# LIVING WITH LINEAR FEATURES: IMPACT OF ROADS, RECREATIONAL TRAILS AND TRANSMISSION LINE RIGHTS-OF-WAY ON SMALL MAMMALS IN NEWFOUNDLAND

KARLA RAE LETTO

# LIVING WITH LINEAR FEATURES:
# IMPACT OF ROADS, RECREATIONAL TRAILS AND TRANSMISSION LINE RIGHTS-OF-WAY ON SMALL MAMMALS IN NEWFOUNDLAND

By

© Karla Rae Letto

A thesis submitted to the
School of Graduate Studies
in partial fulfilment of the
requirements for the degree of
Master of Science

Department of Biology
Memorial University

August 2013

St. John`s Newfoundland and Labrador

# ABSTRACT

Linear features have the capacity to degrade landscapes and impact wildlife. I used mark-recapture methods to examine the relationship between small mammals and linear features (roads, trails and transmission lines) in boreal forest and barren ecosystems on the island of Newfoundland, Canada. I live-trapped 25 *Microtus pennsylvanicus* and 314 *Sorex cinereus* at increasing distances from linear features over a total of 3600 trap nights. Relative abundance of these species did not differ between trapping distances. However, the relationship between linear features and abundance differed between species, favouring the introduced habitat generalist *S. cinereus* and disturbing the native habitat specialist *M. pennsylvanicus*. PCA suggests that microhabitat and food availability are important determinants of population density of these species. This study is the first to examine the relationship between Newfoundland's small mammals and anthropogenic linear features on the landscape, making the findings an important contribution to the management planning for the environmental impacts of linear features.

**Keywords:** linear feature; *Microtus pennsylvanicus*; Newfoundland; recreational ecology; road ecology; small mammal; *Sorex cinereus*

# ACKNOWLEDGEMENTS

I extend my sincere gratitude to the following people and organizations for helping to make this thesis possible.

I would like to begin by thanking my supervisor, Dr. Yolanda Wiersma, for giving me the amazing opportunity to conduct this research in her lab, providing guidance and constant encouragement throughout the duration of this project, and always supporting my personal and academic decisions. Yolanda has always encouraged me to pursue new opportunities and to break out of my "shell". She has been an amazing mentor and I owe a large part of my future career as a landscape ecologist to her.

I also thank my two committee members, Ted Miller and Bruce Rodrigues for their time, thoughts and technical contributions to this thesis.

I acknowledge my field assistant, Sheldon Kallio, for his hard work during my field season. I don't think I would have been able to find another field assistant willing to spend 40 mornings bush-whacking through alders at 4:00 AM in St. John's in near-constant drizzle without a single word of complaint. I also thank the Women in Science and Engineering program of Memorial University for providing the funding to hire Megan Haines as an additional field assistant. It was a pleasure to work with Megan and I trust that she will become a great scientist one day.

I also acknowledge the Institute for Biodiversity, Ecosystem Science and Sustainability and Natural Sciences and Engineering Research Council for financial support.

As well, I thank Pippy Park for allowing me to conduct this research within the park. It was a pleasure to work there. I thank the patrons of Pippy Park for respecting my research sites, participating in my intensity of trail use survey, and showing interest in maintaining the ecological integrity of the Three Pond Barrens trail system.

I thank John Morrissey of the Department of Transportation and Works for providing me with traffic data for the Trans Canada Highway and Dr. Ben Bolker for taking the time to reply to my plea for help with mixed-effects modeling on the online R-sig-mixed-models forum. As well, I acknowledge Rudy Boonstra (University of Toronto) for providing me with the supplies necessary to make small mammal tracking spools, and Greg Robertson (Environment Canada) for providing me with a MARK tutorial. I was not able to incorporate these latter components in my thesis, but the assistance was greatly appreciated.

I extend a huge thank you to my friends and lab mates for their support, thoughts and technical help. Particularly, I acknowledge Sheena Roul, Brittany Baril, Victoria Howse,

Stacey Camus and Patricia Howse for their assistance in mapping the trails in Pippy Park, and the members of the LESA lab for their assistance in data analysis.

I extend a big thank you to my mom and sister for their love and support over the last few years and always showing interest in what I was studying (even when they could only describe it as "studying rats").

Finally, I acknowledge my boyfriend, Mark Abbott, for his constant encouragement, love and moral support throughout this degree. I would like to thank Mark for always reading and editing my thesis when I needed another opinion, allowing me to turn our deep-freeze into a storage area for shrew specimens, trekking through a marsh in the early hours with plastic bags on his feet as waterproof footwear, and for putting up with all my complaints when things got tough. I could not have done this without him.

## Table of Contents

## List of Tables

## List of Figures

## List of Appendices

# CHAPTER 1. INTRODUCTION AND LITERATURE REVIEW

## 1.1. Introduction

Industrial development over the past two centuries has led to an increase in humanity's requirements for electric power transmission and transportation of people and goods. As a result, the expansion of human-made linear features such as roads and transmission lines has increased dramatically (Forman et al. 2003; Phillips 2010). Furthermore, with the rise of environmentalism in the mid 19th century came the proliferation of recreational trails that allowed easy access to once-undisturbed habitat by recreationists (Liddle 1997; Jensen and Guthrie 2006). Roads, railways, recreational trails and electric utility corridors ("linear features" hereafter), are all forms of human-made linear infrastructure that as a society we have become dependent upon. Unfortunately, these linear features can have numerous direct and indirect ecological effects on natural ecosystems including habitat loss and fragmentation, reduced habitat quality, spread of invasive or exotic species, spread of infectious diseases and wildlife disturbance (Liddle 1997; Forman et al. 2003).

Linear features may act as dispersal routes for some animals while blocking movement for others (Fuentes-Montemayor et al. 2009). Grassy roadsides and other linear features are used as movement corridors by some animals, such as small mammals (Kuykendall and Keller 2011) and feral predators, thus increasing the potential for gene flow in a fragmented landscape (Balkenhol and Waits 2009). They also act as dispersal routes by aiding in the spread of infectious diseases (Jules et al. 2002; Urban 2006) and exotic species (Tyser and Worley 1992; Forman 2000; Brown et al. 2006). However, linear features also have the potential to act as barriers to animal movement. Through behavioural avoidance or traffic mortality (when the linear feature is associated with traffic) linear features may decrease movement, thus reducing gene flow (Forman et al. 2003; Shepard et al. 2008; Balkenhol and Waits 2009) and increasing genetic structure (i.e., the distribution of genetic variation) and decreasing genetic diversity (i.e., the amount of genetic variation) within the population (Balkenhol and Waits 2009). Linear features that are perceived by an animal as a barrier also block the animal's access to resources, such as food and mates. Ultimately, these linear features result in lower habitat quality, individual fitness and population viability (Shepard et al. 2008; Balkenhol and Waits 2009).

Anthropogenic linear features also result in direct habitat loss, which invariably is much greater than the linear feature itself. Linear features create an abrupt transition between two different habitat types, one of which is usually a hostile environment (e.g., paved highway). This creates a noticeable edge (often referred to as road-effect zone when adjacent to a transportation route) where noticeable environmental effects extend outward from the linear feature. There have been numerous estimates for how far the road-effect zone extends into the habitat interior, and it has been estimated to be as high as several hundred metres (Forman and Deblinger 2000). Flora and fauna vary in their response to edge effects. For some, the alteration in light, moisture or noise renders the edge unsuitable (Dyer et al. 2001; Forman et al. 2002). However, other species, particularly small mammals, are able to thrive in edge habitat (Bissonette and Rosa 2009; Ascensão et al. 2012).

Whether a linear feature facilitates or inhibits animal movement, or creates a noticeable edge effect depends on numerous factors such as type of linear feature (Gerlach and Musolf 2000; Dyer et al. 2002), traffic intensity (Clevenger et al. 2001), width of linear feature (Rico et al. 2007) and habitat (Grilo et al. 2009). Furthermore, an animal's response to a linear feature is species-specific and dependent on certain life-history traits, such as home range size or reproductive rate (Forman et al. 2003; Shepard et al. 2008). Therefore, a linear feature that is a barrier to one species may be a dispersal corridor for another. Similarly, an edge created by a linear feature may degrade the habitat for one species and improve habitat quality for another. Canada has approximately 900 000 km of roads (Government of Canada 2012) and more than 278 000 km of managed trails (Norman 2010). With the increasing number of linear features in today's landscapes it has become important to increase our understanding of the impacts that these linear boundaries have on long-term viability of adjacent wildlife populations (van der Ree et al. 2011).

This literature review will first outline what is currently known about the impacts of linear infrastructure on wildlife (excluding small mammals) from the peer-reviewed literature. This will be done by describing trends observed in select invertebrate, avian, anuran, reptilian and other mammalian groups for each examined type of linear feature (i.e., roads, trails and powerline rights-of-way). I will then describe the importance of small mammals and why small mammals are a good choice of study species to examine the ecological impacts of linear features on the island of Newfoundland, Newfoundland and Labrador (hereafter "Newfoundland"). This will be followed by a review of similar studies examining the interaction between small mammals and the three different types of linear features in other geographic locations. The chapter will conclude with sections specific to Newfoundland, including a description of Newfoundland's terrestrial mammalian community and food web dynamics and description of the two study species.

## 1.2. Impacts of Linear Features on Wildlife

### *1.2.1. Roads*

The discipline of road ecology, which uses ecology and landscape ecology to study the interactions of roads and traffic with their surrounding environment, was developed because of the need to understand how roads affect nature and the environment in order to mitigate negative outcomes (Forman et al. 2003). The majority of research that has been conducted on linear infrastructure has focused on roads, and a large proportion of this has dealt with the interaction between roads and wildlife.

Vehicular traffic on roads can result in high rates of wildlife mortality that can greatly alter the demographics of surrounding populations (Row et al. 2007; Boves and Belthoff 2012). In extreme situations, the mortality rate due to road kills may even exceed natural mortality rates (Forman et al. 2003). If increased levels of mortality are not compensated for by higher birth rates, then population persistence may be affected, possibly leading to local extirpations (Jaeger et al. 2005; Boves and Belthoff 2012). Certain wildlife species are more vulnerable to road mortality than others. Those that are attracted to roads are at a greater risk than those that have developed behavioural avoidance. For example, snakes are attracted to road surfaces for thermoregulation (Rosen and Lowe 1994). Snakes and other cold-blooded animals bask on roads to increase their body temperature, putting them at a high risk for road-related mortality (Rosen and Lowe 1994, Row et al. 2007). Roads are also attractive habitats for scavengers. For example, scavenging raptors are attracted to roads due to high densities of road-kill food resources and perching sites. Scavengers feeding near roads, especially large raptors that are slow to take off, are vulnerable to vehicle collision (Lambertucci et al. 2009). In addition, certain life history traits make some species more vulnerable than others. For example, species with large home ranges or territories, or highly mobile species that make frequent long range movements over the landscape are also more susceptible to becoming road mortality victims because they interact with roads more often than less mobile species (Carr and Fahrig 2002; Rytwinski and Fahrig 2011; 2012). For example, Carr and Fahrig (2002) showed that highly mobile leopard frogs (*Rana pipiens*) experience population density declines near roads with much traffic, while the less mobile green frog (*Rana clamitans*) does not. On a population level, wildlife groups that are most susceptible to population declines due to traffic mortality are large animals with low reproductive rates and long generation times (Forman et al. 2003; Rytwinski and Fahrig 2011). Such species are less likely to rebound quickly following population declines (Rytwinski and Fahrig 2011).

Roads also affect wildlife by altering their behaviour. For example, a recent study on the effects of long-term exposure to traffic on bow-winged grasshoppers (*Chorthippus biguttulus*) showed that males from roadside habitats had altered their courtship signals by producing signals at a higher local frequency maxima compared to control populations. It is uncertain what effect this change could have on mate choice by females (Lampe et al. 2012). Other taxa, such as birds and anurans, have also altered their acoustic communication signals to avoid signal degradation or masking near roads (Parris and Schneider 2009; Parris et al. 2009).

However, the most reported wildlife behavioural alteration due to the presence of roads is behavioral avoidance. Animals may avoid roads because of avoidance of noise, the road surface or vehicles (Jaeger et al. 2005). For example, west European hedgehogs (*Erinaceus europaeus*; common and scientific names of mammals following Wilson & Reeder 2005) do not appear to be bothered by traffic noise since they often utilize road verges for foraging and as dispersal corridors. However, telemetry studies have shown that they avoid the road surface, possibly due to the pavement surface and different microclimate (Rondinini and Doncaster 2002). In contrast, certain bird species display noise avoidance behaviour and avoid roads with busy traffic and habitat in the road-effect zone (Jaeger et al. 2005). No matter the reason, animals that avoid roads suffer barrier effects. Barriers to animal movement cause isolation which may reduce gene flow and cause genetic effects such as inbreeding depression, genetic drift or reduced genetic variation and diversity (Forman et al. 2003; Shepard et al. 2008; Balkenhol and Waits 2009). In contrast to roads acting as barriers, they may also act as dispersal routes. For example, the rapid spread of the invasive cane toad (*Bufo marinus*) in Australia has been aided by roads (Brown et al. 2006).

Roads also impact wildlife by causing habitat loss. Habitat loss can be direct, where habitat is removed for road construction or indirect, where habitat near roads is reduced in such a way that it is rendered inhospitable by certain wildlife species (Jaeger et al. 2005). In Canada there are approximately 900 000 km of paved road (Government of Canada 2012) and in the United States there is approximately 3.65 million km (Forman et al. 2003). Therefore, in these two countries alone 4.55 million km of potential habitat has been replaced by pavement and road verges. Forman and Deblinger (2000) estimated that the road-effect zone along a suburban highway in Massachusetts extended outward 300 m from the road surface with the area affected being approximately 0.6-$km^2$ per kilometer of road length. Using this estimation, 2.73 million $km^2$ of habitat is impacted by road effects in Canada and the United States.

Besides direct habitat loss, roads can also reduce the quality of adjacent habitat. For example, Ortega and Capen (1999) examined the interaction between roads and Ovenbird (*Seiurus aurocapilla*) territory density and size in Vermont and found lower territory densities and larger territory sizes near roads compared with forest interiors. This suggests that road edges have a lower habitat quality for Ovenbirds than forest interiors. However, for some species, roads improve habitat quality. This is particularly true for edge specialists that are able to thrive in disturbed environments. For example, Laurance (2004) reported higher abundances of rainforest edge-specialist birds near roads than interior habitats. As well, the creation of unique road microhabitats, such as ditches for amphibians and bridges for bats, improves habitat quality for certain species (Forman et al. 2003).

### *1.2.2. Trails*

The increasing demand for outdoor recreation and ecotourism has led to an increase in the number of recreational trails (Liddle 1997; Jensen and Gutherie 2006). These trails are used for a variety of activities including hiking, dog walking, biking, snowmobiling and off-road vehicle (ORV) use. Because of this, managers are faced with the challenge of discerning how trail use and its various forms of recreation are affecting wildlife.

#### *1.2.2.1. Dog Walking*

Dog walking is one of the most popular forms of outdoor recreation, engaging millions of people and canines world-wide (Banks and Bryant 2007). Most studies that have described the relationship between dog walking and wildlife have reported negative effects. Domestic dogs have been shown to evoke physiological (e.g., changes in heart rate or stress hormone release) and behavioural (e.g., flushing, evasion or alteration in foraging and activity patterns) responses from various wildlife species (Steven et al. 2011; Lenth et al. 2008; Forrest and St. Clair 2006; Miller et al. 2001).

Domestic dogs, especially those that are not restrained by a leash, may chase wildlife. For the individual chased, this may result in increased energy expenditure, lost foraging and mating opportunities and the possibility of injury or death. Even without being chased, animals that are natural prey for wild canine species may perceive the domestic dog as a threat and display a negative fear-based response (Lenth et al. 2008).

Birds, especially ground-dwelling species, usually exhibit a negative response to dog-walking (Steven et al. 2011; Bank and Bryant 2007). Out of eleven papers reviewed by Steven et al. (2011) that addressed the effects of domestic dogs and dog-walking trails on birds, all eleven authors reported a negative effect.

Lenth et al. (2008) used various detection methods, including pellet and scat surveys, wildlife cameras, prairie dog burrow location mapping and track plates to measure and compare wildlife activity along trails that allowed dog-walking to those that prohibited dogs. The authors found that species that are natural prey for wild canine predators, such as mule deer (*Odocoileus hemionus*), rabbit (*Sylvilagus* spp.) and black-tailed prairie dog (*Cynomys ludovicianus*), had lower activity levels along trails that allowed dog-walking compared with those that prohibited dog-walking. It is hypothesized that these animals may perceive domestic dogs as a threat and therefore reduce activity to minimize chances of detection. A similar mule deer reaction was reported by Miller et al. (2001). In contrast to this prey pattern reported by Lenth et al (2008), most detected native carnivores were more active along trails that allowed dogs compared with areas that prohibited dogs. However, scat surveys from domestic dogs and native carnivores suggest that native carnivores mark areas around the periphery of trail heads where domestic dog scat is concentrated, suggesting an increase in vigilance by carnivores within their home range in response to the presence of domestic dogs (Lenth et al. 2008).

#### *1.2.2.2. Off-Road Vehicles (ORV)*

According to the Newfoundland and Labrador Provincial Parks Act, RSNL1990 cP-32, an off-road vehicle is defined as "a motorized vehicle designed for, or capable of, travel on unprepared surfaces including water, snow, ice, marsh, bog or swampland or on other natural terrain." This includes low-pressure tire vehicles, four-wheel drive vehicles, all-terrain vehicles, personal watercrafts, two-wheel motorized vehicles, snowmobiles, minibikes, amphibious machines and trail bikes. The Newfoundland and Labrador Motorized Snow Vehicles and All-Terrain Vehicles Act, RSNL 1990 cM-20, defines an all-terrain vehicle as "a wheeled or tracked motorized vehicle, excluding a 2 wheeled vehicle, designed or adapted for off-road use." For the purposes of this thesis, an ORV includes four-and three-wheel all-terrain vehicles, two-wheeled motorized vehicles (dirt bikes) and low-pressure tire vehicles designed for off-road use.

ORVs became popular as a form of recreation in the 1990s, and are now amongst the fastest growing forms of recreation (Jensen and Gutherie 2006; Davenport and Switalski 2006). Advances in ORV technology, along with more accurate positioning with GPS, has allowed ORV recreationist to traverse almost any type of landform and access even the most extreme remote wilderness areas (Davenport and Switalski 2006). This has led to widespread ecological problems that have become increasingly difficult to manage.

Similar to roads, ORV trails can have cascading effects on the ecosystem, causing serious impacts to soil, vegetation, wildlife and aquatic environments. For example, ORVs cause both soil compaction and erosion. Soil compaction, resulting from passing ORVs, increases soil bulk density and decreases soil porosity and infiltration capacity, causing increased rates of erosion. This lowers soil fertility, prevents seedlings from penetrating the soil and decreases plant nutrient uptake, altering plant community composition (Davenport and Switalski 2006). ORVs also alter vegetation by assisting the spread of non-native and invasive vegetation. For example, seeds adhering to the mud in ORV tire treads can be carried and deposited along trails (Liddle 1997). These impacts can change the entire vegetation community, in turn greatly altering the food and cover resources of various wildlife species (Bury et al. 1977). As well, soil erosion increases the rate of sedimentation in aquatic environments, increasing water turbidity, and lowering aquatic flora and fauna habitat quality. Such impacts on soil erosion can be detected after only one or two passes (Davenport and Switalski 2006; Van Vierssen Trip and Wiersma in review). Therefore, even areas with limited amount of ORV use can result in dramatic habitat modifications that affect vegetation, wildlife and aquatic ecosystems.

The most obvious effect of ORVs on wildlife is mortality from collisions (Davenport and Switalski 2006); burrowing animals are particularly vulnerable (Burge 1983; Knight and Cole 1991; Sheppard et al. 2009). ORVs can also have detrimental effects on burrowing animals by collapsing burrows (Burge 1983), compacting soil and thereby inhibiting the establishment of new burrows (Davenport and Switalski 2006) and direct mortality of animals within burrows (Burger et al. 2007). These impacts can decrease reproductive success (Burger et al. 2007) or cause population declines (Burge 1983). For example, pine snakes (*Pituophis melanoleucus*) nest in shallow underground burrows that can be crushed or exposed by ORVs (Burger et al. 2007). Burger et al. (2007) found that the reproductive success of pine snakes nesting in the pine forests of New Jersey decreased significantly during years of ORV use compared with years when ORVs were denied access. Another similar, yet more extreme example is the endangered desert tortoise (*Gopherus agassizii*) of the Mohave Desert whose severe population declines have been linked to ORV use. The desert tortoise relies on burrows and extensive tunnel systems to escape the intense desert heat. ORVs can crush burrowed tortoises or collapse their tunnels. This can cause suffocation of tortoises trapped within the tunnels and prevent those outside from seeking shelter from the sun (Burge 1983).

