Tufted vetch	-	-	-	**-**	7	2	**9**	**9**

Appendix IV: Native species observed at forest sites

Table AIV.1. All native species observed within 2018 forest survey sites adjacent to anthropogenic linear disturbances. Avalon Peninsula, Newfoundland and Labrador, Canada.

Latin name	Common name
Abies balsamea	Balsam fir
Acer spicatum	Mountain maple
Agrostis mertnsii	Northern bentgrass
Alnus viridis subsp. *crispa*	American green alder
Arctostaphylos uva-ursi	Common bearberry
Aronia x prunifolia	Purple chokeberry
Betula papyrifera	Paper birch
Carex billingsii	Billing’s sedge
Carex deweyana	Dewey’s sedge
Carex echinata	Star sedge
Carex foenea	Straw sedge
Carex lasiocarpa	Wiregrass sedge
Carex nigra	Black sedge
Carex recta	Estuary sedge
Carex stipata	Prickly sedge
Carex trisperma	Three-seeded sedge
Chamerion angustifolium	Fireweed
Chelone glabra	White turtlehead
Cinna latifolia	Drooping woodreed
Circaea alpina	Small enchanter’s-nightshade
Clintonia borealis	Corn lily
Cornus canadensis	Bunchberry
Cornus stolonifera	Red-osier dogwood
Empetrum nigrum	Black crowberry
Galium palustre	Marsh bedstraw
Gaultheria hispidula	Creeping snowberry
Gaylussacia bigeloviana	Dwarf huckleberry
Glyceria canadensis	Rattlesnake manna grass
Hieracium umbellatum	Narrow-leaved hawkweed
Iris versicolor	Blue flag
Juncus articulatus	Jointed rush
Juncus effuses	Common rush
Kalmia angustifolia	Sheep laurel
Larix laricina	Eastern larch
Linnaea borealis	Twinflower
Maianthemum canadense	Wild lily-of-the-valley
Moneses unifloria	One-flowered wintergreen
Monotropa uniflora	Ghostflower
Myrica gale	Sweet gale
Orthilia secunda	One-sided wintergreen
Picea glauca	White spruce

Latin name	**Common name**
Picea mariana	Black spruce
Prunus pensylvanica	Pin cherry
Prunus virginiana	Chokecherry
Rhododendron groenlandicum	Common Labrador tea
Ribes glandulosum	Skunk currant
Rubus idaeus	Wild red raspberry
Rubus pubescens	Dewberry
Rumex acetosella	Sheep sorrel
Solidago macrophylla	Large-leaved goldenrod
Solidago rugosa	Rough-stemmed goldenrod
Sorbus americana	American mountain-ash
Sorbus decora	Showy mountain-ash
Symphyotrichum novi-belgii	New York aster
Taxus canadensis	Canada yew
Trientalis borealis	Northern starflower
Vaccinium angustifolium	Early lowbush blueberry
Vaccinium macrocarpon	Large cranberry
Vaccinium oxycoccus	Small cranberry
Viburnam nudam var. *cassinoides*	Northern wild raisin
Viburnum opulus subsp. *trilobum*	Highbush cranberry

Appendix V: Stream characteristics

Table AV.1. Stream characteristics of 30 stream survey sites collected in 2019 on the Avalon Peninsula, Newfoundland and Labrador, Canada.

Site	Average stream width (cm)	Culvert flow space (cm)	Sediment type
ss01	384.5	405	Cobblestone & boulders
ss02	528.5	500	Gravel & mud
ss03	403.5	420	Cobblestone
ss04	184	164	Gravel
ss05	710	550	Cobblestone & boulders
ss06	214	420	Cobblestone & boulders
ss07	834	1240	Cobblestone & boulders
ss08	179	290	Cobblestone
ss09	330.5	280	Gravel
ss10	782.5	520	Gravel
ss11	567.5	430	Cobblestone & boulders
ss12	1350	1600	Cobblestone & boulders
ss13	153	60	Gravel & mud
ss14	320	320	Cobblestone
ss15	698	460	Cobblestone & bedrock
ss16	345	250	Gravel & bedrock
ss17	1410	2170	Cobblestone
ss18	506	510	Cobblestone
ss19	262.5	220	Gravel
ss20	615	530	Cobblestone
ss21	382.5	450	Gravel
ss22	442.5	400	Cobblestone & bedrock
ss23	971.5	610	Cobblestone
ss24	835	540	Gravel
ss25	926.5	850	Gravel & bedrock
ss26	716	550	Gravel & bedrock
ss27	251	180	Gravel
ss28	1223.5	980	Cobblestone
ss29	907.5	470	Cobblestone & bedrock
ss30	302.5	320	Cobblestone

