A Comparison of All-Terrain Vehicle (ATV) Trail Impacts in Boreal Forest, Heath and Bog Habitats within the Avalon Wilderness Reserve and Surrounding Area

by

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Abstract

Recreational trails are a source of anthropogenic disturbance in nature reserves and other low human impact areas. Effective management must balance the desire of recreationists to use these natural areas with the need to maintain the ecological integrity of these areas. Low productivity environments may be particularly susceptible due to low resilience to recreational impacts. My study examined 28 all-terrain vehicle (ATV) trails within the Avalon Wilderness Reserve and the adjacent surrounding area in Newfoundland, Canada. My research showed that different habitat types (boreal forest, heaths and bogs) differ in resistance and resilience to both direct on-trail erosion and indirect off-trail vegetation impacts of ATV trails. Dry forested sites were more resistant to direct on-trail erosion but less resistant to indirect off-trail vegetation disturbance. Heath sites were less resistant to direct on-trail erosion but highly resistant to indirect off-trail disturbance. Bogs sites had low resistance to both direct and indirect trail disturbance. There have been limited studies on ATV trail impacts in boreal environments, and these findings provide guidance for managers in Newfoundland and Labrador to manage recreational vehicle use.

Keywords: Recreation Ecology, All-Terrain Vehicle, Off-Highway Vehicle, Off-Road Vehicle, Heath, Bog, Boreal Forest

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List of Symbols, Nomenclature or Abbreviations

A- statistic from multi-response permutation procedure (MRPP), is the chance-corrected within group agreement, a measure of group homogeneity

AIC- Akaike's Information Criterion

AIC_c- corrected AIC, used when sample size is small relative to number of parameters

ATV- All-Terrain Vehicle

AWR- Avalon Wilderness Reserve

CAN- Canadian

COHV- Canadian Off Highway Vehicle Distributors Council

Δi- AIC differences relative to the smallest AIC value among candidate models

D- Dry- one of the factors of this study, refers to dry sites

DEM- Digital Elevation Model

DNR- Department of Natural Resources (Newfoundland and Labrador)

EOSD- Earth Observation for Sustainable Development of Forests

F- Forest-one of the factors of this study, refers to forested sites

F/D- one of the factors of this study, refers to forested/dry sites

F/W- one the factors of this study, refers to forested/wet sites

GIS- Geographic Information System

GPS- Global Positioning System

GzLM- Generalized Linear Model

K- Number of Estimated Parameters in the Model

MBE- Maritime Barrens Ecoregion

MMIC- Motorcycle & Moped Industry Council

MRPP- Multi-Response Permutation Procedure

NMDS- Non-Metric Multidimensional Scaling

N/R- Non-Reserve -one of the factors of this study, refers to sites outside the Avalon Wilderness Reserve

NRCan- Natural Resources Canada

O- Open-one of the factors of this study, refers to open sites

O/D- Open/Dry -one the factors of this study, refers to open/dry sites

OHV- Off-Highway Vehicle

ORM-Off-Road Motorcycle

ORV- Off-Road Vehicle

O/W- Open/Wet-one of the factors of this study, refers to open/wet sites

PA- Protected Area

R- Reserve-one of the factors of this study, refers to sites inside the Avalon Wilderness Reserve

T- t statistic from multi-response permutation procedure, it is a measure of group separation

TCI- Topographic Convergence Index

TRMI- Topographic Relative Moisture Index

w_i- Akaike's weights, a measure of the probability that model *i* is the best model considered

W- Wet- one of the factors of this study, refers to wet sites

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1 Chapter 1: Introduction and Overview