ORVs also allow hunters or poachers easier access to wildlife. ORVs can be used to penetrate deep into wildlands and easily transport large game animals (Davenport and Switalski 2006). The use of ORVs by hunters is legal in many areas (e.g., in Newfoundland and Labrador it is legal to use ORVs to access a hunting site or transport game from a hunting site, but illegal to chase or kill game from an ORV), there are also many incidences where they are used illegally. For example, there have been increased reports of poaching of large carnivores and ungulates, such as wolves (*Canis lupus*) near Glacier National Park, Montana (Boyd and Pletscher 1999) and mountain gazelle (*Gazella gazelle*) in the Ibex Reserve, Saudi Arabia (Attum 2007). As well, ORVs are sometime used to illegally chase and herd ungulates such as pronghorn (*Antilocapra americana*) and gazelle (Canfield et al. 1999; Attum 2007). This often results in overharvest, increase risk of injury while fleeing and increased levels of stress (Canfield et al. 1999).

Noise associated with ORVs also impact wildlife by increasing stress (Creel et al. 2002; Tull and Brussard 2007), altering movement patterns (Janis and Clark 2002; Preisler et al. 2006) and disrupting reproductive or foraging activities (Hayward et al. 2011; Ciuti et al. 2012). For example, Ciuti et al. (2012) found that elk (*Cervus elaphus*) altered their activity patterns by becoming more vigilant on public lands with ORV use compared with national parks with no ORV access. In these areas disturbed elk spent more time scanning for threats and less time foraging and grooming. As well, Barton and

Holmes (2007) found increased rates of songbird nest abandonment, and Hayward et al. (2011) report fewer fledged Northern Spotted Owl (*Strix occidentalis*) young, from nests within close proximity to ORV trails compared with more remote areas. Such behavioural changes can result in increased levels of energy expenditure and decreased reproductive success and fitness.

### *1.2.3. Powerline Rights-of-Way*

Unlike roads, powerline rights-of-way do not create an abrupt transition to an asphalt surface, nor are they associated with intense vehicular traffic like some major highways. Therefore, they do not appear to be as ecologically damaging as roads. However, powerline corridors create permanent linear openings bordered by prominent edges that define the transition between interior habitat and managed grass/shrub habitat. As well, utility roads used for powerline maintenance are commonly used as ORV trails, and are therefore susceptible to traffic effects. Therefore, like roads and ORV trails, powerline rights-of-way still have the potential to disrupt ecosystem functioning by creating barriers that fragment landscapes, prevent gene flow and lower genetic diversity, and promote the expansion of edge species while limiting the success of interior habitat specialist (Nekola 2012).

Tall vegetation along powerline corridors is a major safety concern for humans. Trees that are allowed to grow within reach of high voltage powerlines may result in wildfires and power outages. Traditional management methods of powerline corridor include complete clearing of all vegetation within the corridor over short rotation times. This high disturbance frequency maintains an early-successional grassland vegetation stage within the powerline corridor (Clarke et al. 2006; Russell et al. 2005). Over the past two decades, powerline right-of-way management has shifted to employing integrated vegetation management techniques that cause less ecological damage. These techniques focus on promoting habitat for wildlife while accounting for risks associated with high growing vegetation (Clarke and White 2008). Newfoundland Hydro, one of the two major suppliers of electricity in Newfoundland and Labrador, has a program to manage vegetation growth within powerline corridors. Management practices include tree trimming, brush removal and the application of herbicides. The herbicide product and conditions of use are regulated through Health Canada, Pesticide Management Regulatory Agency, and the Pest Control Products Act and Regulations (Nalcor Energy 2009).

How wildlife respond to the presence of a powerline largely depends on how the corridor is managed. Management techniques involving the complete removal of vegetation by clear-cutting, mowing and extensive herbicide spraying appear to be the most ecologically damaging (Russell et al. 2005; Clarke et al. 2006; Clarke and White 2008). Such practices create an abrupt edge or barrier between an early-successional, grassland corridor and its surrounding forest matrix. The powerline right-of-way, consisting mainly of short grasses, is less structurally complex, limiting the amount of functional habitat within the corridor. This can have negative effects on biodiversity, particularly for specialist species (Clarke et al. 2006). For example, Kroodsma (1982) found that bird community density in powerline corridors in east Tennessee was inversely correlated with percent grass coverage.

However, some studies have shown that this pattern can be beneficial for wildlife. For example, Nekola (2012) found that the creation of grassland habitat within powerline corridors has increased the overall gastropod biodiversity in Michigan. The powerline corridor has allowed for an expansion of lowland grassland specialist species into areas that were previously unoccupied, with little to no negative impact on the forest specialist species. Forest specialist gastropods, including species of conservation concern, were still able to persist within 30 m of the powerline edge.

More current management techniques, involving extraction of tall vegetation and selective herbicide spraying, foster a dense shrub community instead of a grassland community. Shrub communities are more structurally complex than grassland habitats, therefore offering numerous niches to support a diverse fauna community (Clarke et al. 2006). As well, these sites require less maintenance, and therefore less habitat disturbance, due to the tree-resistant nature of the resulting dense shrub community (Niering and Goodwin 1974; Russell et al. 2005). Therefore, this management technique is more cost-effective for power companies, and appears to be less damaging to wildlife. For example, a study examining how this management regime affects native bee habitat in Maryland compared with corridor mowing, found that powerline corridors managed this way supported a richer bee community with more rare, parasitic and cavity-nesting species than nearby grassland locations (Russell et al. 2005). Numerous other studies have shown that powerline rights-of-way managed by tall vegetation extraction and selective herbicide spraying provide foraging and nesting sites for early-successional shrubland bird species (Askins 1994; King and Byers 2002; Marshall and VanDruff 2002;

Confer et al. 2008). However, some authors have voiced the concern that powerline corridors are only enhancing habitat quality for opportunistic generalist species (edge species) and allow the invasion of introduced or exotic species at the cost of forest or interior specialist and native species (Goosem and Marsh 1997; Baker et al. 1998; Clarke et al. 2006).

Some authors suggest that even though powerline rights-of-way provide habitat for early-successional species, they also promote higher rates of predation. For example, birds of prey, such as Red-tailed Hawks (*Buteo jamaicensis*), are abundant along powerline corridors because powerline poles are good perching and nesting sites (Knight and Kawashima 1993). It has also been suggested that large-bodied mammalian carnivores, such as coyotes (*Canis latrans*) and bobcats (*Lynx rufus*), prefer powerline rights-of-way (Smith et al. 2008). Such species use the powerline corridor for easy travel and foraging opportunities (Gates 1991, Smith et al. 2008). In cases where powerlines intersect forested regions, this provides a connection between open habitat and interior forest habitat, serving as a conduit for open habitat carnivores to forest interior prey (Askins 1994). Similarly, Brown-headed Cowbirds (*Molothrus ater*) and other brood-parasitic birds display a preference for edge habitat adjacent to powerline corridors (Evans and Edward Gates 1997). King and Byers (2002) examined whether powerlines are source or sink habitats for shrubland birds by examining Chestnut-sided Warblers (*Setophaga pensylvanica*) inhabiting a powerline corridor in Massachusetts. They tested whether or not Chestnut-sided Warbler reproductive output within the corridor compensated for losses due to nest predation and brood parasitism. The authors found that more young were produced than lost from mortality, and that average daily nest survival rates were similar to extensive patches of remote shrubland habitat, suggesting that powerline corridors can operate as functional source habitats. In contrast, Chasko and Gates (1982) reported lower passerine nest success rates in nests adjacent to powerline corridors compared to interior forest nests.

### 1.3. The Importance of Small Mammals:

Small mammals have been recognized as important components of boreal forest ecosystems. Their burrowing behaviour mixes soil and helps decompose organic matter and litter (Pearce and Venier 2005). Herbivorous species consume various plants and plant products, fungi and lichen (Carey and Harrington 2001; Pearce and Venier 2005). This can have positive and negative effects on the vegetation community. Their feeding habits are important for disseminating seeds, spores and propagules of various species. For example, hypogeous mycorrhizal fungi, important symbiotic fungi for coniferous trees, are dependent on small mammals for spore dispersal (Maser et al. 1978; Terwillinger and Pastor 1999). Herbivores are also important for the regulation of nutrient dynamics, particularly nitrogen cycling (Sirotnak and Huntly 2000). Therefore, herbivorous small mammals are important for forest regeneration and sustainability. However, in high-abundance years, over-browsing by these small herbivores can be harmful for young forests (Kamler et al. 2011) and agriculture (Brown et al. 2007; Gebhardt et al. 2011). Insectivorous small mammals also play an important role as controls for various pests. For example, the cinereus shrew (*Sorex cinereus*; throughout this thesis study species including cinereus shrew and meadow vole (*Microtus pennsylvanicus*) will be referred to by their scientific names) was introduced in Newfoundland to control larch sawfly infestations and is presumably responsible for controlling this invertebrate pest (MacLeod 1960; Whitaker 2004).

Besides their role as consumers, small mammals also play an important role in the terrestrial food web as prey items. They are an important prey source for a diverse community of avian and mammalian carnivores. For example, in Newfoundland small mammals are the main food source for the threatened Newfoundland marten (*Martes americana atrata*) and vulnerable Short-eared Owl (*Asio flammeus*).

Small mammals are an ideal taxonomic group to study the impacts of disturbance. They have small home ranges; therefore any behavioural response to the disturbance will be detected in the local population dynamics (Steele et al. 1984). They have short generation times that allows for relatively fast detection of environmental change (Bissonette and Rosa 2009). Furthermore, they are particularly sensitive to habitat alterations and usually respond to these changes in a measurable way (Olsen and Brewer 2003). Their abundance and composition are strongly linked to vegetation and woody material structure and complexity (Carey and Harrington 2001; Forman et al. 2003; Olsen and Brewer 2003). Therefore, when a habitat's spatial heterogeneity is altered by natural or anthropogenic disturbance, the small mammal community is often affected. Because of this, small mammals have been used as indicator species to evaluate sustainable forest management methods (Carey and Harrington 2001; Pearce and Venier 2005) and to monitor effects of disturbance on ecosystem functioning (Steele et al. 1984). Lastly, small mammals are easily trapped for mark-recapture studies, making them an appealing choice for abundance studies. Therefore, small mammals, which are the focus of my thesis, are a good choice of study species to examine the ecological effects of linear infrastructure in Newfoundland.

## 1.4. Impacts of Linear Features on Small Mammals

### *1.4.1. Roads*

Studies examining the interaction between small mammals and roads can be divided into two main categories: those examining barrier effects and those examining edge effects.

Most studies examining small mammals and barrier effects have shown that roads are complete or partial barriers to small mammal movement (Gerlach and Musolf 2000; McDonald and St. Clair 2004; Rico et al. 2007; McGregor et al. 2008). As discussed above, animals may avoid roads because of noise avoidance, road surface avoidance or car avoidance (Jaeger et al. 2005). McGregor et al. (2008) used translocation studies on the white-footed deermouse (*Peromyscus leucopus*) and eastern chipmunk (*Tamias striatus*) to test each of the above road avoidance hypotheses and found that small mammals avoid crossing roads because of the road surface and its openness, as opposed to the traffic (gas emissions and noise) associated with them. Evidence supporting their conclusion included reduced small mammal movement across roads, a nonsignificant relationship between road crossing and traffic intensity and similar densities of small mammals near roads with increasing traffic intensity (McGregor et al. 2008). Gerlach and Musolf (2000) demonstrated genetic effects of road barriers on small mammals by showing a significant genetic subdivision of bank vole (*Myodes glareolus*) populations separated by a highway. Therefore, roads may have the potential to genetically separate small mammal populations.

Similar to McGregor et al. (2008), other studies have found that roadside edge habitats are able to support high abundances of small mammals (Adams and Geis 1983; Garland and Bradley 1984; Bissonette and Rosa 2009; Fahrig and Rytwinski 2009). For example, Bissonette and Rosa (2009) found that a highway in a desert ecosystem in southern Utah had no negative effects on small mammal abundance, density or diversity. In fact, they reported higher or similar abundances of 11 of their 13 captured small mammals in sites adjacent to the highway compared to sites further away. There are several hypotheses for why inflated small mammal abundances have been reported in road verge habitats. Small mammals have small movement ranges and territory sizes, high reproductive rates and avoid going on roads. Therefore, there is minimal traffic mortality and even though their movement may be blocked by a road, viable populations can still persist in patches bounded by roads (Fahrig and Rytwinski 2009). Further, several studies have demonstrated that small mammals are not disturbed by traffic noise (McGregor et al. 2008). Others suggest that small mammal populations are able to increase because of reduced predation due to the negative effect roads have on several carnivore species (Fahrig and Rytwinski 2009). Finally, road verges are usually maintained as a grassland environment that provides prime foraging habitat for small herbivores and can have  a structurally complex edge. Edge habitat structure, such as vertical stratification and spatial heterogeneity of vegetation, strongly influences small mammal populations and communities (Naxara et al. 2009).

However, even though many small mammal species thrive in road edge habitat, the response is usually species specific (Adams and Geis 1983; Rico et al. 2007; Bissonnette & Rosa 2009). For example, Adams and Geis (1983) found that small mammal density and species richness were greater in the road verge than in the adjacent forest habitat. However, five of their captured species did not prefer the verge. The species that preferred the verge were either grassland specialists or habitat generalists; forest specialists were less abundant or completely absent in these habitats.

Few studies have investigated relationship between roads or traffic volume on small mammal body mass or body condition. These variables are important life-history traits since they are both related to the animal's quality and health, representative of the animal's overall energy reserves and possibly an important determinant of fitness (Peig and Green 2010). Limited evidence suggests that small mammal body condition is not negatively affected by roads (Fuentes-Montemayor et al. 2009), and in some cases, such as the greater white-toothed shrew (*Crocidura russula*), may even increase body condition (Sabino-Marques and Mira 2011).

Recognizing that many small mammal species are able to inhabit road verge and edge habitat, several authors have described the importance of maintaining road verges as habitat for small mammals (Bellamy et al. 2000; Sabino-Marques and Mira 2011;

Ascensão et al. 2012). This is especially true in developed landscapes lacking suitable habitat.

### *1.4.2. Trails*

#### *1.4.2.1. Dog Walking*

Limited studies have addressed the effect domestic dogs and dog-walking trails have on small mammals. Lenth et al. (2008) used track plates to measure small mammal activity patterns along trails used by dog walkers. Within 5 m of trails, small mammals were significantly less active in areas that allowed dogs compared to areas that prohibited dogs, and within areas that allowed dogs, small mammal activity increased with distance from the trail (Lenth et al. 2008). Small mammals are natural prey for many wild dog species, such as coyotes. Therefore, similar to other wild dog prey species described earlier, small mammals may perceive domestic dogs as a potential predator and alter their behaviour accordingly. Small mammal response to domestic dogs appears to be similar regardless of whether or not the dog is leashed (Forrest and St. Clair 2006).

#### *1.4.2.2. Off-Road Vehicles*

One of the main concerns for small mammals with regards to ORV use is direct mortality through collision. Small mammal species are not easily seen by ORV drivers and are therefore easily crushed (Davenport and Switalski 2006). As well, fossorial small mammals are vulnerable to crushing when inside their burrows. For example, small mammals in dune environments have been crushed or buried within their collapsed burrows by passing ORVs (Luckenback and Bury 1983).

As mentioned above, soil compaction by ORVs increases soil bulk density and decreases the infiltration rate of water. This makes it difficult for seedlings to penetrate soil (Brooks 1995; Davenport and Switalski 2006). Small mammal population densities and species richness are often correlated with microhabitat variables such as plant primary productivity, density and diversity (Brooks 1995; Monamy and Fox 2000; Moro and Gadal 2007). Therefore, ORVs indirectly impact small mammals by altering the plant communities they depend upon (Brooks 1995). For example, by constructing a fence around the Desert Tortoise Research Natural Area in the Mojave Desert, California that excludes ORVs and sheep grazing, above ground annual plant biomass, percent cover of perennial shrubs, and seed biomass all increased. This resulted in a greater diversity and density of desert rodents inside the fenced area (Brooks 1995). Similarly, desert rodents inhabiting the Algodones Dunes, California were negatively affected by ORV use. The density, diversity and biomass of small mammals was lower in ORV impacted sites compared to control sites (Luckenback and Bury 1983) and was inversely related to ORV traffic intensity (Bury et al. 1977). Vegetation sampling and analysis supported the hypothesis that the observed impacts on small mammals were mainly attributed to changes in the volume of perennial vegetation, which affects both cover and food resources (Luckenback and Bury 1983).

Few studies have examined the impacts of ORV use on small mammals in non-desert ecosystems. Jeffery (2009) studied the impact of ORVs on vegetation and small mammals in pine and prairie habitats within Big Cypress National Preserve, Florida, and found that small mammal response to ORVs varied among species, with some having lower densities in ORV sites and others having higher densities, when compared with controls. Different species responses may be due to different home range requirements, habitat requirements, sensitivity to disturbance, or differences between generalist and specialist species (Jeffery 2009). Bogs have also been suggested as sensitive ecosystems. ORV use in bogs can result in soil compaction, destruction of moss mats and other wetland vegetation, disruption of surface water flows and drainage patterns and an overall reduction in invertebrate biomass (Hickman et al. 1999; Van Vierssen Trip and Wiersma in review). Therefore, ORVs reduce habitat quality and alter food resource availability for bog specialists, such as bog lemmings (*Snyaptomys* spp.) and shrews (*Sorex* spp.; Hickman et al. 1999).

### *1.4.3. Powerline Rights-of-Way*

Similar to the other taxonomic groups reviewed previously, small mammal response to powerline rights-of-way is highly dependent on the vegetation management regime (Clarke et al. 2006; Clarke and White 2008). Small mammal density and diversity are often related to vegetation diversity and habitat structure (Wilson et al. 1999). Therefore, in lieu of the benefits described for selective herbicide spraying and removal of tall vegetation, it is not surprising that powerline rights-of-way can be made into valuable habitat for small mammals (Johnson et al. 1979; Clarke et al. 2006; Clarke and White 2008; Yahner et al. 2007).

For example, Clarke and White (2008) investigated the effects that different management practices had on small mammal recolonization in southern Victoria, Australia. They found that corridors managed by slashing on a three-year cycle had the lowest abundance of native small mammals and highest abundance of opportunistic, generalist species, such as the house mouse (*Mus domesticus*). In contrast, an unmanaged section of the powerline had a well established community of all five native small mammal species with a low abundance of house mice. Therefore, powerline corridors in a mid to late-successional vegetation stage supported a more diverse, native assemblage of small mammals. Similar patterns have been reported elsewhere (Goldingay and Whelan 1997; Yahner et al. 2007). For example, Goldingay and Whelan (1997) compared the abundance and diversity of small mammals in two mowed powerline corridors to an unmanaged powerline corridor. No small mammals were captured in the mowed powerline corridors while the unmanaged corridor supported three different native small mammal species.

Johnson et al. (1979) compared the density, species richness and evenness of small mammals within a powerline corridor, edge and interior forest in Tennessee. They found that small mammal density and species richness were highest in the edge habitat and lowest in the adjacent forest. However, species evenness was highest in the forest. Powerline corridors and edges had reduced species evenness due to the presence of uncommon species or high abundance of a few dominant generalist species, while the undisturbed forest interior had few species that were nearly equal in abundance (Johnson et al. 1979).

Other studies have examined the potential barrier effect created by powerline corridors on small mammals (Schreiber and Graves 1977; Goosem and Marsh 1997). Goosem and Marsh (1997) report that powerline corridors managed by mowing and complete vegetation removal are perceived as barriers to small mammal movement. In contrast, Schreiber and Graves (1977) report that small mammals are able to cross a powerline corridor made up of a well-established shrub community. This further emphasizes the conservation value of maintaining a mid to late successional shrub community within powerline rights-of-way.

### 1.5. Newfoundland Terrestrial Mammal Community and Food Web Dynamics

Historically, Newfoundland had 14 native mammal species; less than 50% of the diversity found on the adjacent mainland (Cameron 1958): seven Carnivora, three Rodentia, one Artiodactyla, one Lagomorpha, and two Chiroptera (Dodds 1983; Table 1). Therefore, the depauperate native assemblage of mammals was dominated by predators with few prey species (Hearn et al. 2006). From this native assemblage, *M. pennsylvanicus* was an important food species, serving as the primary food resource for notable predators that included American marten (*Martes americana*) and red fox (*Vulpes vulpes*).