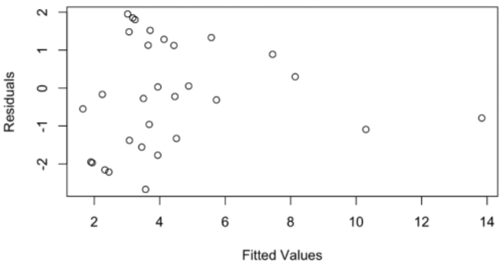
Appendix VI: Outcome of Generalized linear model

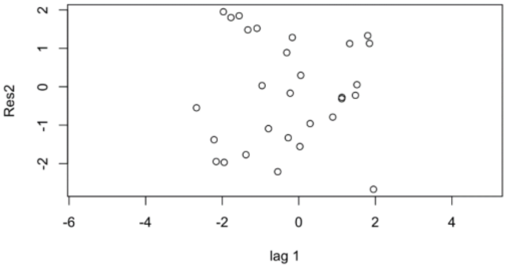
AVI.1. A generalized linear model comparing the effects of i) surrounding kilometres of road, ii) average trail width, and iii) average stand age as well as combined effects of iv) surrounding kilometres of road with average trail width and v) surrounding kilometres of road with average forest stand age on non-native species richness of linear disturbances of 30 forest sites (2018).

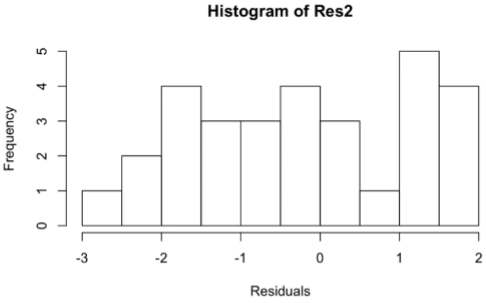
Results: Not significant

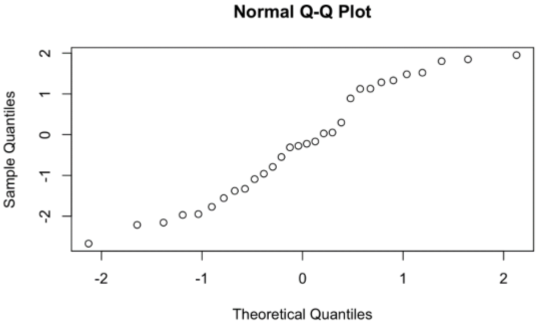
```
Call:
glm(formula = trail.nnsr ~ road + trail.width + average.stand.age + (road * trail.width) + (road *
average.stand.age) +
        (trail.width * average.stand.age), family = poisson, data = Sitedata)

Deviance Residuals:
         Min        1Q        Median      3Q        Max
        -2.6715   -1.3674    -0.2497   1.1257    1.9515

Coefficients:
                                        Estimate    Std. Error     z value   Pr(>|z|)
        (Intercept)                   -0.3200885   1.0413813    -0.307     0.759
        road                            0.2353413   0.2583471     0.911      0.362
        trail.width                    0.4453024   0.3303560      1.348     0.178
        average.stand.age             0.0076031   0.0150069      0.507    0.612
        road:trail.width              -0.0371637   0.0432996     -0.858     0.391
        road:average.stand.age      -0.0016003  0.0034869      -0.459    0.646
```

```
        trail.width:average.stand.age      0.0002375   0.0073918      0.032      0.974


(Dispersion parameter for poisson family taken to be 1)


Null deviance: 94.698  on 29  degrees of freedom

Residual deviance: 58.678  on 23  degrees of freedom

AIC: 155.49


Number of Fisher Scoring iterations: 6
```