1.1 Introduction

Species loss is a serious problem that needs to be appreciated and addressed not only by the scientific community but also by policy makers and the general public (Dirzo and Raven 2003). Two major contributing processes to the loss of biodiversity are habitat loss and habitat fragmentation (Andrén 1994; Parker and Mac Nally 2002; Cushman 2006). Habitat loss can occur in the absence of habitat fragmentation however most often these two separate processes occur simultaneously (Fahrig 1997). As habitat loss and habitat fragmentation increase, species become increasingly restricted to remnant habitat patches. These remnant habitat patches may be too small to allow species to meet their minimum habitat requirements (Simberloff and Cox 1987; Fahrig 2003). In addition to the patch size effects (a metric of both habitat loss and fragmentation), increasing patch isolation (the increasing disconnectedness of remnant habitat patches) is often interpreted as a metric for habitat fragmentation; however increasing patch isolation can also be viewed as less overall habitat in the landscape surrounding that patch (Fahrig 2003). Thus habitat fragmentation *per se* is much more difficult to conceptualize and quantify than habitat loss.

Strictly speaking, habitat loss is the reduction in overall habitat (i.e., a change in landscape composition) whereas habitat fragmentation is a change in the overall configuration of the landscape (Fahrig 2003). A landscape is a mosaic of habitat patches and matrix (or areas of non-habitat) (Wiens 1989). Landscape composition refers to the presence and amount of each habitat type within the landscape whereas landscape configuration refers to the spatial arrangement of habitat patches within the landscape (McGarigal et al. 2005). Therefore in theory it is possible to have only habitat fragmentation (the breaking apart of habitat into separate pieces or patches) occurring with no net loss of habitat. However in reality, these two processes are often linked (McGarigal et al. 2005). The simultaneous co-occurrence of habitat loss and fragmentation make it difficult to determine which process has a greater effect on biodiversity loss (Fahrig, 1997). That said, habitat fragmentation may cause a greater loss of biodiversity than pure habitat loss via increasing patch isolation and edge effects (Parker and Mac Nally 2002). Increasing patch isolation leads to increasingly isolated populations, which restricts gene flow and limits re-colonization if local patch extinction takes place (Lande 1988). Edge effects as defined by Leopold (1933 as cited in Ries et al. 2004) are the abiotic and biotic differences that exist within a transition zone (edge) between two adjacent communities (Murica 1995; Forman and Alexander 1998; Forman et al. 2003). Edge effects degrade the habitat patch causing the actual amount of original habitat to be smaller than the habitat patch itself (Parker and Mac Nally 2002); therefore species sensitive to edge effects will become restricted to the interior of habitat patches (Murica 1995).

My thesis focuses on the ecological impact of ATV trails. I apply recreation ecology knowledge of on-trail impacts and add off-trail impacts through the use of the concept of edge effects from road ecology. Examining ecological impacts directly on and off trail allows for the better spatial partitioning of these impacts.

1.2 Overview of Ecological Impacts of Linear Features: Road Ecology

A major contributor to habitat fragmentation is the construction of linear elements. These linear elements can range from roads, railroads, rights-of-way for highways, utility corridors, underpasses and tunnels (Simberloff 1992). Due to the ever increasing amount of such linear features, a whole sub-discipline of ecology dubbed “road ecology” has arisen and is dedicated to the study of the ecological effects of such linear elements (Forman and Alexander 1998; Forman et al. 2003). Since the early twentieth century it has been known that the creation of linear elements (particularly roads) causes edge effects (Clements 1907; Leopold 1933 as cited in Ries et al. 2004). However the way in which edge effects have been viewed has changed dramatically over the course of the twentieth century. From the early part of the twentieth century to mid-century, edge creation was encouraged because it was believed to be beneficial to wildlife, particularly game species (Forman et al. 2003; Ries et al. 2004; Harper et al. 2005). For some animals (i.e., ungulates), edges can act as corridors for animal movement (i.e., low cost travel through open strips rather than dense forest) and provide additional food resources (i.e., increased grazing opportunities) not found in the interior (Wallmo et al. 1972; Collins et al. 1978; Beier and Noss 1998; Haddad et al. 2003; Forman et al. 2003). By the 1970s the detrimental effects of edges began to be recognized. Avian breeding success was decreased through parasitism and predation near forest edges (Gates and Gysel 1978). Caribou (*Rangifer tardandus*) have been found to be at an increased risk of predation near linear elements which are utilized by wolves (*Canis lupus*) (James and Stuart-Smith 2000; Whittingston et al. 2011). Additionally, although edge creation (through such activities as linear element construction) results in very little actual habitat loss, the resulting habitat fragmentation can allow the spread of detrimental factors such as fire, pests/pathogens and invasive species (Rosenberg et al. 1997; Forman et al. 2003; Hansen and Clevenger 2005).