Over the past century, Newfoundland's mammalian community has been greatly altered. The wolf became extinct in the early 1930s (Allen and Barbour 1937; Maunder 1982), and between 1958 and 1998, six small mammal species were introduced to the island. Even with the most recent addition to the Newfoundland terrestrial ecosystem, the coyote (Moore and Parker 1992), these introductions and colonizations have created a shift to a less carnivore-dominated system.

However, *M. pennsylvanicus* is still an important component of Newfoundland's terrestrial food web. Even with the introduction of various small mammal species and snowshoe hare (*Lepus americanus*), the most recent dietary analyses showed that *M. pennsylvanicus* composes 80% of the summer diet and 47.5% of the winter diet of the threatened Newfoundland marten (Gosse and Hearn 2005).

## 1.6. Study Species

### *1.6.1. Microtus pennsylvanicus*

*M. pennsylvanicus* are distributed widely throughout North America. Their distribution spans across Canada, from Newfoundland to British Columbia and into the Yukon, across northern and eastern United States and as far north as Alaska and south to Mexico (Reich 1981, Hansen and Boonstra 2000). This vast range makes them the most widely distributed *Microtus* species (Reich 1981) and the only small mammal native to the island of Newfoundland (Dodds 1983).

Across most of its range, *M. pennsylvanicus* is considered to be a habitat specialist, favouring grassland and moist meadow-like environments that support their preferred diet of herbaceous vegetation, grass, sedge, fruits and seeds (Folinsbee et al. 1973). However, forested areas also may provide suitable habitat (Cameron 1958; Folinsbee et al. 1973; Reich 1981). This occurrence is more commonly reported in island than mainland populations (Grant 1971). Cameron (1958) reported higher densities of *M. pennsylvanicus* in forested areas than grasslands on the island of Newfoundland. This led to the suggestion that *M. pennsylvanicus* prefers forested areas and therefore differs from mainland populations in habitat (Cameron 1958). Cameron (1964) hypothesized that in Newfoundland *M. pennsylvanicus* were able to inhabit forested areas due to the absence of competitors, mainly southern red-backed vole (*Myodes gapperi*), which can exclude *M. pennsylvanicus* from forests. Folinsbee et al. (1973) disagreed with Cameron (1958) stating that *M. pennsylvanicus* shows a strong preference for meadow-like environments and rarely inhabits disturbed or coniferous forests on the island of Newfoundland. Grant (1971) agreed with Cameron (1964), but also suggested that *M. pennsylvanicus* displays a preference for grassland environments and will only move into forested areas due to intraspecific competition during high density years, thus explaining why *M. pennsylvanicus* is found in forested areas in Newfoundland more frequently than elsewhere.

*M. pennsylvanicus* can reproduce at any time of the year but usually does so from May through September. They have approximately four litters per year. Mean litter sizes range from 4.0 to 6.2, but can reach numbers of up to 11 young. Litter size is usually positively correlated with body size and does not vary with population density (Reich 1981). Young are weaned by two weeks of age (Reich 1981). In the wild an adult's life span does not usually exceed one year. In areas where populations are found in both grassland and forest environments, woodland populations have lower reproductive success and adult survival rates (Grant 1971). Similar to other small mammals, the species exhibits multiannual periodic density fluctuations or cycles, reaching peak densities every four years (Northcott 1974; Dodds 1983). Early studies conducted in

Newfoundland found that the island's cycle duration matched that observed elsewhere (Cameron 1958).

*M. pennsylvanicus* is active during both the day and night. However, daytime activity is usually associated with vegetative cover (Reich 1981). Home range size has been estimated to be approximately 183 $m^2$ in Virginia (Bowers et al. 1996), and 804 $m^2$ for males and 200 $m^2$ for females during periods of medium density in New York (Pugh and Ostfield 1998). *M. pennsylvanicus* exhibits aggressive behaviour in intraspecific encounters (Reich 1981). However, when encountering another mouse species, including southern red-backed vole and deermouse (*Peromyscus maniculatus*), they are usually the subordinate (Cameron 1958). As the only native mouse species in Newfoundland, *M. pennsylvanicus* traditionally didn't experience aggressive encounters with other species. However, following the colonization of the southern red-backed vole it has been hypothesized that *M. pennsylvanicus* may be excluded from previously held territories by the more aggressive southern red-backed vole (Hearn et al. 2006).

### *1.6.2. Sorex cinereus*

*S. cinereus* occurs throughout most of Alaska and Canada, from Labrador to British Columbia (excluding Prince Edward Island), as well as portions of the Northwest Territories, Yukon and Nunavut. They are also found throughout the northern and central United States (Whitaker 2004). *S. cinereus* was intentionally introduced into western Newfoundland in 1958. Ten males and 12 females were introduced from the Green River watershed, New Brunswick, into the St. Georges area to control larch sawfly (MacLeod

1960; Northcott 1974; Dodds 1983). Following their introduction, *S. cinereus* expanded its range quickly across the island. Dispersal rates have been reported as high as 33 km per year, with little interference from topographic barriers (Folinsbee 1971). With such dispersal capabilities, *S. cinereus* colonized the entire island of Newfoundland, including some offshore islands, by 1970 (Dodds 1983). A 1974 annual report of the Forest Insect and Disease Survey reported that the larval population of larch sawfly was low in most parts of Newfoundland, presumably due to the introduction of *S. cinereus* (Whitaker 2004).

*S. cinereus* is a habitat generalist that has been found in nearly every habitat type across its range, including arid grasslands, bogs, various kinds of woodlands, and tundra environments. Within these habitats it appears that they prefer cool, moist areas often containing extensive moss or litter coverage. This small mammal is usually classified as an insectivore. Studies on their diet indicate that they mainly consume insects (mainly Diptera, Coleoptera, and Lepidoptera larvae), spiders, centipedes, worms and mollusks. However, the species is sometimes classified as an omnivore since they may also consume seeds, vegetation and fungi (Whitaker 2004). Due to their small size *S. cinereus* relies on frequent feeding intervals to maintain their rapid metabolic rate. Mean metabolic rate of a 3.41 g animal has been recorded as 16.8 ml $O_2\ g^{-1}\ h^{-1}$, with a minimum rate of 10.6 ml $O_2\ g^{-1}\ h^{-1}$ and a maximum rate of 18.9 ml $O_2\ g^{-1}\ h^{-1}$. When under stress, their metabolic rate can reach extreme values of 30 ml $O_2\ g^{-1}\ h^{-1}$ (Churchfield 1990). On average individuals consume 3.3 times their body mass per day. During non-feeding periods they may experience depressed metabolic rates (Whitaker 2004). It is because of this fast metabolic rate that live-trapping of shrews usually results in high trap mortality rates.

The life span of *S. cinereus* usually does not exceed 16 months. The breeding season ranges from May through October (Bellocq and Smith 2003). Shrews make well-insulated, complex nests out of various materials such as grass, moss and shredded leaves. At age 22-25 days the young are fully grown and begin to disperse from the nest (Churchfield et al. 1990). They reach sexual maturity by age 2 months. However, they usually do not breed until the following spring. During one year, an individual may have one to five litters with a mean litter size of 6.5 (Whitaker 2004). Due to their breeding season and short life span, *S. cinereus* seasonal population trend exhibits a unimodal pattern, with abundance increasing steadily during spring and summer, peaking by late summer and declining again throughout autumn (Bellocq and Smith 2003). During spring and early summer the population is mainly young born the previous summer. Upon reproducing, this cohort usually dies by late autumn leaving the population dominated by juveniles who overwinter as subadults (Whitaker 2004). *S. cinereus* densities also varies between years. However, unlike most other small mammals, they rarely experience multiannual periodic density fluctuations. Instead, high densities appear to coincide with years of high prey density (Churchfield 1990).

This small mammal is usually classified as a nocturnal species. However, due to their foraging behaviour they are also commonly active throughout the day. Their night activity level is positively related to temperature, rain and cloud cover. Their home range size has been estimated between 5500–6000 $m^2$ (Pagels 1994; Whitaker 2004) and individuals usually display aggressive behaviour in intraspecific encounters (Whitaker 2004).

Although an introduced species to Newfoundland, the presence of *S. cinereus* has provided several benefits with little negative consequences. They were able to decrease larch sawfly populations (Whitaker 2004) and are now considered an alternate food source for predators, such as hawks, owls and large fish such as trout and salmon. Few mammalian predators prey upon *S. cinereus* due to a skin gland on the animal's side that produces a musky odour. However, there have been reports of marten feeding on this small mammal (Northcott 1974).

### 1.7. Objective and Hypotheses

The objective of my thesis was to examine the relationship between linear features and small mammal abundance and body condition in boreal forest and barren ecosystems in Newfoundland, Canada. I hypothesized that: 1) small mammal abundance is not related to proximity to linear features in either of these ecosystems because small mammals are usually not affected by traffic noise or emissions, they have small home ranges and they have high reproductive rates that offset the negative effects of traffic mortality (Adams and Geis 1983; McGregor et al. 2008; Bissonnette and Rosa 20009; Fuentes-Montemayor et al. 2009); 2) small mammals are sensitive to habitat characteristics, such as vertical stratification and spatial heterogeneity of vegetation and woody debris (Naxara et al. 2009), so microhabitat characteristics are the best predictor of small mammal presence and abundance; and 3) because edge habitats have higher abundance of food resources due to increased light availability and hence higher primary productivity, greater fruit production, longer fruiting periods and greater propagule and seed outputs than interior habitats (de Casenave et al. 1995; Forman et al. 2003), small mammal body condition will be positively related to proximity to linear features.

## 1.8. Research Benefits for Newfoundland

The few studies on ecological effects of linear features in this Province have concerned vegetation. Karim and Mallik (2008) examined zonation of plant communities along a highway right-of-way in Terra Nova National Park, Newfoundland, and Van Vierssen Trip and Wiersma (in review) examined on-trail soil erosion and indirect off-trail vegetation impacts along ORV trails. Both of these studies showed vegetation zonation patterns along linear transportation corridors as a function of microhabitat gradients created by the transportation route.

Overall, there is a lack of knowledge on how wildlife in Newfoundland responds to the direct presence of linear features and indirect alterations in the vegetation community adjacent to linear features, like those reported by Karim and Mallik (2008) and Van Vierssen Trip and Wiersma (in review). As previously stated, small mammals are key components of the terrestrial ecosystem. They help with seed dispersal and control invertebrate densities and are an important food source for birds of prey and carnivores. With Newfoundland's depauperate mammal community they play an important role in the terrestrial food web. Therefore, their abundance and distribution can influence other predator and prey species. This research will shed some light on how small mammals in Newfoundland are impacted by linear features, with the goal of being able to make recommendations that will help managers and land-use planners manage and regulate the use of linear infrastructure with minimal wildlife disturbance.

# CHAPTER 2. METHODS

## 2.1 Study Area

The study was conducted in the Maritime Barrens Ecoregion on the Avalon Peninsula of insular Newfoundland, Newfoundland and Labrador, Canada. This ecoregion is characterized by heath barrens, shallow fens and stunted *Abies* and *Picea* forests. The climate in this region consists of cool summers and relatively mild winters. Fog and precipitation are common (Damman 1983).

Trapping sites were divided into two main sections (Fig. 1): four trapping sites adjacent to trails were located within the Three Pond Barrens area of Pippy Park, St. John's, Newfoundland; and two trapping sites adjacent to a transmission line corridor and two sites adjacent to the Trans Canada Highway (TCH) were located approximately 50 km outside of St. John's near the eastern boundary of Butter Pot Provincial Park. These sites are described in further detail below.

Pippy Park is located along the northern edge of St. John's, Newfoundland. It is one of Canada's largest urban parks and covers an area of 13.76 km$^2$. The park has a combination of outdoor attractions including a campground, two golf courses and extensive hiking and ski trails, as well as urban institutions such as a hospital and university. One of the sample locations (Three Pond Barrens) is located north of the TCH and extends as far as the park boundary. It consists of an extensive network of highly used backcountry trails and power lines. Illegal use of ORV and dirt bikes is common in the park. A two-day voluntary survey (one weekday and one weekend day during July) conducted to determine intensity of use by people and domestic dogs yielded an average of 109 people and 69.5 dogs over a 9-hr period.

The sites adjacent to the TCH and transmission line were located in the transition zone between the Maritime Barrens and Avalon Forest ecoregions. The Avalon Forest Ecoregion is a 500 $km^2$ area occupying the central portion of the Avalon Peninsula. This ecoregion is similar in climate to the Maritime Barrens. However, the Avalon Forest Ecoregion has the most commercially productive *Abies* and *Picea* forests on the Avalon Peninsula. As well, due to its glacial history, the area has an irregular topography shaped by an underlying ribbed moraine (Damman 1983). Therefore, in contrast to sites in Pippy Park, forest sites in this section were along low, steep-sided, heavily forested hills. Traffic estimates for this section of the TCH are approximately 33,760 vehicles per day (J. Morrisey 2011, pers. comm.). The transmission line corridor in this area also is used by ORVs but less so than in Pippy Park.

### 2.2. Mapping linear features

I tracked the trails and power lines in Three Pond Barrens, Pippy Park and the transmission line outside Butter Pot Provincial Park using a Garmin 76 GPS unit. All trails and power lines in Pippy Park were tracked in January and February 2011. The transmission line was tracked in July 2011. Tracklogs were uploaded to ArcGIS using Garmin DNR software and converted into vector shapefiles using the ArcGIS Hawth's Tools extension (Beyer 2004).

To map the TCH and roads within Pippy Park I obtained vector data from the National Road Network (NRN) 2.0. The NRN was created by Natural Resources Canada, Centre for Topographic Information (CTI) between 2001 and 2009. Data for the NRN were acquired from varying sources including GPS, orthoimages, orthophoto and photogrammetry. The linear shapefile represents the center line of all non-restricted usage roads wider than 5 m. The planimetric accuracy of the NRN varies depending on the data source but is estimated to be ~10 m or less (Centre for Topographic Information 2009).

## 2.3. Trail study sites selection

I used ArcGIS (ESRI, version 9.2) to find suitable small mammal trap sites (meeting minimum size and habitat requirements) adjacent to trails within Pippy Park. To map land cover, I obtained data from the Earth Observation for Sustainable Development of Forests (EOSD) Land Cover Classification created by The Canadian Forest Service and the Canadian Space Agency. The land cover map is based on Landsat 7 Enhanced Thematic Mapper (ETM+) data and represents circa year 2000 conditions. The legend is based on the hierarchical National Forest Inventory (NFI) Land Cover Classification Scheme (Wulder and Nelson 2003). Out of the 36 cover types described by the EOSD, 16 were represented in Three Pond Barrens. Definitions of the represented cover types are displayed in Table 2. The EOSD raster layer had a cell size of 25 m and planimetric accuracy of 30 m or less.

In ArcGIS I identified suitable study sites adjacent to trails by creating two binary suitability maps based on habitat and Euclidean distance from the nearest trail. I converted the land cover map into two weighted overlays, one for optimal forest sites and the other for optimal barren sites. I assigned values to the cells based on forest habitat and barren habitat characteristics. To identify suitable barren sites dense coniferous, broadleaf and mixed wood were given the lowest value of 1. Open coniferous, broadleaf, and mixed wood were given a value of 3. Shrub was given a value of 5. Sparse coniferous, broadleaf and mixed wood were given a value of 8. Rock, exposed ground and herbs were given a value of 10, and water, wetlands, and developed areas were classified as restricted. To identify suitable forest sites rock, exposed ground and herbs were given a value of 1. Sparse coniferous, broadleaf and mixed wood were given a value of 5. Shrub was given a value of 6. Open coniferous, broadleaf and mixed wood were given a value of 8. Dense coniferous, broadleaf and mixed wood were given a value of 10, and wetlands and developed areas were again classified as restricted. I aggregated suitable habitat cells within each surface into patches according to an eight-neighbour rule.

A small mammal trap site required a minimum area of 26 880 $m^2$ (refer to Fig. 2). Therefore, patches smaller than 26 880 $m^2$ were eliminated. The GIS analysis located three suitable forest sites and four suitable barren sites adjacent to a trail within Three Pond Barrens. All sites met the size criteria. However, sites had to be ground-truthed before they could be selected. Verification of the sites revealed that one of the barren sites was actually better suited as a forest site. All others were true to their classification. Of the suitable sites, two barren and two forest sites were chosen based on travel distance between a forest and barren combination to limit the amount of time between checking small mammal traps located in each site.

## 2.4 Field Methods

### *2.4.1 Small Mammal Sampling*

Small mammal sampling was conducted from 2 July to 3 September 2011. I established eight sampling transects distributed among the three different types of linear features. Four transects were adjacent to recreational trails in the Three Pond Barrens area of Pippy Park, two transects were adjacent to the transmission line corridor, and the remaining two transects were adjacent to the Trans Canada Highway. The eight transects were divided evenly between forested and barren habitats. The sites were chosen to include variation in habitat and traffic types (ORVs, foot traffic, bicycles, and highway vehicles).

A sampling transect consisted of three parallel trap lines separated by 70 m each. This distance was chosen based on the home range size of the two study species. Each trap line was composed of 15 small Sherman Live Traps (2 x 2.5 x 6.5") spaced 12 m apart (Fig. 2). Traps were baited with a peanut butter and dry oatmeal mixture and insulated with non-absorbent cotton wool. I considered a trapping session to be a setup like this in a forest and a barren site for five consecutive nights. If weather did not permit trapping, then the traps remained closed for the evening and trapping began again the following night. Traps were opened 60 min before sunset and checked 60 min before sunrise. Two trapping sessions (cycles) were conducted for each transect. Sessions were separated by approximately 5 weeks. This resulted in a total of 3600 trap nights.

Upon capture, live *M. pennsylvanicus* were marked with two self-piercing ear tags (size 1005-1, National Band and Tag Company) and *S. cinereus* were marked with a unique coding system of black and/or red permanent hair dye (Powell and Proulx 2003). All live animals were weighed using an Ohaus 8261-MO 100 g spring scale to ±1 g. Live *M. pennsylvanicus* and dead *S. cinereus* were measured (body, tail, left hind foot and left ear length) using a Duratool carbon fibre digital caliper to ±0.1 mm. Dead *S. cinereus* were sexed via dissection, and weighed upon return from the field using an Ohaus cs-200 200 g portable digital scale to ±0.1 g. Live animals were released at the trap station. All small mammal handling was carried out following a protocol approved by the Memorial University Institutional Animal Care Committee (Protocol 11-03-YW) and approved under a research permit by the Newfoundland and Labrador Department of Environment and Conservation, Wildlife Division (permit number IW2011-06).

### *2.4.2. Traffic Volume*

Traffic volume was measured along the linear feature for each trapping site. I used TRAFx Off-Highway Vehicle Counters (generation 3) to measure ORV (ATV and dirt bike) traffic volume on the trails and the transmission line corridor (TRAFx Research Ltd. 2012). TRAFx counters contain magnetometers that are designed to detect disruptions to the earth's magnetic field from metallic objects (Hunt and Hosegood 2008). When a metal object, such as an ORV, passes within the counter's detection zone, the device records the passing date and time. At the four trail sites, one counter was buried approximately 10 cm below ground in the centre of the trail for 4 weeks. This procedure could not be used for the transmission line sites because the high voltage power line created a magnetic field that interfered with the counter's sensor, resulting in an elevated number of counts. Therefore, for the two transmission line sites counters were buried in the centre of two access trails approximately 50 m away from the lines. These counts were used as a proxy for traffic intensity along the transmission line corridor. Like the trail sites, counters were buried along the access trails for 4 weeks. All counters were set on OHV mode which has a detection zone diameter of 2 m. Since the counters were buried in the centre of the trails, and none of the trails was wider than 4 m, all ORVs that passed the counter should have been recorded. Tests performed on the devices revealed that some slow moving vehicles were counted twice or thrice (TRAFx Research Ltd. 2006). Therefore, to avoid over-estimating traffic volume, counts < 3 seconds apart from a previous count were discarded.

I obtained traffic volume data for the highway sites from the Department of Transportation and Works. I chose to use these data instead of using a TRAFx counter because the section of the TCH that was within the study area was wider than the detection zone of the counter and, due to the high traffic volume, deployment of the counter would have been difficult and potentially dangerous. The traffic count survey was conducted at the Foxtrap access ramp on 17 July 2007 from 07:00 to 15:00. There are no other exits between this location and the study sites so all vehicles counted at this survey site would have had to have passed in front of the two highway sites. Traffic count was converted by the Department of Transportation and Works into Average Annual Daily Traffic (AADT) by applying monthly, daily and time-of-day correction factors to account for variances. To account for the change in traffic volume between 2007 and 2011, I factored in a 1.5% growth rate per year (J. Morrisey 2011, pers. comm.).