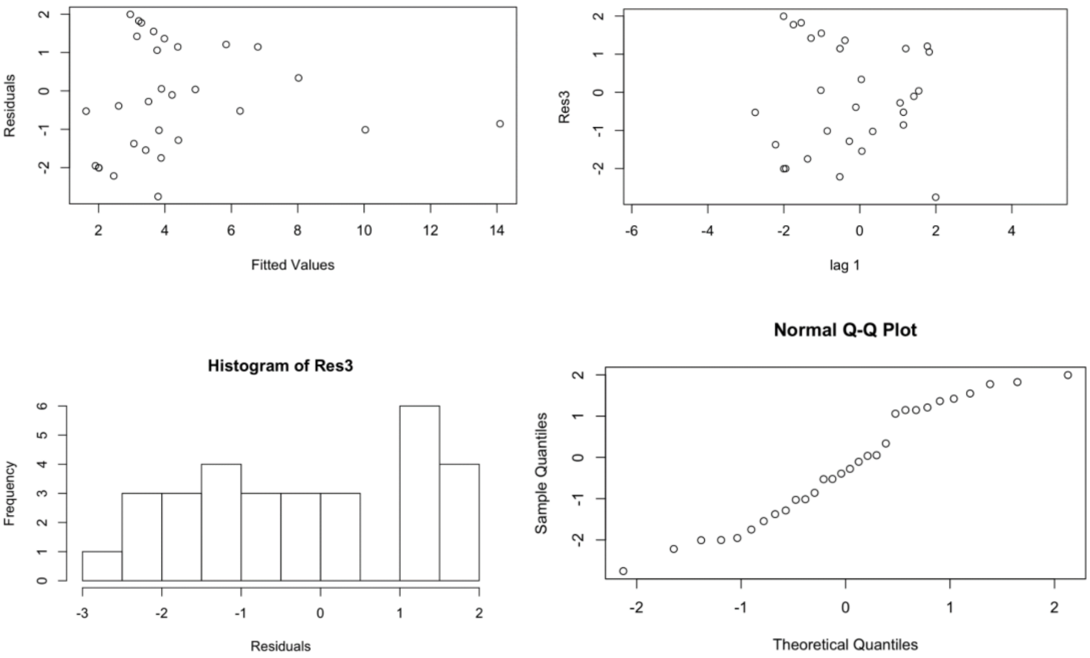




AVI.2. Elimination of the surrounding kilometres of road and average stand age interaction term

Results: Showed significance but plots show an over dispersal of data

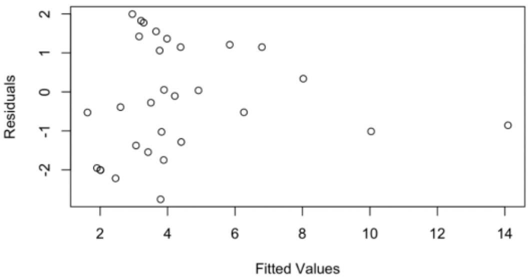
```
Call:
glm(formula = trail.nnsr ~ road + trail.width + average.stand.age + (road * trail.width), family = poisson, data =
Sitedata)

Deviance Residuals:
       Min       1Q          Median       3Q      Max
    -2.7541  -1.3525  -0.3343   1.1469   1.9953

Coefficients:
                              Estimate       Std. Error      z value    Pr(>|z|)
    (Intercept)               1.526e-01      6.260e-01       0.244      0.8074
    road                      1.374e-01      9.608e-02       1.430      0.1526
    trail.width               4.213e-01      1.746e-01       2.413      0.0158 *
    average.stand.age        -6.428e-05      6.735e-03      -0.010      0.9924
    road:trail.width         -3.043e-02      3.304e-02      -0.921      0.3571
    ---
    Signif. codes:  0 ‘***’  0.001 ‘**’  0.01 ‘*’  0.05 ‘.’  0.1 ‘ ’ 1

(Dispersion parameter for poisson family taken to be 1)

 Null deviance: 94.698  on 29  degrees of freedom
Residual deviance: 59.011  on 25  degrees of freedom
AIC: 151.82

Number of Fisher Scoring iterations: 5
```

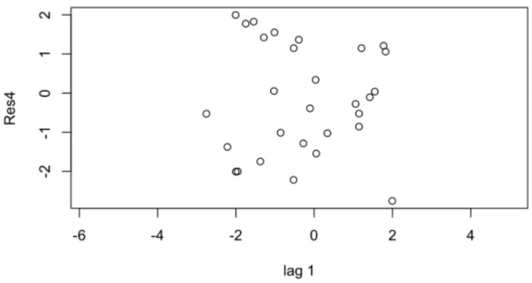


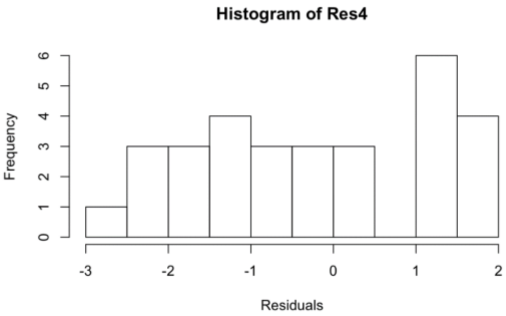
AVI.3. Use of same formula, Quassi-Poisson was completed to attempt to fix over dispersal of data seen in above Poisson