While the actual surface area of the road only accounts for a small proportion of the total landscape, the road-effect zone extends the ecological impact of roads into the surrounding landscape. The road-effect zone is the area in which ecological impacts of roads extend and are highly detectable (Forman and Alexander 1998). Road-effect zones can extend ten to hundreds of metres from the road edge (Forman and Alexander 1998). This wider spatial influence of roads and other linear elements reduces patch size and core interior giving roads and other linear elements a disproportionate impact on the landscape.

In general, open habitats are more susceptible to invasion by non-native plant species than forested habitats particularly since many exotic invasive species are adapted to high light conditions (Pardenes and Jones 2000). Hansen and Clevenger (2005) found railway and highway depth of edge to be greater in grasslands than in forest ecosystems in western Canada. In grassland sites non-native species were detected up to 150 m from the edge compared to only 10 m from the edge in forested sites (Hansen and Clevenger 2005). In oak (*Querus petraea* Liebel.) stands in France the main road effects (i.e., the presence of non-forest species and absence of sensitive species such as bryophytes) only extended 5 m from the road edge (Avon et al. 2010). In contrast Dubé and colleagues (2011) found colonization by non-native species further from the edge of power line rights-of-way in mostly wooded fens compared to bog sites in southern Quebec, Canada. Dubé and colleagues (2011) noted that barriers to bog invasion in a temperate zone (i.e., nutrient poor and water-logged soil) may not exist in boreal areas where xeric and mesic sites are more similar.

Road effects remain under quantified at the community, ecosystem and landscape levels (van de Ree et al. 2011). In order to gain a better understanding of higher order road effects whole communities and functional guilds need to be examined (Rotholz and Mandelik 2013). My study takes a plant community level approach to the impact of ATV trails in a boreal ecosystem. By taking a whole community level approach the broader trail impacts may be better quantified which can inform higher level policy and management decisions.

1.3 Recreation Ecology

Recreation ecology, like road ecology, has seen the bulk of its development as a sub-discipline of ecology (both conceptually and in practice) in recent decades. It began with observations and early experimental studies in the first half of the twentieth century, but the bulk of rigorous research was conducted in the 1970s and management applications only began to be implemented in the 1980s (Cole et al. 1987; Liddle 1997; Leung and Marion 2000). Recreation ecology arose out of the need to manage the negative impacts of visitors and users (i.e., hikers, campers, livestock users and Off-Road Vehicle [ORV] users) of wilderness areas and Protected Areas (PAs) (Cole et al. 1987; Liddle 1997; Leung and Marion 2000). As a sub-discipline, recreation ecology is highly management oriented. Recreation ecologists seek ways to minimize negative impacts of recreational users on wilderness resources so that these resources may be protected while meeting the requirements of recreational users (Cole et al. 1987; Liddle 1997; Leung and Marion 2000).

Wall and Wright (1977) divide recreational impacts into four categories: soils, vegetation, wildlife, and aquatic environments (as cited in Liddle 1997). Of these, soils and vegetation are most salient to this thesis, and will be discussed in more detail in sections 1.4 and 1.5. Impacts on wildlife include: road avoidance behavior (barriers to movement), direct mortality (road kill), noise effects, and increased human access (increased hunting pressure and introduction of feral and non-native species) (Buckley 2004). The literature on recreation impacts on wildlife and aquatic environments has been reviewed by Liddle (1997), Warnken and Byrnes (2004) and Mosisch and Arthington (2004). Neither of these latter categories of impacts is germane to this thesis and thus is not discussed further.