### *2.4.3 Microhabitat Sampling*

Microhabitat sampling was conducted on every third trap per trap line. At each of these traps I established a 10 m radius circular plot with the trap location at the centre of the plot. Three sampling units were associated with each plot: (1) four quarters of the circular plot; (2) two perpendicular 20 m transects; and (3) four 1 $m^2$ quadrants 1 m away from the plot centre in each of the cardinal directions. This sample plot protocol was adopted from Rodgers et al. (2008). Within each quarter of the plot I measured tree stump, fallen log and dead tree density. Along the two 20 m transects I measured shrub density and diversity. Shrub diversity was calculated using Simpson's Diversity Index (1-D):

$$1 - D = 1 - \left(\frac{\sum ni(ni - 1)}{N(N - 1)}\right)$$

Where $N$ is the total number of shrub species in the sample plot and $n_i$ is the total number of individuals of species $i$ (Itô 2007). Within the four 1-m$^2$ quadrants I measured percent cover of woody stems, herbaceous stems, grass and sedge, moss or lichen, rock or ground and litter. In addition, I calculated percent canopy open above the trap using a spherical densitometer. Details on how each measurement was taken are in Table 3. Tree stump, fallen log, dead tree density, shrub density, woody stem cover and herbaceous stem cover were measured using the measurement protocols in Rodgers et al. (2008). Grass and sedge, moss or lichen, rock or ground and litter cover were modified from the surface component measurement described in Rodgers et al. (2008) by breaking the measurement down into its constituent parts. Percent canopy open was measured with a spherical densitometer instead of using photographs as suggested in Rodgers et al. (2008).

## 2.5 Data Analysis

### *2.5.1 Abundance*

I used the relative abundance measurement, catch per unit effort (CPUE) corrected for sprung traps, as a proxy for abundance for both study species (Nelson and Clark 1973). Relative abundance measures are accurate proxies for absolute abundance especially for comparative abundance (Hopkins and Kennedy 2004). Only first-time captures were included in the calculations of relative abundance.

I used general/generalized linear mixed-effect models (GLMMs) to model the response of *S. cinereus* and *M. pennsylvanicus* relative abundance to the presence of linear features. I tested for a difference in relative abundance among the three lines, three types of linear features, two habitat types and two trapping cycles. I incorporated trapping cycle (first or second) nested within site and line nested within site as random effects to account for temporal and sacrificial pseudoreplication (Hurlbert 1984, Heffner et al. 1996, Miller and Anderson 2004). Model assumptions and goodness-of-fit were evaluated using residual plots (Breslow 1996).

*S. cinereus* data were modelled using a Gaussian distribution with CPUE as the response variable. *M. pennsylvanicus* data had a high number of zero observations. Therefore, in order to meet model assumptions *M. pennsylvanicus* data were modelled using a Poisson distribution with catch as the response variable and log-base 10 of effort added to the model as an offset (Maunder and Punt 2004). All models were carried out with the statistical package R (v. 2.14.1; R Development Core Team 2010) using the package lme4 (Bates and Maechler 2010). Significance of fixed effects was evaluated by performing a type II ANOVA using Wald chi-square tests in the R package car (Fox et al. 2012). All statistical tests were conducted with $\alpha = 0.05$. Significant group means were compared using Tukey multiple comparison procedures from the R package multcomp (Hothorn et al. 2008; 2012).

### *2.5.2 Body condition*

Body condition was estimated for all dead *S. cinereus* specimens using a condition index. I defined body condition as an individual's energy reserve as a result of feeding. This definition assumes that body condition is an indicator of the energetic or nutritional state of an individual and is related to the animal's overall fitness (Peig and Green 2009, 2010). Condition indices are often used in conservation biology to explore relationships of body condition to environmental degradation (e.g., habitat loss or climate change) or to ecological interactions (e.g., diet or density; Stevenson and Woods 2006). In this study, I assessed the relationship of *S. cinereus* body condition to proximity to linear features. Population dynamics are linked to the health of the population's individuals and individual health may be altered due to environmental alterations and edge effects created by linear features (Stevenson and Woods 2006). Therefore, a change in body condition with proximity to linear features would portray an important relationship between small mammals and linear features that may not be evident by examining abundance alone. I estimated *S. cinereus* body condition using a scaled mass index ($\widehat{M}_i$) (Peig and Green 2009):

$$\widehat{M}_i = M_i \left[\frac{L_0}{L_i}\right]^{b_{SMA}}$$

Where $M_i$ and $L_i$ are the mass and body length of individual *i* respectively, $L_0$ is the arithmetic mean of the entire sample's body length and $b_{SMA}$ is the scaling exponent estimated by fitting a standardized major axis (SMA) regression line to ln-transformed *M* and *L* data (Fig. 3). Body length was used in the condition index instead of other body measurements because it had the strongest correlation with mass on a log-log scale (Table

4) and therefore is the most likely measurement to explain that fraction of mass associated with structural size (Peig and Green 2009). I used the scaled mass index proposed by Peig and Green (2009) instead of other competing indices (mainly Residual Index using OLS regression; Jakob et al. 1996) and ANCOVA (García-Berthou 2001)) because of its success in accounting for the changing relationship between body mass and length in small mammals as the individual's body size changes due to growth (Peig and Green 2010).

I examined the difference in body condition between the trapping lines and types of linear features adjacent to the trapping lines using general linear mixed-effect models with a Gaussian distribution. Site was a random effect for both models. Significance of fixed effects was evaluated by performing a type II ANOVA using Wald chi-square tests in the R package car (Fox et al. 2012).

### *2.5.3. Traffic Volume*

I used average number of vehicles per day to represent traffic volume for each type of linear feature. I tested the difference between traffic volume among the types of linear features with a one-way ANOVA and Tukey HSD post hoc analysis. Additionally, I examined the relationships between *S. cinereus* abundance and traffic volume and *M. pennsylvanicus* abundance and traffic volume using one-way ANOVAs.

### *2.5.4 Microhabitat*

To determine if microhabitat could explain differences in small mammal abundance among sites, I analyzed the correlation between *M. pennsylvanicus* and *S. cinereus* abundance and microhabitat variables using principal components analysis (PCA). Using the R package FactoMineR (Husson et al. 2012) I ran a PCA of microhabitat variables using the abundance of *M. pennsylvanicus* and *S. cinereus* as quantitative supplementary variables (de Lima et al. 2010). In the R package FactoMineR, supplementary variables are not included in the computation of the PCA. Once the PCA is performed, the correlation coefficient between the supplementary variables and the components is computed and projected onto the components (Abdi and Williams 2010). Therefore, this method enabled a comparison between microhabitat variables and species abundance (de Lima et al. 2010). All microhabitat variables were standardized to unit variance prior to PCA. This was necessary since the microhabitat variables were measured in different units (Husson et al. 2011). Only first-time captures from sites where microhabitat sampling was conducted were included in the analysis to ensure data independence. The Kaiser Criterion was used to determine the number of components to retain for interpretation.

## CHAPTER 3. RESULTS

### 3.1 Abundance

In total, 314 *S. cinereus* and 25 *M. pennsylvanicus* were captured during 3600 trap nights, for trapping success of 9.4%. Trapping success was 12.6 times as high for *S. cinereus* as for *M. pennsylvanicus*. The recapture rate for *S. cinereus* was 2.8% and for *M. pennsylvanicus* was 7.4%. Further details on raw capture data can be found in appendix A.

The CPUE of both species did not differ between trapping lines (*S. cinereus* $p$ = 0.653; *M. pennsylvanicus* $p$ = 0.230). However, the abundance of *S. cinereus* was highest near edges and the abundance of *M. pennsylvanicus* was highest in interior habitats (Fig. 4). The relative abundance of both species differed among types of linear features, although significantly so only for *S. cinereus* ($p$ = 0.012, for *M. pennsylvanicus* $p$ = 0.074; Fig. 4). Post-hoc comparisons indicated that pair-wise comparison of trail sites to transmission line sites were significant only for *S. cinereus* ($p$ = 0.012), but all others were not. *S. cinereus* CPUE was higher during the second trapping cycle ($p$ = 0.019), while *M. pennsylvanicus* CPUE was statistically similar during both trapping cycles. CPUE was not statistically different between forested and barren habitats for both species.

### 3.2 Body Condition

Body measurement data was collected from 94 *S. cinereus* specimens. Mean mass was 3.98±0.98 g, mean body length was 44.01±3.99 mm, mean tail length was 40.01±2.26 mm, mean left ear length was 3.84±0.48 mm and mean left hind foot length was 10.74±0.36 mm (all values expressed as mean ± SD). *S. cinereus* body condition was similar among the three trapping lines ($p = 0.927$) and three types of linear features ($p$=0.326). Further details on raw *S. cinereus* body measurement data can be found in appendix B.

### 3.3 Traffic Volume

Traffic volume differed across the three types of linear features ($p$<0.01). Traffic volume was highest along the highway sites and lowest along the transmission line sites. Average daily traffic volume was 33762 vehicles at the highway sites (one measurement for both sites), 1.223 vehicles at the transmission line sites (one measurement for both sites) and 5.57±1.47 vehicles at the four trail sites. The abundances of both *S. cinereus* and *M. pennsylvanicus* were unrelated to traffic volume (*S. cinereus* $p = 0.517$; *M. pennsylvanicus* $p = 0.444$).

### 3.4 Microhabitat

The first axis in PCA explained 50.9% of the variance (Table 5; Fig. 5). High positive values along this axis correspond with microhabitat variables that were associated with a mature forest, including high log, stump and dead tree densities, and a large abundance of litter. In the opposite direction, high negative values were associated with microhabitat variables that describe an open, grassland environment with high grass density and high percentage of open canopy (Table 6). The second axis in the PCA explained 24.5% of the variation (Table 5; Fig. 5). High positive values along this axis were indicative of younger forest with high shrub density and diversity (Table 6). The third axis explained 11.5% of the variation (Table 5) and was negatively associated with a high percentage of rocky ground cover (Table 6). The abundance of *S. cinereus* and *M. pennsylvanicus* were both positively correlated with the first PC. *S. cinereus* abundance was mainly associated with high abundance of litter, while *M. pennsylvanicus* was more strongly related to a high percentage of herbaceous ground cover (Fig. 5).

# CHAPTER 4. DISCUSSION AND CONCLUSIONS

## 4.1. DISCUSSION

I found no strong association of the abundance of *S. cinereus* or *M. pennsylvanicus* and distance from linear features. This result agrees with other studies on road and edge effects on small mammals and supports my initial hypothesis (Fahrig and Rytwinski 2007; McGregor et al. 2008; Bisonnette and Rosa 2009). Possible reasons for the lack of relationship include small movement distances and territory sizes, high reproductive rates that allow small mammals to persist in light of negative effects resulting from traffic mortality, minimal traffic disturbance and reduced predation due to the negative effects of linear features on predators (Forman et al. 2003; Fahrig and Rytwinski 2009). However, there were some interesting patterns observed between the two species suggesting a species-specific interaction. For example, edges had a higher abundance of *S. cinereus* and lower abundance of *M. pennsylvanicus* compared with the interior habitat for all three types of linear features.

*S. cinereus* is a habitat generalist that is found in nearly every habitat type across its range (Whitaker 2004). Structurally complex edges support high abundances of insects and other invertebrate prey species (Forman et al. 2003), so it is not surprising that *S. cinereus* occurs in disturbed edge environments. In contrast, *M. pennsylvanicus* is a habitat specialist favouring grassland and moist meadow-like environments (Folinsbee et al. 1973; Reich 1081). In general, habitat specialist species are more prone to experience negative effects from landscape fragmentation and disturbance than are generalist species (Devictor et al. 2008). Similar relationships between small mammals and linear features have been observed in other small mammal species including the habitat generalists

Coues' oryzomys (*Oryzomys couesi*) on Cozumel Island, Mexico (Fuentes-Montemayor et al. 2009) and white-footed deermouse in southern Ontario, Canada (Fahrig and Rytwinski 2007), and the habitat specialist San Diego pocket mouse (*Chaetodipus fallax*) in California, USA (Brehme 2003).

This analysis was performed on a small sample size of *M. pennsylvanicus*. CPUE was low for this species compared with *S. cinereus*. There are two possible explanations for this difference in abundance between these two species. First, trapping was conducted during a year with high precipitation. *S. cinereus* activity is positively correlated with rainfall (Vickery and Bider 1978; Whitaker 2004), whereas *M. pennsylvanicus* is more active on warm dry nights (Reich 1981). Second, *M. pennsylvanicus* undergo a population cycle and reach a peak every 3-5 years (Reich 1981). *S. cinereus* density varies from year to year but not so predictably or with such extremes. Generally, *S. cinereus* has a seasonal unimodal population cycle, with abundance increasing steadily during spring and summer, peaking by late summer and declining again throughout autumn (Bellocq and Smith 2003). This cycle explains why the CPUE of this species was highest during the second trapping cycle (August) compared with the first (July). It would be interesting to compare the difference in *M. pennsylvanicus* abundance with proximity to linear features during a high-density year to confirm that abundance is unrelated to linear features.

The relative abundance of *S. cinereus* varied among trapping sites, with highest abundances in sites adjacent to trails and lowest abundances in sites adjacent to the transmission line. I found a similar but statistically insignificant pattern for *M. pennsylvanicus*. We tested two competing hypotheses for why the sites adjacent to trails supported higher abundances of small mammals than the sites adjacent to the highway or transmission line: (1) noise pollution and traffic mortality created by high traffic volume creates a "road-effect zone" greater than 170-m in sites adjacent to the highway or transmission line; and (2) the habitat in sites adjacent to trails can support higher *S. cinereus* and *M. pennsylvanicus* carrying capacities than the sites adjacent to the highway or transmission line.

I rejected the traffic hypothesis because comparisons of traffic volume to *S. cinereus* and *M. pennsylvanicus* abundance showed no relationship. Sites with highest traffic volume did not have lowest abundances. Results of other studies on the effects of traffic on small mammal abundance are similar (Goosem 2002; Rico et al. 2007; McGregor et al. 2008).

Results from the PCA support the habitat suitability hypothesis as an explanation for the higher abundance of small mammals in the trail sites than in the highway or transmission line sites. The abundances of both study species was positively correlated with the first PC. High positive values along this axis correspond with microhabitat variables that are associated with a mature forest, including high log, stump and dead tree densities, and a large abundance of litter. The CPUE of *S. cinereus* was strongly associated with litter cover. Litter depth and moisture are important habitat requirements for this species (Brannon 2000; Whitaker 2004; Nocera and Dawe 2008). Dense litter cover provides protective cover and a good foraging environment for this species (Nocera and Dawe 2008). The decomposing litter provides moisture to support diverse and abundant invertebrate prey species. As well, the moisture retained in the litter is important for counteracting the high respiratory water losses associated with the species' high metabolic rate (Brannon 2002). The CPUE of *M. pennsylvanicus* was strongly associated with herbaceous ground cover. I interpret this finding in terms of diet, which consists mainly of herbaceous vegetation, grass, sedge, fruits and seeds (Reich 1981).

Trail site A had the highest abundance of *M. pennsylvanicus* and second highest abundance of *S. cinereus*. This site also had the highest litter and herbaceous ground cover, the two microhabitat variables that were most closely related to abundance of *S. cinereus* and *M. pennsylvanicus*, respectively. Trail site C had the highest abundance of *S. cinereus* and was closely associated with stump, log and dead tree density, the microhabitat variables describing the first PC. This highlights the preference for mature forest by *S. cinereus*. Sites E and G were on the other extreme of the first axis. These sites were open, grassy habitats. Surprisingly, these sites were not associated with the highest abundance of *M. pennsylvanicus*. Overall, the PCA results suggest that the trail sites, mainly site A, B and C, were more suitable habitat and offered more resources to support a higher abundance of small mammals than the other sites.

Across most of its range *M. pennsylvanicus* is considered a habitat specialist, favouring grassland and moist meadow-like environments (Folinsbee et al. 1973; Reich 1981). Therefore, it was surprising that the abundance of *M. pennsylvanicus* was positively correlated with the first PC. However, other studies have shown that forested areas also may provide suitable *M. pennsylvanicus* habitat (Folinsbee et al. 1973). The occurrence of *M. pennsylvanicus* in forest habitats is more commonly reported in island than mainland populations. Cameron (1958) reported higher densities of *M. pennsylvanicus* in forested areas than grasslands in Newfoundland compared with other locations and he hypothesized that Newfoundland *M. pennsylvanicus* could inhabit forested areas due to the absence of competitors (mainly southern red-backed vole on the mainland; Cameron 1964). Folinsbee et al. (1973) disagreed with this interpretation and showed that *M. pennsylvanicus* strongly prefers meadow-like environments and rarely inhabits disturbed or coniferous forests in Newfoundland, except during high abundance years. My results suggest that *M. pennsylvanicus* can inhabit forested habitats, particularly those associated with a high abundance of coarse woody debris. The southern red-backed vole has recently colonized western Newfoundland (Hearn et al. 2006) and the species is spreading rapidly across the island (Rodrigues 2010) so interspecific interactions may affect *M. pennsylvanicus* distribution on the Avalon Peninsula in the future.

I found no relationship between *S. cinereus* body condition and linear features. I hypothesized that since edge habitats have higher abundance of food resources due to increased light availability and hence higher primary productivity (de Casenave et al. 1995; Forman et al. 2003), small mammal body condition would be positively related to proximity to linear features. Fuentes-Montemayor et al. (2009) also found that road edges had no effect on small mammal body condition, with the exception of Coues' oryzomys, whose males had better body condition than their female counterparts. However, in contrast to this, Sabino-Marques and Mira (2011) found that greater white-toothed shrews had larger body size and better body condition near roads than in riparian habitats, suggesting that roadside habitats have increased prey availability due to higher abundances of insects and other invertebrates. The results from this study suggest that *S. cinereus* prey availability and foraging activity are similar in both edge and interior habitats adjacent to the examined linear features.

### 4.2. Study Limitations and Future Directions

It is important to note some of the potential limitations associated with this study and outline areas where future studies might improve upon the results. First, as mentioned above, this analysis was performed on a small sample size of *M. pennsylvanicus*. Small sample sizes reduce the power of a study and increases the probability of making a type II error (Macfarlane 2003). With a sample size of 25 *M. pennsylvanicus* captures, it is possible that the model did not detect a relationship between *M. pennsylvanicus* abundance and linear features when there was one. A future study could address this issue by examining the *M. pennsylvanicus* historic abundance cycle and choosing to trap in a peak abundance year. As well, a second field season of small mammal trapping at the eight sites would improve the overall strength of the study and may possibly show different patterns. Unfortunately, due to time and resource constraints this was beyond the scope of this study but should be considered in the future.

Secondly, the study design did not include a control setup. Therefore, I cannot be certain that the distributions of *M. pennsylvanicus* and *S. cinereus* among the three types of linear features would differ from an undisturbed area. A future study that combined field observations with a controlled experiment, such as the inclusion of a linear feature-free control treatment would address this issue and improve upon the findings (Eberhardt and Thomas 1991).

Thirdly, a future study that involved multiple field seasons of trapping pre- and post-creation of a linear feature would address issues related to the time-scale of this study. Such a study could examine pre- and post-disturbance vegetation succession and add value to the microhabitat analysis performed within this study.

Another potential limitation in this study is the use of only *S. cinereus* mortalities in the body condition analysis. It is possible that individuals that died in the traps were less fit thus introducing a bias in the analysis. Similarly, many shrew mortalities during live-trapping studies are due to starvation because of the species` fast metabolic rate (Churchfield 1990; Whitaker 2004). Therefore, the mass of these specimens may be compromised because these specimens were depleted of their energy stores. Unfortunately, due to time constraints all live *S. cinereus* captures could not be measured. Therefore, caution should be taken when interpreting the results of this analysis.

## 4.3. Conclusion

In conclusion, I found few relationships between small mammal abundance in the Maritime Barrens ecoregion of Newfoundland and linear features. Therefore, microhabitat and food availability may be more important than linear features in affecting population density of these species. However, the two species differed in their relationships to linear features, possibly benefiting the introduced generalist *S. cinereus* and disturbing the native specialist *M. pennsylvanicus.* Given that vegetative cover is affected by linear features (Forman et al. 2003; Van Vierssen Trip and Wiersma in review), there is strong potential for habitat quality to change adjacent to linear features thus impacting the small mammal community. Given the importance of small mammals in the Newfoundland food chain, and the amount of recreational activity on the island, as well as planned developments for increased roads and transmission line rights-of-way, land managers should be mindful of the cumulative impacts of linear features on flora and fauna.

## References

Abdi, H. and Williams, L. J. 2010. Principal component analysis. Wiley Interdisciplinary Reviews: Computational Statistics 2, 433–459.

Adams, L. W. and Geis, A. D. 1983. Effects of roads on small mammals. Journal of Applied Ecology 20, 403–415.