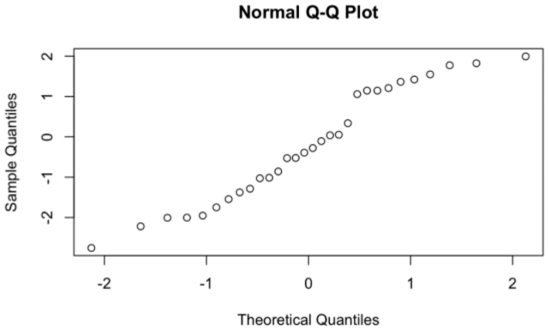
```
Call:

glm(formula = trail.nnsr ~ road + trail.width + average.stand.age + (road * trail.width), family = quasipoisson,

data = Sitedata)


Deviance Residuals:

        Min          1Q       Median      3Q      Max

    -2.7541    -1.3525    -0.3343    1.1469   1.9953


Coefficients:

                                 Estimate     Std. Error     t value      Pr(>|t|)
```

```
(Intercept)              1.526e-01     8.872e-01    0.172        0.865

road                       1.374e-01     1.362e-01    1.009        0.323

trail.width              4.213e-01     2.475e-01    1.703        0.101

average.stand.age    -6.428e-05     9.546e-03   -0.007       0.995

road:trail.width       -3.043e-02     4.683e-02    -0.650       0.522


(Dispersion parameter for quasipoisson family taken to be 2.00878)


Null deviance: 94.698  on 29  degrees of freedom

Residual deviance: 59.011  on 25  degrees of freedom

AIC: NA


Number of Fisher Scoring iterations: 5
```









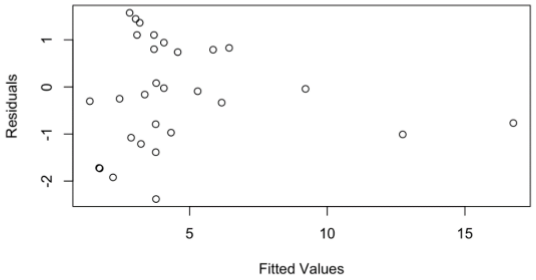
AVI.4. Quassi-Poisson was not effective. A negative binomial distribution was attempted using the same formula above. Negative binomial distribution had the best fit for our data.

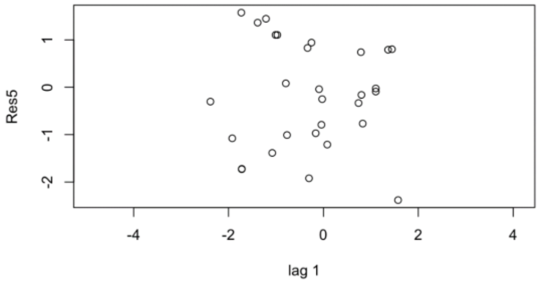
```
Call:

glm.nb(formula = trail.nnsr ~ road + trail.width + average.stand.age + (road * trail.width), data = Sitedata,

init.theta = 5.153777641, link = log)


Deviance Residuals:

            Min      1Q   Median      3Q      Max

        -2.3817  -1.0612  -0.2065   0.7988   1.5738


Coefficients:

                              Estimate      Std. Error      z value      Pr(>|z|)

        (Intercept)          -0.170342      0.842414       -0.202        0.8398

        road                  0.169637      0.134124        1.265        0.2060

        trail.width           0.551054      0.253783        2.171        0.0299 *

        average.stand.age     0.000839      0.008970        0.094        0.9255

        road:trail.width     -0.046058      0.049038       -0.939        0.3476

        ---

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


(Dispersion parameter for Negative Binomial(5.1538) family taken to be 1)


Null deviance: 59.193  on 29  degrees of freedom

Residual deviance: 38.570  on 25  degrees of freedom

AIC: 150.2


Number of Fisher Scoring iterations: 1
```

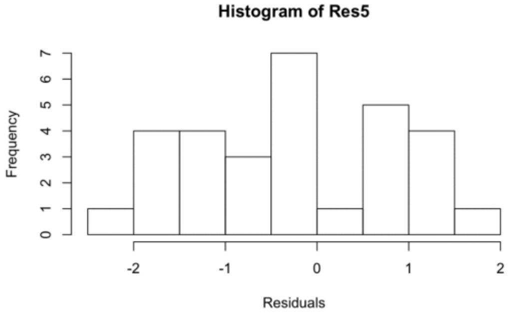
Theta: 5.15

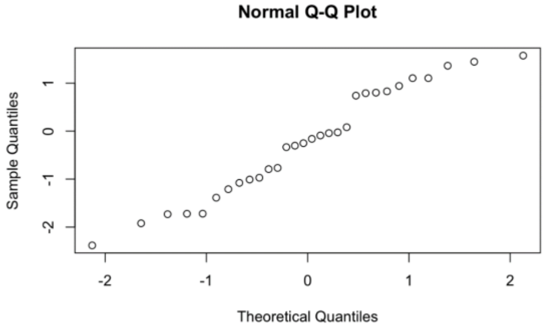
Std. Err.: 3.64

2 x log-likelihood: -138.195






Histogram of Res5


Normal Q-Q Plot