1.4 Recreation Impacts on Soil

Recreational trails can alter the hydrology and the geomorphology including soils of an ecosystem (Hawkins and Weintraub 2011; Arp and Simmons 2012). Soil degradation is classified into three categories: 1) physical degradation (i.e. soil compaction and erosion), 2) chemical degradation (i.e., changes in nutrient levels and soil pH) and 3) biological degradation (i.e., loss of soil biodiversity and disruption of nutrient cycling) (Snakin et al. 1996; Lal et al. 1997). All three types of degradation have been documented for recreational trails (Webb et al. 1978; Adams et al. 1982; Lei 2004; Hawkins and Weintraub 2011; Arp and Simmons 2012). These processes do not occur in isolation from one another but rather are a series of positive feedbacks that lead to further soil degradation (Lal et al. 1997; Crisfield et al. 2012). This study focuses on the physical aspects of soil degradation which are the drivers for chemical and biological degradation in the recreation impact context (Snakin et al. 1996; Harden 2001). Simply put, compaction is the re-arrangement of soil particles and the shrinking of pore spaces between soil particles (Webb 1983; Liddle 1997; Forman et al. 2003). It is the formation of a dense subsurface soil layer (Nortjé et al 2012). Virtually all forms of recreation (hiking, horse-back riding, biking, snowmobiling and All-Terrain Vehicle [ATV] riding) cause soil compaction (Liddle 1997). The soil type, soil moisture (i.e., water-logged, well-drained), topography and weight of the compacting force (i.e., hiker, horse or vehicle) greatly affect the amount of compaction (Radforth 1972; Nagy and Scotter 1974 (as cited in Yorks et al. 1997); Liddle 1975a; Weaver and Dale 1978; Adams et al. 1982). Generally fine-grained soils such as loam or clay are more susceptible to compaction than coarser grained soils (Liddle 1997). Compacted coarse-grained soil can drain much faster than compacted fine-grain soil (Liddle 1997). This is important since higher moisture level has been shown to increase the amount of compaction and erosion (Radforth 1972; Burton 1974 (as cited in Liddle 1975b); Jones, 1978 (as cited in Yorks et al. 1997); Lagocki 1978 (as cited in Liddle 1997).

Soil erosion is chiefly caused by wind or water but may be exacerbated by direct wear (on vegetation) through rutting by an object such as vehicular tires (Radforth 1972; Dale and Weaver 1974; Liddle 1997; Buckley 2004). Erosion shows similar trends to compaction in that soil type, soil moisture, topography and object weight directly affect the amount of erosion (Radforth 1972; Dale and Weaver 1974; Liddle 1975b; Weaver and Dale 1978; Liddle 1997; Buckley 2004). Soils high in silt content are more susceptible to eroding whereas clay soils are less susceptible (Ritcher and Negendank 1977 as cited in Liddle 1997). Steepness of slope, uphill or downhill, and the weight of the recreation user (i.e., a vehicle) have been found to increase overall levels of soil erosion (Radforth 1972; Liddle 1973 as cited in Liddle 1975b; Weaver and Dale 1978; Leung and Marion 1996). Erosion certainly is amongst the most visually striking impacts of recreation. Trail rutting, muddiness and proliferation (including braiding) are all highly visible impacts of ORV recreation (Cole et al. 1987; Leung and Marion 2000; Pickering and Hill 2007; Arp and Simmons 2012). Moreover trails tend to widen when ground is wet (Bayfield 1973) in part from users avoiding overly wet/water-saturated areas (Pickering et al. 2010).

Compaction and erosion differ in impact detectability in relation to the frequency of recreational use. For example, erosion effects are more pronounced at low levels of use (of ORVs) and compaction effects are more pronounced at high levels of use (Buckley 2004). Both soil erosion and trampling effects on vegetation follow the curvilinear use-impact relationship (Liddle 1997; Cole 2004; Quinn and Chernoff 2010; Figure 1.1). Several studies have found intensity of use to be a poor predictor of soil loss on trails (Dale and Weaver 1974; Cole 1992; Olive and Marion 2009). That is to say, at higher use intensities, an impact threshold is reached and further soil erosion is minimal, i.e., the soil has been worn down to the bedrock.