Allen, G. M. and Barbour, T. 1937. The Newfoundland wolf. Journal of Mammalogy 18, 229–234.

Ascensão, F., Clevenger, A. P., Grilo, C., Filipe, J., and Santos-Reis, M. 2012. Highway verges as habitat providers for small mammals in agrosilvopastoral environments. Biodiversity Conservation 21, 3681–3697.

Askins, R. A. 1994. Open corridors in a heavily forested landscape: impact of shrubland and forest interior birds. Wildlife Society Bulletin 22, 339–347.

Attum, O. 2007. Can landscape use be among the factors that potentially make some ungulates species more difficult to conserve? Journal of Arid Environments 69, 410–419.

Baker, J., Goldingay, R. L., and Whelan, R. J. 1998. Powerline easements through forest: a case study of impacts on avifauna. Pacific Conservation Biology 4, 79–89.

Balkenhol, N. and Waits, L. P. 2009. Molecular road ecology: exploring the potential of genetics for investigating transportation impacts on wildlife. Molecular Ecology 18, 4151–4164.

Banks, P. B. and Bryant, J. V. 2007. Four-legged friend or foe? Dog walking displaces native birds from natural areas. Biology Letters 3, 611–613.

Barton, D. C. and Holmes, A. L. 2007. Off-highway vehicle trail impacts on breeding songbirds in northeastern California. The Journal of Wildlife Management 71, 1617–1620.

Bates, D. and Maechler, M. 2010. lme4: Linear mixed-effects models using S4 classes (Version 0.999375-35) [R package]. Available from http://CRAN.Rproject.org/package=lme4.

Bellamy, P. E., Shore, R. F., Ardeshir, D., Treweek, J. R. and Sparks, T. H. 2000. Road verges as habitat for small mammals in Britain. Mammal Review 30, 131–139.

Bellocq, M. I. and Smith, S. M. 2003. Population dynamics and foraging of *Sorex cinereus* (masked shrew) in the boreal forest of eastern Canada. Annales Zoologici Fennici 40, 27–34.

Beyer, H. L. 2004. Hawth's Analysis Tools for ArcGIS [ArcGIS Software]. Available from http://www.spatialecology.com/htools.

Bissonette, J. A. and Rosa, S. A. 2009. Road zone effects in small-mammal communities. Ecology and Society 14, 27–41.

Boves, T. J. and Belthoff, J. R. 2012. Roadway mortality of barn owls in Idaho, USA. The Journal of Wildlife Management 76, 1381–1392.

Bowers, M. A., Gregario, K., Brame, C. J., Matter, S. F., and Dooley, J. L., Jr. 1996. Use of space and habitats by meadow voles at the home range, patch and landscape scales. Oecologia 105, 107–115.

Boyd, D. K and Pletscher, D. H. 1999. Characteristics of dispersal in a colonizing wolf population in the central Rocky Mountains. Journal of Wildlife Management 63, 1094–1108.

Brannon, M. P. 2000. Niche relationships of two syntopic species of shrews, *Sorex fumeus* and *S. cinereus*, in the southern Appalachian Mountains. Journal of Mammalogy 81, 1053–1061.

Brehme, C. S. 2003. Response of small terrestrial vertebrates to roads in a coastal sage scrub ecosystem. (M.S. Thesis). San Diego State University, San Diego, CA.

Breslow, N. E. 1996. Generalized linear models: checking assumptions and strengthening conclusions. Statistica Applicata 8, 23–41.

Brooks, M. L. 1995. Benefits of protective fencing to plant and rodent communities of the western Mojave Desert, California. Environmental Management 19, 65–74.

Brown, G. P., Phillips, B. L., Webb, J. K. and Shine, R. 2006. Toad on the road: use of roads as dispersal corridors by cane toads (*Bufo marinus*) at an invasion front in tropical Australia. Biological Conservation 133, 88–94.

Brown, P. R., Huth, N. I., Banks, P. B. and Singleton, G. R. 2007. Relationship between abundance of rodents and damage to agriculture crops. Agriculture, Ecosystems and Environment 120, 405–415.

Burge, B. L. 1983. Impact of Frontier 500 off-road vehicle race on desert tortoise habitat. Proceedings of the Desert Tortoise Council Symposium 1983, 27–38.

Burger, J., Zappalorti, R. T., Gochfeld, M. and DeVito, E. 2007. Effects of off-road vehicles on reproductive success of pine snakes (*Pituophus (sic) melanoleucus*) in the New Jersey pinelands. Urban Ecosystems 10, 275–284.

Bury, R. B., Luckenback, R. A. and Busack, S. D. 1977. Effects of off-road vehicles on vertebrates in the California desert. (Wildlife Research Report 8). Washington, D.C.: National Fish and Wildlife Laboratory, U.S. Fish and Wildlife Service.

Cameron, A. W. 1958. Mammals of the islands in the Gulf of St. Lawrence (Bulletin No. 154) [Bulletin]. Ottawa, ON: National Museum of Canada.

Cameron, A. W. 1964. Competitive exclusion between the rodent genera *Microtus* and *Clethrionomys*. Evolution 18, 630–634.

Canfield, J. E., Lyon, L. J., Hillis, J. M., & Thompson, M. J. 1999. Ungulates. In G. Joslin and H. Youmans (coordinators), *Effects of recreation on Rocky Mountain wildlife: a review for Montana* (pp. 6.1–6.25). Committee on Effects of Recreation on Wildlife, Montana Chapter of The Wildlife Society.

Carey, A. B. and Harrington, C. A. 2001. Small mammals in young forests: implications for management for sustainability. Forest Ecology and Management 154, 289–309.

Carr, L. W. and Fahrig, L. 2002. Effect of road traffic on two amphibian species of differing vagility. Conservation Biology 15, 107–1078.

Centre for Topographic Information. 2009. National Road Network. Sherbrooke, Quebec: Centre for Topographic Information, Natural Resources Canada.

Chasko, G. G. and Gates, J. E. 1982. Avian habitat suitability along a transmission-line corridor in an oak-hickory forest region. Wildlife Monographs 82, 1–41.

Churchfield, S. 1990. The Natural History of Shrews. Bromley, Kent: Christopher Helm (Publishers) Ltd.

Ciuti, S., Northrup, J. M., Muhly, T. B., Simi, S., Musiani, M., Pitt, J. A. and Boyce, M. S. 2012. Effects of humans on behaviour of wildlife exceed those of natural predators in a landscape of fear. PLOS ONE 7, e50611.

Clarke, D. J., Pearce, K. A. and White, J. G. 2006. Powerline corridors: degraded ecosystems or wildlife havens? Wildlife Research 33, 615–626.

Clarke, D. J. and White, J. G. 2008. Recolonisation of powerline corridor vegetation by small mammals: timing and the influence of vegetation management. Landscape and Urban Planning 87, 108–116.

Clevenger, A. P., Chruszcz, B. and Gunson, K. 2001. Drainage culverts as habitat linkages and factors affecting passage by mammals. Journal of Applied Ecology 38, 1340–1349.

Confer, J. L., Hauck, T., Silvia, M. E., and Frary, V. 2008. Avian shrub land management and shrub land nesting success. In J. W. Goodrich-Mahony, L. Abrahamson, J. Ballard and S. Tikalsky (Eds.), *The Eighth International Symposium on Environmental Concerns in Rights-of-Way Management* (pp. 407–412). Oxford, UK: Elsevier.

Creel, S., Fox, J. E., Hardy, A., Sands, J., Garrott, B. and Peterson, R. O. 2002. Snowmobile activity and glucocorticoid stress responses in wolves and elk. Conservation Biology 16, 809–814.

Damman, A. W. H. 1983. An ecological subdivision of the island of Newfoundland. In G. R. South (Ed.), *Biogeography and Ecology of the Island of Newfoundland* (pp. 163–206). The Hague, Netherlands: Dr. W. Junk Publishers.

Davenport, J. and Switalski, T. A. 2006. Environmental impacts of transport, related to tourism and leisure activities. In J. Davenport & J. L. Davenport (Eds.), *The Ecology of Transportation: Managing Mobility for the Environment* (pp. 333–360). Dordrecht, Netherlands: Springer.

de Casenave, J. L., Pelotto, J. P. and Protomastro, J. 1995. Edge-interior differences in vegetation structure and composition in a Chaco semi-arid forest, Argentina. Forest Ecology and Management 72, 61–69.

de Lima, D. O., Azambuja, B. O., Camilotti, V. L. and Cáceres, N. C. 2010. Small mammal community structure and microhabitat use in the austral boundary of the Atlantic forest, Brazil. Zoologia 27, 99–105.

Devictor, V., Julliard, R. and Jiguet, F. 2008. Distribution of specialist and generalist species along spatial gradients of habitat disturbance and fragmentation. Oikos 117, 507–514.

Dodds. D. G. 1983. Terrestrial Mammals. In G. R. South (Ed.), *Biogeography and Ecology of the Island of Newfoundland* (pp. 509–550). The Hague, Netherlands: Dr. W. Junk Publishers.

Dyer, S. J., O'Neill, J. P., Wasel, S. M. and Boutin, S. 2001. Avoidance of industrial development by woodland caribou. Journal of Wildlife Management 65, 531–542.

Dyer, S. J., O'Neill, J. P., Wasel, S. M. and Boutin, S. 2002. Quantifying barrier effects of roads and seismic lines on movements of female woodland caribou in northeastern Alberta. Canadian Journal of Zoology 80, 839–845.

Eberhardt, L. L. and Thomas, J. M. 1991. Designing Environmental Field Studies. Ecological Monographs 62, 53–73.

Evans, D. R. and Edward Gates, J. 1997. Cowbird selection of breeding areas: the role of habitat and bird species abundance. Wilson Bulletin 109, 470–480.

Fahrig, L. and Rytwinski, T. 2009. Effects of roads on animal abundance: an empirical review and synthesis. Ecology and Society 14, 21–41.

Fahrig, L. and Rytwinski, T. 2007. Effect of road density on abundance of white-footed mice. Landscape Ecology 22, 1501–1512.

Folinsbee, J. 1971. Ecology of the masked shrew, *Sorex cinereus*, on the island of Newfoundland, Canada. (Ph.D. Thesis). University of Manitoba, Winnipeg, MB.

Folinsbee, J., Riewe, R. R., Pruitt, W. O., Jr. and Grant, P. R. 1973. Ecological distribution of the meadow vole, *Microtus pennsylvanicus terraenovae,* (Rodentia: Cricetidae), on the main island of Newfoundland. The Canadian Field-Naturalist 87, 1–4.

Forman, R. T. T. 2000. Estimate of the area affected ecologically by the road system in the United States. Conservation Biology 14, 31–35.

Forman, R. T. T. and Deblinger, R. D. 2000. The ecological road-effect zone of a Massachusetts (U.S.A.) suburban highway. Conservation Biology 14, 36–46.

Forman, R. T. T., Reineking, B. and Hersperger, A. M. 2002. Road traffic and nearby grassland bird patterns in a suburbanizing landscape. Environmental Management 29, 782–800.

Forman, R. T. T., Sperling, D., Bissonette, J. A., Clevenger, A. P., Cutshal, C. D., Dale, V. H., Farig, L., France, R., Goldman, C. R., Heanue, K., Jones, J. A., Swanson, F. J., Turrentine, T. and Winter, T. C. 2003. Road Ecology: Science and Solutions. Washington, D.C.: Island Press.

Forrest, A. and St. Clair, C. C. 2006. Effects of dog leash laws and habitat type on avian and small mammal communities in urban parks. Urban Ecosystems 9, 51–66.

Fox, J., Weisberg, S., Bates, D., Firth, D., Friendly, M., Gorjanc, G., Graves, S., Heiberger, R., Laboissiere, R., Monette, G., Nilsson, H., Ogle, D., Ripley, B. and Zeileis, A. 2012. Package car: Companion to Applied Regression (Version 2.12) [R Package]. Available from http://CRAN.Rproject.org/package=car.

Fuentes-Montemayor, E., Cuarón, A. D., Vázquez-Domínguez, E., Benítez-Malvido, J., Valenzuela-Galván, D. and Andresen, E. 2009. Living on the edge: roads and edge effects on small mammal populations. Journal of Animal Ecology 78, 857–865.

García-Berthou, E. 2001. On the misuse of residuals in ecology: testing regression residuals vs the analysis of covariance. Journal of Animal Ecology 70, 708–711.

Garland, T. and Bradley, W. G. 1984. Effects of a highway on Mojave Desert rodent populations. American Midland Naturalist 111, 47–56.

Gates, J. E. 1991. Powerline corridors, edge effects, and wildlife in forested landscapes of the central Appalachians. In J. E. Rodiek and E. G. Bolen (Eds.), *Wildlife and Habitats in Managed Landscapes* (pp. 13–32). Washington, D.C.: Island Press.

Gebhardt, K., Anderson, A. M., Kirkpatrick, K. N. and Shwiff, S. A. 2011. A review and synthesis of bird and rodent damage estimates to select California crops. Crop Protection 30, 1109–1116.

Gerlach, G. and Musolf, K. 2000. Fragmentation of landscape as a cause for genetic subdivision in bank voles. Conservation Biology 14, 1066–1074.

Goldingay, R. L. and Whelan, R. J. 1997. Powerline easements: do they promote edge effects in eucalypt forest for small mammals? Wildlife Research 24, 737–744.

Gosse, J. W. and Hearn, B. J. 2005. Seasonal diets of Newfoundland marten, *Martes americana atrata*. Canadian Field-Naturalist 119, 43–47.

Goosem, M. and Marsh, H. 1997. Fragmentation of a small-mammal community by a powerline corridor through tropical rainforest. Wildlife Research 24, 613–629.

Goosem, M. 2002. Effects of tropical rainforest roads on small mammals: edge effects and traffic disturbance. Wildlife Research 29, 277–289.

Government of Canada. 2012. Road Transportation. In *Transport Canada*. Retrieved September 5, 2012 from http://www.tc.gc.ca/eng/road-menu.htm.

Grant, P. R. 1971. The habitat preference of *Microtus pennsylvanicus*, and its relevance to the distribution of this species on islands. Journal of Mammalogy 52, 351–361.

Grilo, C., Bissonette, J. A. and Santos-Reis, M. 2009. Spatio-temporal patterns in Mediterranean carnivore casualties: consequences for mitigation. Biological Conservation 142, 301–313.

Hansen, T. F and Boonstra, R. 2000. The best in all possible worlds? A quantitative genetic study of geographic variation in the meadow vole, *Microtus pennsylvanicus*. Oikos 89, 81–94.

Hayward, L. S., Bowles, A. E., Ha, J. C. and Wasser, S. K. 2011. Impacts of acute and long-term vehicle exposure on physiology and reproductive success of the northern spotted owl. Ecosphere 2, 1–20.

Hearn, B. J., Neville, J. T., Curran, W. J. and Snow, D. P. 2006. First record of the southern red-backed vole, *Clethrionomys gapperi,* in Newfoundland: implications for the endangered Newfoundland marten, *Martes americana atrata.* Canadian Field-Naturalist 120, 50–56.

Heffner, R. A., Butler, M. J. and Reilly, C. K. 1996. Pseudoreplication revisited. Ecology 77, 2558–2562.

Hickman, G., Dixon, B. G. and Corn, J. 1999. Small Mammals. In G. Joslin and H. Youmans, (coordinators), *The Effects of Recreation on Rocky Mountain Wildlife. a Review for Montana* (pp. 4.1-4.16). Committee on Effects of Recreation on Wildlife, Montana Chapter of The Wildlife Society.

Hopkins, H. L. and Kennedy, M. L. 2004. An assessment of indices of relative and absolute abundance for monitoring populations of small mammals. Wildlife Society Bulletin 32, 1289–1296.

Hothorn, T., Bretz, F. and Westfall, P. 2012. Simultaneous inference in general parametric models (Version 1.2-12) [R package]. Available from http://CRAN.Rproject.org/package=multcomp

Hothorn, T., Bretz, F. and Westfall, P. 2008. Simultaneous inference in general parametric models. Biometrical Journal 50, 346–363.

Hunt, L. M. and Hosegood, S. 2008. The effectiveness of signs at restricting vehicle traffic: a case of seasonal closures on forest access roads. Canadian Journal of Forest Research 38, 2306–2312.

Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54, 187–211.

Husson, F., Josse, J., Lê, S. and Mazet, J. 2012. FactoMineR: Multivariate exploratory data analysis and data mining with R. (Version 1.14) [R package]. Available from http://factominer.free.fr/

Husson, F., Lê, S., and Pagès, J. 2011. Exploratory multivariate analysis by example using R. Boca Raton, FL: Chapman and Hall/CRC Press.

Itô, Y. 2007. Recommendations for the use of species diversity indices with reference to a recently published article as an example. Ecological Research 22, 703–705.

Jaeger, J. A. G., Bowman, J., Brennan, J., Fahrig, L., Bert, D., Bouchard, J., Charbonneau, N., Frank, K., Gruber, B. and Tluk von Toschanowitz, K. 2005. Predicting when animal populations are at risk from roads: an interactive model of road avoidance behavior. Ecological Modelling 185, 329–348.

Jakob, E. M., Marshall, S. D. and Uetz, G. W. 1996. Estimating fitness: a comparison of body condition indices. Oikos 77, 61–67.

Janis, M. W. & Clark, J. D. 2002. Responses of Florida panthers to recreational deer and hog hunting. The Journal of wildlife management 66, 839–848.

Jeffery, B. M. 2009. Effects of off-road vehicles on small mammals in Big Cypress National Preserve. (M.S. thesis). University of Florida, Gainesville, FL.

Jensen, C. R. and Gutherie, S. P. 2006. Outdoor recreation in America. Champaign, IL: Human Kinetics.

Johnson, W. C., Schreiber, R. K. and Burgess, R. L. 1979. Diversity of small mammals in a powerline right-of-way and adjacent forest in east Tennessee. American Midland Naturalist 101, 231–235.

Jules, E. S., Kauffman, M. J., Ritts, W. D. and Carroll, A. L. 2002. Spread of an invasive pathogen over a variable landscape: a nonnative root rot on Port Orford cedar. Ecology 83, 3167–3181.

Kamler, J., Turek, K., Homolka, M., Baňař, P., Barančeková, M., Heroldová, M., Krojerová, J., Suchomel, J. and Purchart, L. 2011. Inventory of rodent damage to forests. Journal of Forest Science 57, 219–225.

Karim, M. N. and Mallik, A. U. 2008. Roadside revegetation by native plants I. roadside microhabitats, floristic zonation and species traits. Ecological Engineering 32, 222–237.

King, D. I. and Byers, B. E. 2002. An evaluation of powerline rights-of-way as habitat for early successional shrubland birds. Wildlife Society Bulletin 30, 868–874.

Knight, R. L. and Cole, D. N. 1991. Effects of recreational activity on wildlife in wildlands. Transactions of the 56th North American Wildlife and Natural and Natural Resources Conference 56, 239–247.

Knight, R. L. and Kawashima, J. Y. 1993. Responses of raven and red-tailed hawk populations to linear right-of-ways. The Journal of Wildlife Management 57, 266–271.

Kroodsma, R. L. 1982. Bird community ecology on power-line corridors in east Tennessee. Biological Conservation 23, 79–94.

Kuykendall, M. T. and Keller, G. S. 2011. Impacts of roads and corridors on abundance and movement of small mammals on the Llano Estacado of Texas. The Southwestern Naturalist 56, 9–16.

Lambertucci, S. A., Speziale, K. L., Rogers, T. E. and Morales, J. M. 2009. How do roads affect the habitat use of an assemblage of scavenging raptors? Biodiversity Conservation 18, 2063–2074.

Lampe, U., Schmoll, T., Franzke, A. and Reinhold. K. 2012. Staying tuned: grasshoppers from noisy roadside habitats produce courtship signals with elevated frequency components. Functional Ecology 26, 1348–1354.

Laurance, S. G. W. 2004. Reponses of understory rain forest birds to road edges in Central Amazonia. Ecological Applications 14, 1344–1357.

Lenth, B. E., Knight, R. L. and Brennan, M. E. 2008. The effects of dogs on wildlife communities. Natural Areas Journal 28, 218–227.

Liddle, M. 1997. Recreation Ecology: The Ecological Impact of Outdoor Recreation and Ecotourism. London, UK: Chapman and Hall.

Luckenback, R. A. and Bury, R. B. 1983. Effects of off-road vehicles on the biota of the Algodones Dunes, Imperial County, California. Journal of Applied Ecology 20, 265–286.

Macfarlane, T. V. 2003. Sample size determination for research projects. Journal of Orthodontics 30, 99–100.

MacLeod, C. F. 1960. The introduction of the shrew, *Sorex cinereus cinereus* Kerr, into Newfoundland. (Technical Report, Forest Biology Laboratory). Quebec City, QC: Forest Biology Division, Research Branch, Department of Agriculture.

Marshall, J. S. and VanDruff, L. W. 2002. Impact of selective herbicide right-of-way vegetation treatment on birds. Environmental Management 30, 801–806.

Maser, C., Trappe, J. M. and Nussbaum, R. A. 1978. Fungal-small mammal interrelationships with emphasis on Oregon coniferous forests. Ecology 59, 799–809.

Maunder, J. E. 1982. The Newfoundland Wolf. Osprey 13, 36–49.

Maunder, M. N. and Punt, A. E. 2004. Standardizing catch and effort data: a review of recent approaches. Fisheries Research 70, 141–159.

McCue, A. J. 2012. General patterns among generalist: what is revealed by spatial models of coyotes? (M.S. thesis). Memorial University, St. John's, NL.

McDonald, W. R. and St. Clair, C. C. 2004. The effects of artificial and natural barriers on the movement of small mammals in Banff National Park, Canada. Oikos 105, 397–407.

McGregor, R. L., Bender, D. J. and Fahrig, L. 2008. Do small mammals avoid roads because of the traffic? Journal of Applied Ecology 45, 117–123.

Miller, S. G., Knight, R. L. and Miller, C. K. 2001. Wildlife responses to pedestrians and dogs. Wildlife Society Bulletin 29, 124–132.

Miller, R. B. and Anderson, M. J. 2004. Remedies for pseudoreplication. Fisheries Research 70, 397–407.

Monamy, V. and Fox, B. J. 2000. Small mammal succession is determined by vegetation density rather than time elapsed since disturbance. Austral Ecology 25, 580–587.

Moore, G. C. and Parker, G. R. 1992. Colonization by the eastern coyote (*Canis latrans*). In A. H. Boer (ed.), *Ecology and Management of the Eastern Coyote* (pp. 23–37). Fredericton, NB: Wildlife Research Unit, University of New Brunswick.

Moro, D. and Gadal, S. 2007. Benefits of habitat restoration to small mammal diversity and abundance in a pastoral agricultural landscape in mid-Wales. Biodiversity Conservation 16, 3543–3557.

Nalcor Energy. 2009. 2009 Environmental performance report. Retrieved November 15, 2012 from http://www.nalcorenergy.com/2010epr/environmental_policy.html.

Naxara, L., Pinotti, B. T. and Pardini, R. 2009. Seasonal microhabitat selection by terrestrial rodents in an old-growth Atlantic forest. Journal of Mammalogy 90, 404–415.

Nekola, J. C. 2012. The impact of a utility corridor on terrestrial gastropod biodiversity. Biodiversity Conservation 21, 781–795.

Nelson, L. Jr. and Clark, F. W. 1973. Correction for sprung traps in catch/effort calculations of trapping results. Journal of Mammalogy 54, 295–298.

Newfoundland and Labrador Motorized Snow Vehicles and All Terrain Vehicles Act, RSNL 1990 Chapter M-20. Retrieved November 12, 2012 from http://www.assembly.nl.ca/legislation/.

Newfoundland and Labrador Provincial Parks Act, RSNL1990 Chapter P-32. Retrieved November 12, 2012 from http://www.assembly.nl.ca/legislation/.

Niering, W. A. and Goodwin, R. H. 1974. Creation of relatively stable shrublands with herbicides: arresting "succession" on rights-of-way and pastureland. Ecology 55, 784–795.

Nocera, J. J. and Dawe, K. L. 2008. Managing for habitat heterogeneity in grassland agro-ecosystems influences the abundance of masked shrews (*Sorex cinereus*). Journal of Sustainable Agriculture 32, 379–392.

Norman, T. J. 2010. Canadian trail study: A comprehensive analysis of managed trails and trail uses. Halifax, NS: National Trails Coalition.

Northcott, T. H. 1974. The land mammals of insular Newfoundland. St. John's, NL: Wildlife Division, Department of Tourism, Government of Newfoundland and Labrador.

Olsen, R. A. and Brewer, M. J. 2003. Small mammal populations occurring in a diversified winter wheat cropping system. Agriculture, Ecosystems and Environment 95, 311–319.

Ortega, Y. K. And Capen, D. E. 1999. Effects of forest roads on habitat quality for ovenbirds in a forested landscape. The Auk 116, 937–946.

Pagels, J., Uthus, K. and Duval, H. 1994. The Masked Shrew, *Sorex cinereus*, in a relictual habitat of the southern Applachian Mountains. In J. Merritt, G. Kirkland and R. Rose (Eds.), *Advances in the Biology of Shrews* (pp. 103–109). Pittsburgh, PA: Carnegie Museum of Natural History Special Publication No. 18.

Parris, K. M. and Schneider, A. 2009. Impacts of traffic noise and traffic volume on birds of roadside habitats. Ecology and Society 14, 29. [online] URL: http://www.ecologyandsociety.org/vol14/iss1/art29/

Parris, K. M., Velik-Lord, M., and North, J. M. A. 2009. Frogs call at a higher pitch in traffic noise. Ecology and Society 14, 25. [online] URL: http://www.ecologyandsociety.org/vol14/iss1/art25/

Pearce. J. and Venier, L. 2005. Small mammals as bioindicators of sustainable boreal forest management. Forest Ecology and Management 208, 153–175.

Peig, J. and Green, A. J. 2009. New perspectives for estimating body condition from mass/length data: the scaled mass index as an alternative method. Oikos 118, 1883–1891.

Peig, J. and Green, A. J. 2010. The paradigm of body condition: a critical reappraisal of current methods based on mass and length. Functional Ecology 24, 1323–1332.

Phillips, T. 2010. Solar Shield–Protecting the North American Power Grid. In NASA Science News. Retrieved April 21, 2013 from http://science.nasa.gov/science-news/science-at-nasa/2010/26oct_solarshield/.

Powell, R. A. and Proulx, G. 2003. Trapping and marking terrestrial mammals for research: integrating ethics, performance criteria, techniques and common sense. The IALR Journal 44, 259–276.

Preisler, H. K., Ager, A. A. and Wisdom, M. J. 2006. Statistical methods for analysing responses of wildlife to human disturbance. Journal of Applied Ecology 43, 164–172.

Pugh, S. R. and Ostfield, R. S. 1998. Effects of prior population density on use of space by meadow voles, *Microtus pennsylcanicus*. Journal of Mammalogy 79, 551–557.

R Development Core Team. 2010. R: A language and environment for statistical computing (version 2.15.0) [Software]. Vienna, Austria: R Foundation for Statistical Computing. Available from http://www.R-project.org

Reich, L. M. 1981. *Microtus pennsylvanicus*. Mammalian Species 159, 1–8.

Rico, A., Kindlmann, P. and Sedláček, F. 2007. Barrier effects of roads on movements of small mammals. Folia Zoologica 56, 1–12.

Rodgers, A. R., Hutchison, C. and Simpson, M. S. 2008. Methods for sampling small mammals and their habitats in boreal mixedwoods. (CNFER Technical Report TR-001). Thunder Bay, ON: Centre for Northern Forest Ecosystem Research, Ontario Ministry of Natural Resources.

Rodrigues, B. 2010. Newfoundland and Labrador small mammal monitoring network 2009 season report. Corner Brook, NL: Department of Environment and Conservation, Government of Newfoundland and Labrador.

Rondinini, C. and Doncaster, C. P. 2002. Roads as barriers to movement for hedgehogs. Functional Ecology 16, 504–509.

Rosen, P. C. and Lowe, C. H. 1994. Highway mortality of snakes in the Sonoran Desert of southern Arizona. Biological Conservation 68, 143–148.

Row, J. R., Blouin-Demers, G. and Weatherhead, P. J. 2007. Demographic effects of road mortality in black ratsnakes (*Elaphe obsoleta*). Biological Conservation 137, 117–124.

Russell, K. N., Ikerd, H. and Droege, S. 2005. The potential conservation value of unmowed powerline strips for native bees. Biological Conservation 124, 133–148.

Rytwinski, T. and Fahrig, L. 2012. Do species life history traits explain population responses to roads? A meta-analysis. Biological Conservation 147, 87–98.

Rytwinski, T. and Fahrig, L. 2011. Reproductive rate and body size predict road impacts on mammal abundance. Ecological Applications 21, 589–600.

Sabino-Marques, H. and Mira, A. 2011. Living on the verge: are roads a more suitable refuge for small mammals than streams in Mediterranean pastureland. Ecological Research 26, 277–287.

Schreiber, R. K. and Graves, J. H. 1977. Powerline corridors as possible barriers to the movements of small mammals. American Midland Naturalist 96, 504–508.

Shepard, D. B., Kuhns, A. R., Dreslik, M. J. and Phillips, C. A. 2008. Roads as barriers to animal movement in fragmented landscapes. Animal Conservation 11, 288–296.

Sheppard, N., Pitt, K. A. and Schlacher, T. A. 2009. Sub-lethal effects of off-road vehicles (ORVs) on surf clams on sandy beached. Journal of Experimental Marine Biology and Ecology 380, 113–118.

Sirotnak, J. M. and Huntly, N. J. 2000. Direct and indirect effects of herbivores on nitrogen dynamics: voles in riparian areas. Ecology 81, 78–87.

Smith, M. B., Aborn, D. A., Gaudin, T. J. and Tucker, J. C. 2008. Mammalian predator distribution around a transmission line. Southeastern Naturalist 7, 289–300.

Steele, B. B., Bayn, R. L. Jr. and Grant, C. V. 1984. Environmental monitoring using populations of birds and small mammals: analyses of sampling effort. Biological Conservation 30, 157–172.

Steven, R., Pickering, C. and Castley, J. G. 2011. A review of the impacts of nature based recreation on birds. Journal of Environmental Management 92, 2287–2294.

Stevenson, R. D. and Woods, W. A. 2006. Condition indices for conservation: new uses for evolving tools. Integrative and Comparative Biology 46, 1169–1190.

Terwillinger, J. and Pastor, J. 1999. Small mammals, ectomycorrhizae, and conifer seccession in beaver meadows. Oikos 85, 83–94.

Trafx Research Ltd. 2012. Trafx Advanced Counting Systems. In *Trafx*. Retrieved June 8, 2012 from http://www.trafx.net/

Tull, J. C. and Brussard, P. F. 2007. Fluctuating asymmetry as an indicator of environmental stress from off-highway vehicles. The Journal of Wildlife Management 71, 1944–1948.

Tyser, R. W. and Worley, C. A. 1992. Alien flora in grasslands adjacent to road and trail corridors in Glacier National Park, Montana (USA). Conservation Biology 6, 253–262.

Urban, M. C. 2006. Road facilitation of trematode infections in snails of northern Alaska. Conservation Biology 20, 1143–1149.

van der Ree, R., Jaeger, J. A. G., van der Grift, E. A. and Clevenger, A. P. 2011. Effects of roads and traffic on wildlife populations and landscape function: road ecology is moving toward larger scales. Ecology and Society 16, 48–57.

van Vierssen Trip, N. and Wiersma, Y. F. In review. A comparison of all-terrain vehicle (ATV) trail impacts on boreal habitats across scales. Natural Areas Journal.

Vickery, W. L. and Bider, J. R. 1978. The effect of weather on *Sorex cinereus* activity. Canadian Journal of Zoology 56, 291–297.

Whitaker, J. O. 2004. *Sorex cinereus*. Mammalian Species 743, 1–9.

Wilson, D. E. and Reeder, M. E. 2011. Mammal Species of the World, third ed. Retrieved March 10, 2013 from http://www.vertebrates.si.edu/msw/mswcfapp/msw/index.cfm

Wilson, B. A., Robertson, D., Moloney, D. J., Newell, G. R. and Laidlaw, W. S. 1999. Factors affecting small mammal distribution and abundance in the eastern Otway Ranges, Victoria. Proceedings of the Ecological Society of Australia 16, 379–396.

Wulder, M. and Nelson, T. 2003. EOSD Land Cover Classification Legend Report.

Yahner, R. H., Yahner, R. T. and Hutnik, R. J. 2007. Long-term trends in small mammals on a right-of-way in Pennsylvania, U.S. Arboriculture and Urban Forestry 33, 147–152.

**Table 1.** Native, introduced and colonized terrestrial mammals of the island of Newfoundland, Newfoundland and Labrador, Canada showing the disproportionate native terrestrial mammalian predator-prey ratio and high number of rodent introductions or colonizations (after McCue 2012).

| Order | Scientific Species Name | Common Species Name | Origin |
|---|---|---|---|
| Carnivora | *Canis lupus* | Wolf† | Native |
| | *Vulpes vulpes* | Red fox | Native |
| | *Ursus americanus* | American black bear | Native |
| | *Martes americana* | American marten | Native |
| | *Mustela erminea* | Ermine | Native |
| | *Lontra canadensis* | Northern river otter | Native |
| | *Lynx canadensis* | Canadian lynx | Native |
| | *Neovison vison* | American mink | Introduced, 1935 |
| | *Canis lupus* | Coyote | Colonized, 1985 |
| | *Felis catus* | Domestic cat | Introduced |
| Rodentia | *Castor canadensis* | American beaver | Native |
| | *Microtus pennsylvanicus* | Meadow vole | Native |
| | *Ondatra zibethicus* | Common muskrat | Native |
| | *Tamias striatus* | Eastern chipmunk | Introduced, 1962 |
| | *Tamiasciurus hudsonicus* | Red squirrel | Introduced, 1963 |
| | *Rattus norvegicus* | Brown rat | Introduced |
| | *Myodes glareolus* | Bank vole | Introduced, 1967 |
| | *Peromyscus maniculatus* | Deermouse | Introduced, 1968 |

| | | | |
|---|---|---|---|
| | *Myodes gapperi* | Southern red-backed vole | Colonized, 1998 |
| | *Mus musculus* | House mouse | Introduced |
| Insectivora | *Sorex cinereus* | Cinereus Shrew | Introduced, 1958 |
| Artiodactyla | *Rangifer caribou* | Caribou | Native |
| | *Alces americanus* | American moose | Introduced, 1904 |
| Lagomorpha | *Lepus arcticus* | Arctic hare | Native |
| | *Lepus americanus* | Snowshoe hare | Introduced, 1860 |
| Chiroptera | *Myotis lucifugus* | Little brown myotis | Native |
| | *Myotis septentrionalis* | Northern myotis | Native |

† Extinct

**Table 2.** Represented EOSD cover types in Three Pond Barrens, Pippy Park, NL. Definitions taken from the EOSD land cover classification legend report (Wulder and Nelson 2003).

| Cover type | Description |
|---|---|
| Water | Lakes, reservoirs, rivers, streams, or salt water. |
| Rock | Bedrock, rubble, talus, blockfield, rubbley mine spoils, or lava beds. |
| Exposed | <5% vegetation. River sediments, exposed soils, pond or lake sediments, reservoir margins, beaches, landings, burned areas, road surfaces, mudflat sediments, cutbanks, moraines, gravel pits, tailings, railway surfaces, buildings and parking, or other non-vegetated surfaces. |
| Shrub tall | At least 20% ground cover which is at least one-third shrub. Average shrub height > = to 2 m. |
| Shrub low | At least 20% ground cover which is at least one-third shrub. Average shrub height < 2 m. |
| Wetland-shrub | Land with a water table near, at, or above the soil surface for enough time to promote wetland or aquatic processes. The majority of vegetation is shrub. |
| Wetland-herb | Land with a water table near, at, or above the soil surface for enough time to promote wetland or aquatic processes. The majority of vegetation is herb. |
| Herbs | Vascular plant without woody stem (grasses, crops, forbs, gramminoids). Minimum of 20% ground cover **or** one-third of total vegetation must be herb. |
| Coniferous-dense | > 60% crown closure. Coniferous trees are 75% or more of total basal area. |
| Coniferous-open | 26-60% crown closure. Coniferous trees are 75% or more of total basal area. |

| | |
|---|---|
| Coniferous-sparse | 10-25% crown closure. Coniferous trees are 75% or more of total basal area. |
| Broadleaf-dense | > 60% crown closure. Broadleaf trees are 75% or more of total basal area. |
| Broadleaf-open | 26-60% crown closure. Broadleaf trees are 75% or more of total basal area. |
| Broadleaf-sparse | 10-25% crown closure. Broadleaf trees are 75% or more of total basal area. |
| Mixedwood-dense | > 60% crown closure. Neither coniferous nor broadleaf trees account for 75% or more of total basal area. |
| Mixedwood-open | 26-60% crown closure. Neither coniferous nor broadleaf trees account for 75% or more of total basal area. |

**Table 3.** Description of the protocol used to measure each microhabitat variable within the small mammal microhabitat sample plots. Log, tree stump, dead tree and shrub density, and woody stem and herbaceous stem cover protocols were adopted from Rodgers et al. (2008). Shrub's Simpson's Diversity Index, grass and sedge, moss or lichen, rock or ground and litter cover and percent canopy open were modifications of the surface component cover, shrub density and canopy closure protocols described in Rodgers et al. (2008).

| Measurement | Symbol | Unit | Sampling technique |
|---|---|---|---|
| Percent canopy open | PCO | % | Measured using a densitometer held at elbow height in the 4 cardinal directions surrounding the trap site. |
| Log density | LD | # | Number of logs ≥ 1.0 m in length and ≥ 4 cm diameter in each quarter of the circular plot. |
| Tree stump density | SD | # | Number of tree stumps ≤ 1 m in height and ≥ 4 cm diameter in each quarter of the circular plot. |
| Dead tree density | DTD | # | Number of dead trees > 1.0 m in height in each quarter of the circular plot. Tree defined as dead if missing more than 80% needles or angled < 45° from original standing position. |
| Shrub density | SHD | # | Estimated by carrying a metre stick (folded up 1 m square plot) along transect lines at breast height (approximately 1.3 m above ground) and counting the number of woody stems < 5 cm in diameter, and > 0.40 m but ≤ 2.0 m in height, that are intersected. Counted stems include only the main stem. |
| Shrub's Simpson's Diversity Index (1-D) | SHDV | | Estimated by carrying a metre stick (folded up 1 m square plot) along transect lines at breast height (approximately 1.3 m above ground) and recording the species of woody stems < 5 cm in diameter, and > 0.40 m but ≤ 2.0 m in height, that are intersected. Diversity of species was calculated using the formula found in methods. |
| Woody stem cover | GW | % | Percent of the 1-$m^2$ quadrant covered by each live woody-stemmed species < 0.4 m in height. |

| | | | |
|---|---|---|---|
| Herbaceous stem cover | GH | % | Percent of the 1-m$^2$ quadrant covered by each live herbaceous-stemmed species $< 0.4$ m in height. |
| Grass and sedge cover | GG | % | Percent of the 1-m$^2$ quadrant covered by grass or sedge. |
| Moss or lichen cover | GML | % | Percent of the 1-m$^2$ quadrant covered by moss or lichen |
| Rock or ground cover | GR | % | Percent of the 1-m$^2$ quadrant covered by rock or bare ground. |
| Litter cover | GL | % | Percent of the 1-m$^2$ quadrant covered by decayed wood and leaf/needle litter. |

**Table 4.** *S. cinereus* mass has the strongest correlation with body length on a log-log scale as revealed by the Pearson correlation coefficients between mass and the body measurements body length, left ear length, left hind foot length and tail length.

| **Body Measurement** | **r** | **lower 95% C.I.** | **higher 95% C.I.** | **p-value** |
|---|---|---|---|---|
| Body length | 0.53 | 0.37 | 0.67 | 5.389e-08 |
| Left ear length | 0.18 | -0.03 | 0.37 | 0.088 |
| Left hind foot length | 0.10 | -0.11 | 0.30 | 0.331 |
| Tail length | -0.27 | -0.45 | -0.07 | 0.010 |

**Table 5.** Decomposition of variability per component in the principal components analysis of microhabitat variables. The first three components have eigenvalues greater than 1.0 and account for 87.9% of the cumulative explained variance and are therefore the only components retained for interpretation.

| Component | Eigenvalue Variance | Percentage of variance | Cumulative percentage of variance |
|---|---|---|---|
| 1 | 6.11 | 50.9 | 50.9 |
| 2 | 3.06 | 25.5 | 76.4 |
| 3 | 1.38 | 11.5 | 87.9 |
| 4 | 0.66 | 5.54 | 93.4 |
| 5 | 0.33 | 2.75 | 96.2 |
| 6 | 0.29 | 2.40 | 98.6 |
| 7 | 0.17 | 1.44 | 100 |

**Table 6.** Correlation between microhabitat variables and the first three components in Principal Components Analysis. PC 1 is strongly positively correlated with stump density (SD), log density (LD), dead tree density (DTD) and percent ground cover in litter (GL), and strongly negatively correlated with percent canopy open (PCO) and percent ground cover in grass (GG). PC 2 can be characterized by a positive correlation with shrub diversity (SHDV) and shrub density (SHD) and PC 3 is characterized by percent ground cover in rock (GR).

| PC | Microhabitat Variable | Correlation | P-value |
|---|---|---|---|
| 1 | SD | 0.921853 | 0.001124 |
| | LD | 0.861682 | 0.005948 |
| | GL | 0.833045 | 0.010226 |
| | DTD | 0.791553 | 0.01925 |
| | GG | -0.78957 | 0.019774 |
| | PCO | -0.95683 | 0.000195 |
| 2 | SHDV | 0.872135 | 0.004738 |
| | SHD | 0.758079 | 0.029285 |
| 3 | GR | -0.96045 | 0.00015 |

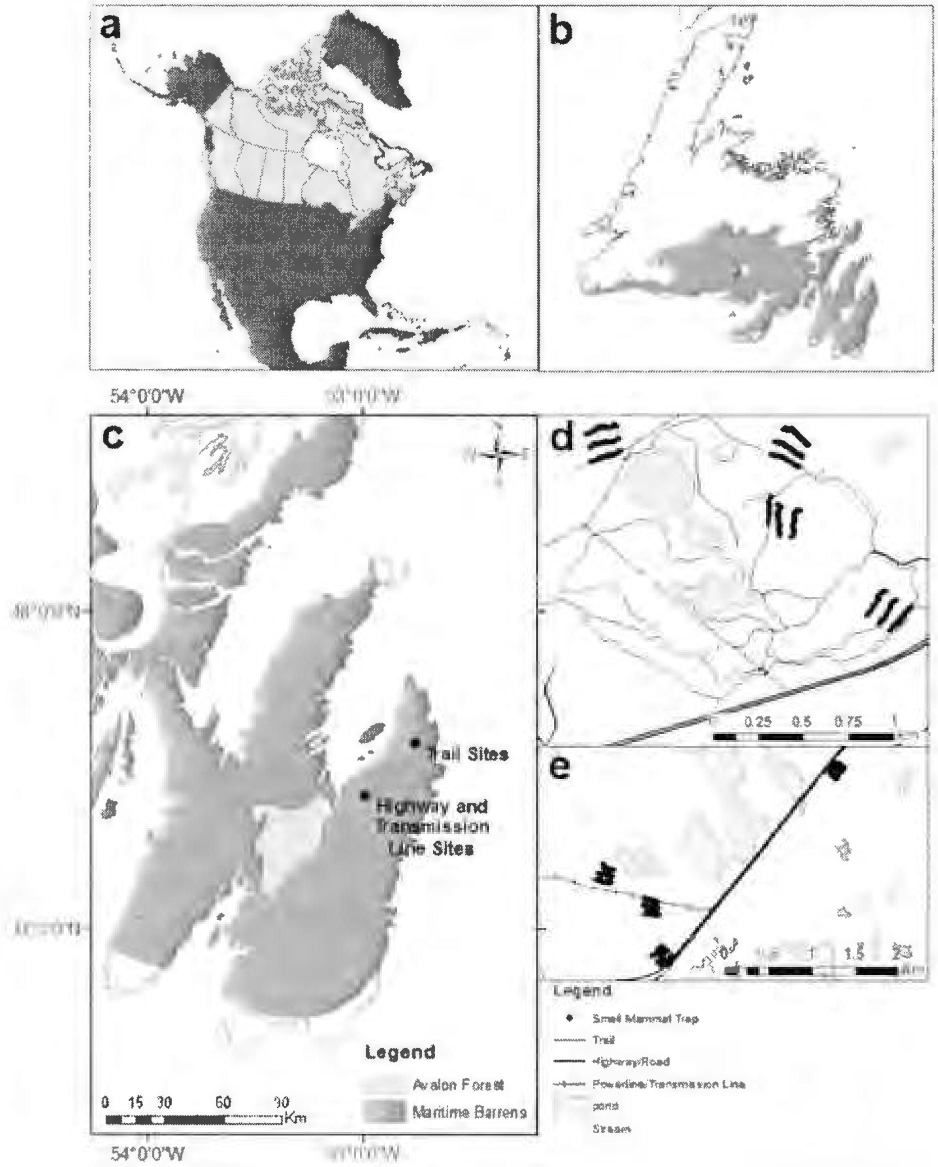

**Figure 1.** Location of study areas on the Avalon Peninsula of Newfoundland, Canada. (a) The province of Newfoundland and Labrador is highlighted in white. (b) The extent of the Maritime barrens Ecoregion within the island of Newfoundland is in dark grey. (c) The Avalon Peninsula of Newfoundland showing the location of the two study areas: (d) Three Pond Barrens, Pippy Park, NL showing the trail system and location of four trapping sites adjacent to trails, and (e) the Trans Canada Highway outside Butterpot Provincial Park, NL showing the location of four trapping sites, two adjacent to the highway and two adjacent to a transmission line that runs perpendicular to the highway.

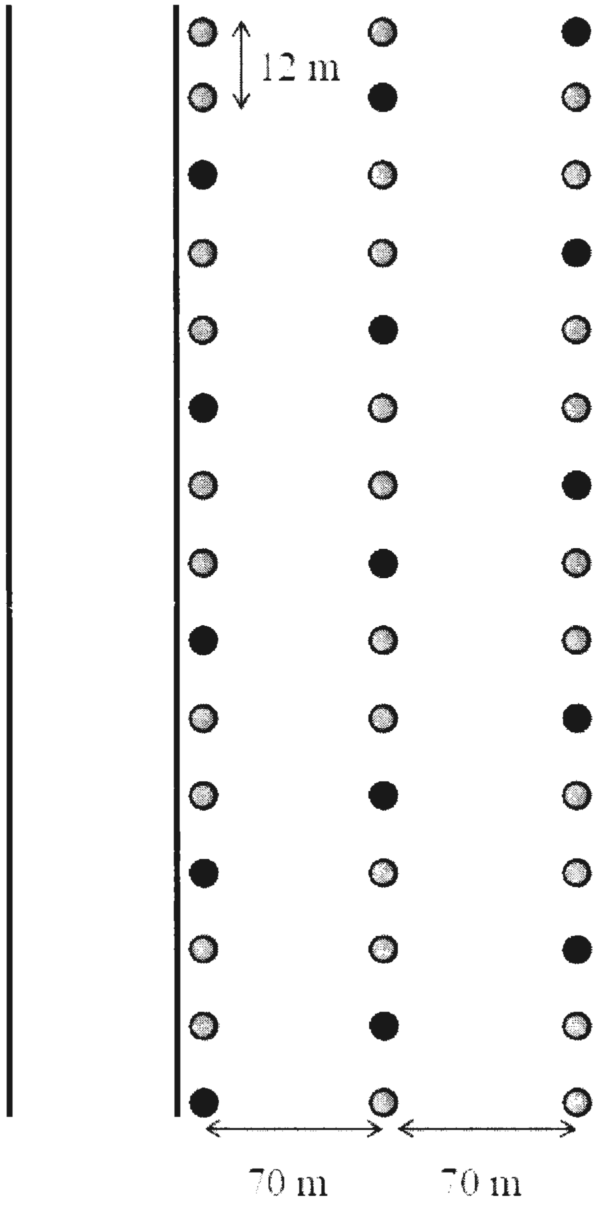

**Figure 2.** Study design schematic. Vertical lines are the boundaries of a linear feature. Circles are small Sherman live traps. Black circles are trap stations also used for microhabitat sampling. Trapping lines ran parallel to the linear feature and were spaced 70-m apart. Traps along a line were spaced 12-m apart.

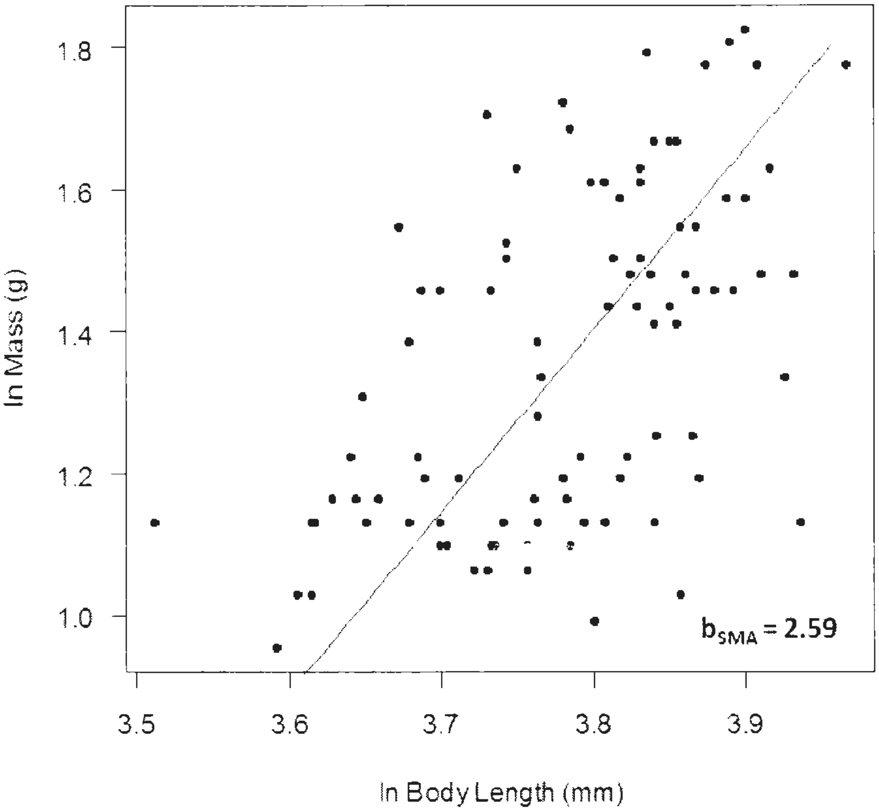

**Figure 3.** The scaling exponent $b_{SMA}$ used to calculate the scaled mass index for *Sorex cinereus* sample specimens was estimated by fitting a standard major axis regression to ln-transformed mass and body length data. Regression resulted in a slope ($b_{SMA}$) of 2.59 (n=94).

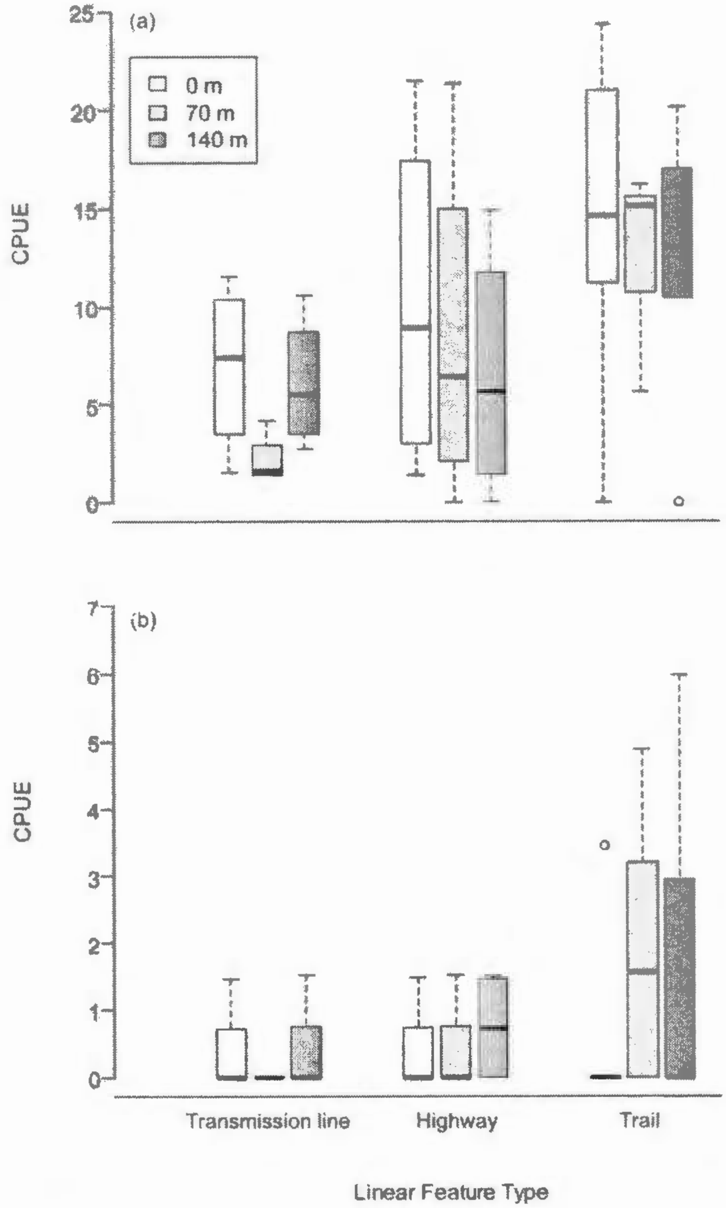

**Figure 4.** Box plots of catch per unit effort (CPUE) of (a) *S. cinereus* and (b) *M. pennsylvanicus* along three types of linear features at three distances from the feature (white: 0 m (adjacent to linear feature); light grey: 70 m away from linear feature edge, dark grey: 140 m away from linear feature edge). CPUE of both species was highest in sites adjacent to a trail compared to sites adjacent to a transmission line or highway and did not vary with proximity to either type of linear feature. Dark horizontal line shows median, boxes outline the first and third quartiles, error bars show 1.5 x interquartile limits, and open circles are outliers.

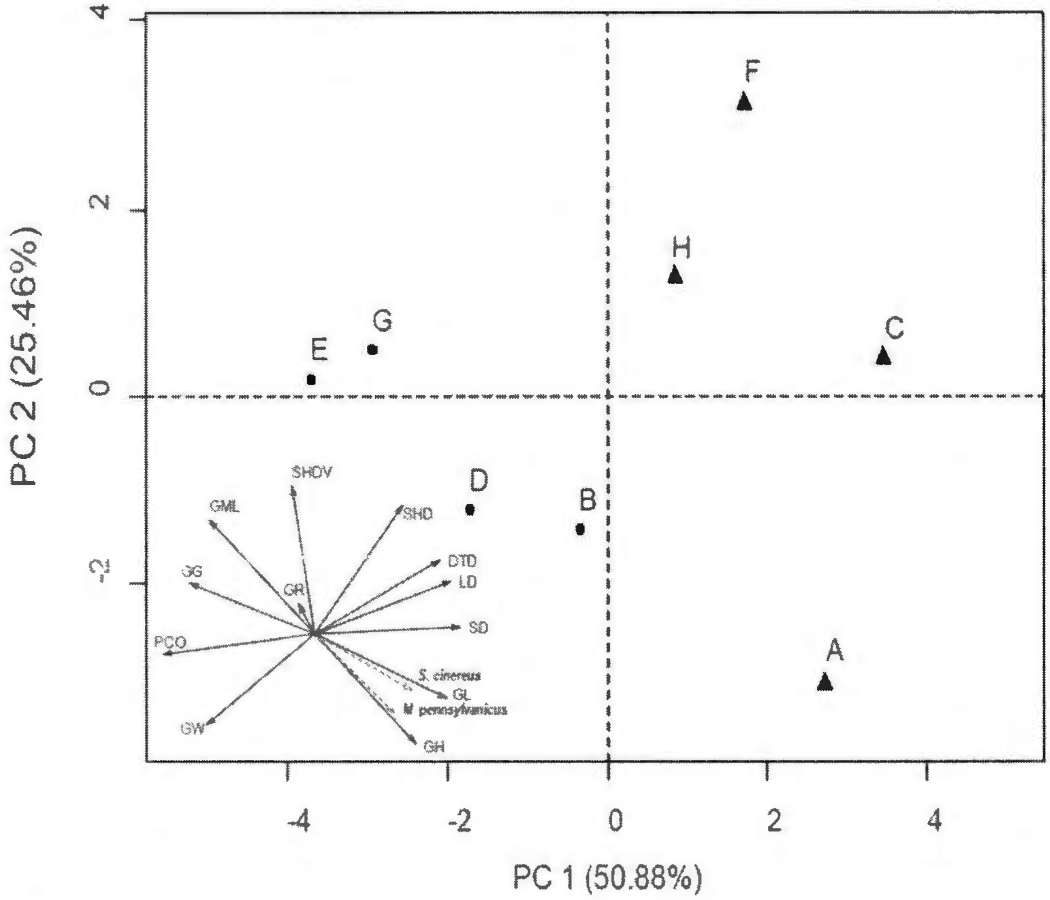

**Figure 5.** Principal Components Analysis of microhabitat variables measured at each sample site with *M. pennsylvanicus* and *S. cinereus* CPUE as supplementary variables (dashed lines). Component one explained 50.9% of the variance and component two explained 24.5% of the variance. Refer to Table 3 for variable abbreviations. The abundance of *S. cinereus* and *M. pennsylvanicus* are both positively correlated with the first PC. *S. cinereus* abundance is mainly associated with high abundances of litter, while *M. pennsylvanicus* is associated with a high percentage of herbaceous ground cover. Forest sites are represented by triangles and barren sites as circles. Sites A-D are trail sites, E-F are transmission line sites and G-H are highway sites.

## Appendix A. Small mammal capture data

**Table A1.** Total number of *M. pennsylvanicus* and *S. cinereus* captures. Cycle-night refers to the trapping cycle and trap night within that cycle. Cycle one was conducted between 3 July and 2 August 2011. Cycle two was conducted between 8 August and 3 September 2011. Trapping nights are not necessarily consecutive due to impeding weather conditions. Sites A and C were forest habitats, and sites B and D were barren habitats, adjacent to trails in Pippy Park, NL. Site E is a barren habitat and site F is a forest habitat adjacent to a transmission line corridor. Site G is a barren habitat and site H is a forest habitat adjacent to the Trans Canada Highway. Sites E, F, G and H were all located near Butter Pot Provincial Park, NL (refer to Fig. 1).

| **Cycle – Night** | **Site** | **Total captures** | |
|---|---|---|---|
| | | ***M. pennsylvanicus*** | ***S. cinereus*** |
| 1-1 | A | 0 | 4 |
| 1-2 | A | 0 | 5 |
| 1-3 | A | 1 | 10 |
| 1-4 | A | 0 | 7 |
| 1-5 | A | 1 | 5 |
| 2-1 | A | 2 | 7 |
| 2-2 | A | 0 | 7 |
| 2-3 | A | 0 | 6 |
| 2-4 | A | 2 | 8 |
| 2-5 | A | 0 | 3 |
| 1-1 | B | 0 | 4 |
| 1-2 | B | 1 | 8 |
| 1-3 | B | 2 | 5 |
| 1-4 | B | 0 | 4 |
| 1-5 | B | 2 | 4 |
| 2-1 | B | 0 | 5 |
| 2-2 | B | 3 | 1 |
| 2-3 | B | 0 | 10 |
| 2-4 | B | 0 | 9 |

| | | | |
|---|---|---|---|
| 2-5 | B | 1 | 3 |
| 1-1 | C | 0 | 1 |
| 1-2 | C | 0 | 2 |
| 1-3 | C | 0 | 13 |
| 1-4 | C | 0 | 9 |
| 1-5 | C | 0 | 5 |
| 2-1 | C | 1 | 8 |
| 2-2 | C | 0 | 8 |
| 2-3 | C | 0 | 9 |
| 2-4 | C | 1 | 4 |
| 2-5 | C | 0 | 6 |
| 1-1 | D | 0 | 0 |
| 1-2 | D | 1 | 0 |
| 1-3 | D | 0 | 0 |
| 1-4 | D | 0 | 4 |
| 1-5 | D | 1 | 4 |
| 2-1 | D | 0 | 1 |
| 2-2 | D | 0 | 3 |
| 2-3 | D | 0 | 4 |
| 2-4 | D | 0 | 7 |
| 2-5 | D | 0 | 3 |
| 1-1 | E | 0 | 0 |
| 1-2 | E | 0 | 0 |
| 1-3 | E | 0 | 1 |
| 1-4 | E | 0 | 0 |
| 1-5 | E | 0 | 8 |
| 2-1 | E | 0 | 0 |
| 2-2 | E | 0 | 4 |
| 2-3 | E | 0 | 0 |
| 2-4 | E | 1 | 7 |

|  |  |  |  |
|---|---|---|---|
| 2-5 | E | 0 | 1 |
| 1-1 | F | 0 | 0 |
| 1-2 | F | 0 | 0 |
| 1-3 | F | 0 | 1 |
| 1-4 | F | 0 | 3 |
| 1-5 | F | 0 | 2 |
| 2-1 | F | 0 | 0 |
| 2-2 | F | 0 | 1 |
| 2-3 | F | 0 | 3 |
| 2-4 | F | 0 | 7 |
| 2-5 | F | 1 | 3 |
| 1-1 | G | 0 | 0 |
| 1-2 | G | 0 | 1 |
| 1-3 | G | 0 | 0 |
| 1-4 | G | 0 | 2 |
| 1-5 | G | 0 | 2 |
| 2-1 | G | 0 | 1 |
| 2-2 | G | 0 | 6 |
| 2-3 | G | 0 | 4 |
| 2-4 | G | 0 | 7 |
| 2-5 | G | 1 | 7 |
| 1-1 | H | 0 | 0 |
| 1-2 | H | 0 | 1 |
| 1-3 | H | 0 | 0 |
| 1-4 | H | 0 | 0 |
| 1-5 | H | 0 | 3 |
| 2-1 | H | 0 | 5 |
| 2-2 | H | 1 | 7 |
| 2-3 | H | 1 | 5 |
| 2-4 | H | 0 | 6 |

| | | | |
|---|---|---|---|
| 2-5 | H | 1 | 10 |
| TOTAL | | 25 | 314 |

**Table A2.** Catch per unit effort (CPUE) of *S. cinereus* by site, type of linear feature, line and habitat type. CPUE was corrected for sprung traps by following the methods of Nelson and Clark (1973). Cycle one was conducted between 3 July and 2 August 2011. Cycle two was conducted between 8 August and 3 September 2011.

| Site | Linear feature | Line (m) | Habitat | Catch/unit effort | |
|---|---|---|---|---|---|
| | | | | Cycle 1 | Cycle 2 |
| A | trail | 0 | Forest | 24.348 | 12.308 |
| A | trail | 70 | Forest | 14.754 | 15.625 |
| A | trail | 140 | Forest | 13.223 | 20.155 |
| B | trail | 0 | Barren | 12.069 | 16.923 |
| B | trail | 70 | Barren | 16.260 | 15.625 |
| B | trail | 140 | Barren | 12.903 | 10.448 |
| C | trail | 0 | Forest | 20.635 | 21.374 |
| C | trail | 70 | Forest | 10 | 15.504 |
| C | trail | 140 | Forest | 17.886 | 16.176 |
| D | trail | 0 | Barren | 0 | 10.370 |
| D | trail | 70 | Barren | 11.511 | 5.634 |
| D | trail | 140 | Barren | 0 | 10.448 |
| E | transmission line | 0 | Barren | 5.479 | 11.594 |
| E | transmission line | 70 | Barren | 4.138 | 1.3699 |
| E | transmission line | 140 | Barren | 2.703 | 4.196 |
| F | transmission line | 0 | Forest | 1.527 | 9.231 |
| F | transmission line | 70 | Forest | 1.681 | 1.429 |
| F | transmission line | 140 | Forest | 6.838 | 10.606 |
| G | highway | 0 | Barren | 4.511 | 21.488 |
| G | highway | 70 | Barren | 0 | 8.633 |
| G | highway | 140 | Barren | 2.703 | 8.633 |
| H | highway | 0 | Forest | 1.379 | 13.333 |
| H | highway | 70 | Forest | 4.138 | 21.374 |
| H | highway | 140 | Forest | 0 | 14.925 |

**Table A3.** Catch per unit effort (CPUE) of *M. pennsylvanicus* by site, type of linear feature, line and habitat type. CPUE was corrected for sprung traps by following the methods of Nelson and Clark (1973). Cycle one was conducted between 3 July and 2 August 2011. Cycle two was conducted between 8 August and 3 September 2011.

| | | | | **Catch/unit effort** | |
|---|---|---|---|---|---|
| **Site** | **Linear feature** | **Line** | **Habitat** | **Cycle 1** | **Cycle 2** |
| A | trail | 0 | Forest | 0 | 0 |
| A | trail | 70 | Forest | 3.279 | 3.125 |
| A | trail | 140 | Forest | 0 | 3.101 |
| B | trail | 0 | Barren | 3.448 | 0 |
| B | trail | 70 | Barren | 4.878 | 0 |
| B | trail | 140 | Barren | 0 | 5.970 |
| C | trail | 0 | Forest | 0 | 0 |
| C | trail | 70 | Forest | 0 | 3.100 |
| C | trail | 140 | Forest | 0 | 0 |
| D | trail | 0 | Barren | 0 | 0 |
| D | trail | 70 | Barren | 0 | 0 |
| D | trail | 140 | Barren | 2.778 | 0 |
| E | transmission line | 0 | Barren | 0 | 1.449 |
| E | transmission line | 70 | Barren | 0 | 0 |
| E | transmission line | 140 | Barren | 0 | 0 |
| F | transmission line | 0 | Forest | 0 | 0 |
| F | transmission line | 70 | Forest | 0 | 0 |
| F | transmission line | 140 | Forest | 0 | 1.515 |
| G | highway | 0 | Barren | 0 | 0 |
| G | highway | 70 | Barren | 0 | 0 |
| G | highway | 140 | Barren | 0 | 1.439 |
| H | highway | 0 | Forest | 0 | 1.481 |
| H | highway | 70 | Forest | 0 | 1.527 |
| H | highway | 140 | Forest | 0 | 1.493 |

## Appendix B. *S. cinereus* Body Measurements Data

**Table B1.** Raw body measurements data for *S. cinereus* mortalities by site and line. Specimens were collected between 3 July and 3 September 2011. Sites A and C were forest habitats, and sites B and D were barren habitats, adjacent to trails in Pippy Park, NL. Site E is a barren habitat and site F is a forest habitat adjacent to a transmission line corridor. Site G is a barren habitat and site H is a forest habitat adjacent to the Trans Canada Highway. Sites E, F, G and H were all located near Butter Pot Provincial Park, NL (refer to Fig. 1).

| Record number | Site | Line | Mass (g) | Body length (mm) | Left ear length (mm) | Left hind foot length (mm) | Tail length (mm) |
|---|---|---|---|---|---|---|---|
| 318 | A | 0 | 6.0 | 46.3 | 4.5 | 11.1 | 34.1 |
| 319 | A | 0 | 4.3 | 49.0 | 4.0 | 10.9 | 42.3 |
| 324 | A | 0 | 4.5 | 45.3 | 3.5 | 10.6 | 42.1 |
| 174 | A | 0 | 4.4 | 47.5 | 4.1 | 10.4 | 41.1 |
| 185 | A | 0 | 4.1 | 46.5 | 4.2 | 11.1 | 39.0 |
| 325 | A | 0 | 4.6 | 42.2 | 3.7 | 10.5 | 40.6 |
| 176 | A | 0 | 3.4 | 45.7 | 4.1 | 10.9 | 40.4 |
| 159 | A | 0 | 4.5 | 46.1 | 4.1 | 11.1 | 37.6 |
| 160 | A | 70 | 6.2 | 49.4 | 4.9 | 10.6 | 37.9 |
| 328 | A | 70 | 4.2 | 46.0 | 3.6 | 10.7 | 39.5 |
| 336 | A | 70 | 5.0 | 46.1 | 3.5 | 11.1 | 37.9 |
| 178 | A | 70 | 3.1 | 45.0 | 2.9 | 10.6 | 36.8 |
| 322 | A | 70 | 4.7 | 39.3 | 4.1 | 10.8 | 35.8 |
| 337 | A | 70 | 3.1 | 43.1 | 3.6 | 10.7 | 40.6 |
| 150 | A | 70 | 3.6 | 43.1 | 3.4 | 11.2 | 40.2 |
| 320 | A | 70 | 3.2 | 38.8 | 4.2 | 10.9 | 39.1 |
| 149 | A | 70 | 4.3 | 48.4 | 3.2 | 11.0 | 41.3 |
| 327 | A | 70 | 3.1 | 44.4 | 4.3 | 10.9 | 40.7 |
| 177 | A | 70 | 2.8 | 37.1 | 2.9 | 10.1 | 39.1 |
| 344 | A | 140 | 5.9 | 52.8 | 3.3 | 10.7 | 40.2 |
| 311 | A | 140 | 3.2 | 37.6 | 3.4 | 11.2 | 37.9 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| 308 | A | 140 | 3.2 | 43.0 | 3.5 | 10.8 | 42.5 |
| 179 | A | 140 | 3.8 | 50.7 | 4.6 | 10.9 | 41.7 |
| 340 | A | 140 | 4.9 | 48.8 | 3.7 | 10.3 | 35.6 |
| 167 | A | 140 | 4.2 | 47.0 | 4.6 | 11.0 | 39.4 |
| 330 | A | 140 | 4.3 | 47.8 | 4.0 | 10.5 | 38.2 |
| 166 | A | 140 | 2.7 | 44.7 | 4.5 | 10.4 | 41.6 |
| 202 | B | 0 | 5.1 | 42.5 | 3.5 | 10.9 | 37.1 |
| 201 | B | 0 | 4.4 | 46.4 | 3.5 | 10.4 | 37.9 |
| 234 | B | 70 | 2.9 | 42.8 | 4.3 | 11.4 | 46.4 |
| 253 | B | 140 | 3.4 | 39.8 | 4.1 | 10.9 | 42.8 |
| 254 | B | 140 | 4.9 | 49.4 | 4.0 | 10.4 | 39.7 |
| 236 | B | 140 | 4.3 | 39.9 | 4.2 | 10.8 | 41 |
| 265 | B | 140 | 5.1 | 46.1 | 5.0 | 11.3 | 39.3 |
| 207 | C | 0 | 5.3 | 46.5 | 4.2 | 10.9 | 35.3 |
| 257 | C | 0 | 3.1 | 46.5 | 3.7 | 10.7 | 41.4 |
| 220 | C | 0 | 5.1 | 50.2 | 4.1 | 10.8 | 42.6 |
| 209 | C | 0 | 5.9 | 49.8 | 4.2 | 10.9 | 40.8 |
| 255 | C | 0 | 3.0 | 41.8 | 3.1 | 10.6 | 42.9 |
| 239 | C | 70 | 3.1 | 37.2 | 3.6 | 10.7 | 39.5 |
| 222 | C | 70 | 4.3 | 40.4 | 3.3 | 10.2 | 34.9 |
| 259 | C | 70 | 3.1 | 38.5 | 3.7 | 9.70 | 38.4 |
| 272 | C | 70 | 2.8 | 47.3 | 4.0 | 10.9 | 37.8 |
| 212 | C | 70 | 5.4 | 44.0 | 4.7 | 10.2 | 40.9 |
| 240 | C | 70 | 5.3 | 47.0 | 3.7 | 11.4 | 39.5 |
| 245 | C | 140 | 4.4 | 51.0 | 4.3 | 10.6 | 39.5 |
| 103 | C | 140 | 3.1 | 33.5 | 4.8 | 10.5 | 42.3 |
| 260 | C | 140 | 3.0 | 42.8 | 3.7 | 11.0 | 40.7 |
| 226 | C | 140 | 3.3 | 43.8 | 3.6 | 10.1 | 44.4 |
| 333 | D | 0 | 5.5 | 41.7 | 3.8 | 10.7 | 38.6 |
| 332 | D | 0 | 4.5 | 42.2 | 3.0 | 11.2 | 39.4 |

|  |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|
| 346 | D | 0 | 4.0 | 43.1 | 3.6 | 10.5 | 38.5 |
| 313 | D | 70 | 2.6 | 36.3 | 3.7 | 11.1 | 37.6 |
| 170 | D | 70 | 4.4 | 45.8 | 3.8 | 10.5 | 39.0 |
| 169 | D | 70 | 4.2 | 45.1 | 3.2 | 10.8 | 40.6 |
| 357 | D | 70 | 3.0 | 41.9 | 3.7 | 10.9 | 42.2 |
| 181 | D | 70 | 4.9 | 45.5 | 4.3 | 11.1 | 40.4 |
| 351 | D | 140 | 3.4 | 38.1 | 3.2 | 10.7 | 40.8 |
| 140 | E | 0 | 3.3 | 47.9 | 4.7 | 10.8 | 42.2 |
| 276 | E | 0 | 3.0 | 44.0 | 3.3 | 10.3 | 42.4 |
| 133 | E | 0 | 4.4 | 49.9 | 4.3 | 10.4 | 41.4 |
| 283 | E | 0 | 4.7 | 47.8 | 3.4 | 10.3 | 37.2 |
| 141 | E | 70 | 3.2 | 38.2 | 4.0 | 10.9 | 41.8 |
| 288 | E | 70 | 3.1 | 37.1 | 4.0 | 11.3 | 38.9 |
| 145 | E | 140 | 4.3 | 41.8 | 4.2 | 11.2 | 41.3 |
| 279 | F | 0 | 5.0 | 45.0 | 3.7 | 11.4 | 42.8 |
| 290 | F | 0 | 5.3 | 47.2 | 4.0 | 10.3 | 39.9 |
| 291 | F | 0 | 3.3 | 40.9 | 4.7 | 10.6 | 36.8 |
| 299 | F | 0 | 3.3 | 40.0 | 4.3 | 10.8 | 40.9 |
| 135 | F | 0 | 2.9 | 42.8 | 3.0 | 10.6 | 40.8 |
| 280 | F | 0 | 5.6 | 43.8 | 3.0 | 10.8 | 39.3 |
| 146 | F | 70 | 5.0 | 44.6 | 3.9 | 11.3 | 41.9 |
| 292 | F | 140 | 4.1 | 47.2 | 4.2 | 10.9 | 41.0 |
| 295 | F | 140 | 6.1 | 48.9 | 3.3 | 10.6 | 38.1 |
| 296 | F | 140 | 2.8 | 36.8 | 3.8 | 10.4 | 41.6 |
| 293 | F | 140 | 3.5 | 46.6 | 3.6 | 11.1 | 45.1 |
| 278 | F | 140 | 2.9 | 41.3 | 3.5 | 10.3 | 41.6 |
| 381 | G | 0 | 3.1 | 39.6 | 3.6 | 10.7 | 36.1 |
| 405 | G | 0 | 3.1 | 42.1 | 3.4 | 10.7 | 42.8 |
| 365 | G | 0 | 4.0 | 39.6 | 3.7 | 10.5 | 39.3 |
| 393 | G | 70 | 3.1 | 51.2 | 4.7 | 12.0 | 41.2 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| 406 | G | 70 | 3.8 | 43.2 | 3.5 | 10.4 | 38.2 |
| 392 | G | 70 | 3.0 | 42.8 | 3.4 | 10.9 | 41.1 |
| 408 | G | 70 | 3.0 | 40.4 | 3.5 | 10.4 | 36.5 |
| 197 | G | 140 | 3.0 | 40.6 | 4.1 | 10.1 | 38.5 |
| 382 | G | 140 | 3.3 | 45.5 | 4.4 | 10.7 | 41.4 |
| 198 | H | 0 | 3.1 | 40.4 | 3.5 | 11.3 | 41.8 |
| 414 | H | 0 | 3.4 | 44.3 | 3.9 | 10.4 | 36.8 |
| 412 | H | 0 | 2.9 | 41.7 | 3.7 | 10.8 | 41.0 |
| 199 | H | 70 | 3.2 | 43.9 | 3.6 | 10.5 | 41.0 |
| 375 | H | 70 | 3.5 | 47.7 | 4.1 | 11.0 | 40.7 |
| 421 | H | 140 | 4.7 | 47.3 | 3.2 | 10.3 | 41.9 |
| 389 | H | 140 | 5.9 | 48.1 | 3.9 | 11.2 | 38.9 |
| 402 | H | 140 | 3.7 | 38.4 | 4.2 | 9.30 | 42.9 |

## Appendix C. Principal Components Analysis

**Table C1.** Correlations between all microhabitat variables measured and analyzed in the PCA. All microhabitat variables were left in the PCA. Variable abbreviations: PCO = percent canopy open, SD = stump density, LD = log density, DTD = dead tree density, SHDV = shrub diversity measured using Shannon-Weiner Diversity Index, SHD = shrub density, GW = percent ground covered by woody stem vegetation, GH = percent ground covered by herbaceous vegetation, GG = percent ground covered by grass, GR = percent ground covered by rock, GML = percent ground covered by moss or lichen and GL = percent ground covered by litter and woody debris.

| | **PCO** | **SD** | **LD** | **DTD** | **SHDV** | **SHD** | **GW** | **GH** | **GG** | **GR** | **GML** | **GL** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **PCO** | 1.000 | -0.882 | -0.785 | -0.744 | 0.002 | -0.634 | 0.695 | -0.452 | 0.795 | -0.143 | 0.600 | -0.817 |
| **SD** | -0.882 | 1.000 | 0.758 | 0.657 | -0.041 | 0.481 | -0.738 | 0.561 | -0.668 | -0.281 | -0.543 | 0.767 |
| **LD** | -0.785 | 0.758 | 1.000 | 0.981 | 0.122 | 0.674 | -0.665 | 0.438 | -0.598 | -0.239 | -0.290 | 0.473 |
| **DTD** | -0.744 | 0.657 | 0.981 | 1.000 | 0.216 | 0.755 | -0.638 | 0.285 | -0.547 | -0.127 | -0.168 | 0.380 |
| **SHDV** | 0.002 | -0.041 | 0.122 | 0.216 | 1.000 | 0.494 | -0.338 | -0.609 | 0.428 | 0.135 | 0.619 | -0.389 |
| **SHD** | -0.634 | 0.481 | 0.674 | 0.755 | 0.494 | 1.000 | -0.790 | -0.181 | -0.196 | 0.219 | 0.099 | 0.247 |
| **GW** | 0.695 | -0.738 | -0.665 | -0.638 | -0.338 | -0.790 | 1.000 | -0.158 | 0.207 | 0.051 | 0.128 | -0.361 |
| **GH** | -0.452 | 0.561 | 0.438 | 0.285 | -0.609 | -0.181 | -0.158 | 1.000 | -0.532 | -0.435 | -0.863 | 0.687 |
| **GG** | 0.795 | -0.668 | -0.598 | -0.547 | 0.428 | -0.196 | 0.207 | -0.532 | 1.000 | -0.128 | 0.669 | -0.750 |
| **GR** | -0.143 | -0.281 | -0.239 | -0.127 | 0.135 | 0.219 | 0.051 | -0.435 | -0.128 | 1.000 | -0.002 | 0.012 |
| **GML** | 0.600 | -0.543 | -0.290 | -0.168 | 0.619 | 0.099 | 0.128 | -0.863 | 0.669 | -0.002 | 1.000 | -0.885 |
| **GL** | -0.817 | 0.767 | 0.473 | 0.380 | -0.389 | 0.247 | -0.361 | 0.687 | -0.750 | 0.012 | -0.885 | 1.000 |

**Table C2.** Contribution of each small mammal trapping site to the construction of the first three components in the PCA of microhabitat variables. Sites A and C were forest habitats, and sites B and D were barren habitats, adjacent to trails in Pippy Park, NL. Site E is a barren habitat and site F is a forest habitat adjacent to a transmission line corridor. Site G is a barren habitat and site H is a forest habitat adjacent to the Trans Canada Highway. Sites E, F, G and H were all located near Butter Pot Provincial Park, NL (refer to Fig. 1).

| **Site** | **PC 1** | **PC 2** | **PC3** |
|---|---|---|---|
| A | 15.41 | 36.81 | 5.727 |
| B | 0.25 | 8.1 | 3.041 |
| C | 24.75 | 0.82 | 4.705 |
| D | 6.08 | 5.85 | 13.433 |
| E | 28.3 | 0.18 | 1.678 |
| F | 6 | 40.35 | 3.098 |
| G | 17.86 | 1.09 | 21.482 |
| H | 1.34 | 6.8 | 46.834 |

**Table C3.** Contribution of microhabitat variables to the construction of the first three components in the PCA (which cumulatively explains 87.9% of the variance). Variable abbreviations: PCO = percent canopy open, SD = stump density, LD = log density, DTD = dead tree density, SHDV = shrub diversity measured using Shannon-Weiner Diversity Index, SHD = shrub density, GW = percent ground covered by woody stem vegetation, GH = percent ground covered by herbaceous vegetation, GG = percent ground covered by grass, GR = percent ground covered by rock, GML = percent ground covered by moss or lichen and GL = percent ground covered by litter and woody debris.

| Variable | PC 1 | PC 2 | PC3 |
|---|---|---|---|
| PCO | 14.99 | 0.45 | 4.11 |
| SD | 13.92 | 0.06 | 1.89 |
| LD | 12.16 | 3.23 | 4.20 |
| DTD | 10.26 | 6.24 | 1.74 |
| SHDV | 0.34 | 24.89 | 0.07 |
| SHD | 5.11 | 18.81 | 1.73 |
| GW | 7.57 | 9.33 | 0.73 |
| GH | 6.67 | 13.55 | 7.10 |
| GG | 10.21 | 2.97 | 6.13 |
| GR | 0.17 | 1.04 | 66.70 |
| GML | 7.24 | 14.75 | 2.25 |
| GL | 11.37 | 4.68 | 3.38 |