Soil compaction and erosion due to recreational use have been studied in a number of different habitats including deserts/arid regions (Iverson et al. 1981; Adams et al. 1982;

Wilshire 1983; Cole 1986; Lei 2004; Goossens and Buck 2009; Nortjé et al. 2012), coastal sand dunes and coastal environments (Liddle and Greig-Smith 1975a; Keddy and Wisheu 1989; Anders and Leatherman 1987; Davenport and Davenport 2006), alpine/tundra (Willard and Marr 1970, 1971; Radforth 1972; Greller et al. 1974; Willard et al. 2007; Törn et al. 2009; Arp and Simmons 2012) and coniferous and deciduous forests (Dale and Weaver 1974; Weaver and Dale 1978; Wilson and Seney 1994; Cole and Spildie 1998; Thurston and Reader 2001; Sack and da Luz 2003; Hawkins and Weintraub 2011). Of these habitats, deserts, coastal sand dunes, alpine and tundra have been the most extensively studied (Liddle 1997). Studies suggest that tundra is most sensitive to soil erosion due to disturbance of the permafrost, followed by alpine due to high amounts of visitor traffic, followed by coastal dunes due to the removal of protective vegetation and finally deserts where tracks may be visible for centuries but overall amounts of erosion are lessened due to low amounts of rainfall (Dregne 1983; Liddle 1997; Buckley 2004).

Trail erosion, including ORV trails, occurs with net deposition of sediment adjacent to the trail and with an increase of trail surface run-off due to compaction resulting in the fluvial transportation of sediment away from the trail (Harden 2001; Sack and da Luz 2003). In my study the amount of soil compaction, soil flux, soil deposition, bulk density and soil nutrient levels are not the primary focus as this has been documented in manipulative and observational ORV trail studies (Webb et al. 1978; Iverson et al. 1981; Adams et al. 1982; Sack and da Luz; 2003; Lei 2004; Hawkins and Weintraub 2011; Arp and Simmons 2012; Nortjé et al. 2012). The intent was to obtain a simplified measure of the vulnerability of the substrate to erosive forces and compare this to off-trail impacts.. One simple measure of the physical degradation of soil on trails is displacement (i.e., gully/rut depth) (Snakin et al. 1996; Meyer 2002). Gully or rut formation is a common indicator of soil loss on ORV trails (Meyer 2002). Rut depth was used as a proxy for erosion in this study.

1.5 Recreation Impacts on Vegetation

In the assessment of recreation impacts, vegetation trampling studies are a widely used tool. Trampling studies have been conducted in a number of different habitats and the effects of different types of recreational users have been compared. As in the soil erosion studies certain habitats have been the focus of research; the most common have been arid environments (Iverson et al. 1981; Wilshire 1983; Cole 1986) and coastal dunes (Liddle and Greig-Smith 1975b; Rickard et al. 1994; Anders and Leatherman 1987). The alpine and tundra ecosystems have also been well studied, however it is noteworthy to mention that the tundra has largely been studied in the context of impacts of seismic vehicles for oil and gas exploration (Willard and Marr 1970, 1971; Radforth 1972; Bliss and Wein 1972; Greller et al. 1974; Racine and Johnson 1988; Emers et al. 1995; Whinam and Chilott, 1999; Willard et al. 2007; Törn et al. 2009 [sub-alpine was compared to boreal forest]; Jorgenson et al. 2010). The findings of such studies are still applicable to impacts caused by recreational ORV use. Comparatively less attention has been given to wetland areas (i.e., bogs, marsh, floodplains and riparian zones) but see Keddy et al. (1979), Ross (1991), Charman and Pollard (1993), Cole and Marion (1998), and Hunkapiller et al. (2009). Other habitats which have had limited study are forested habitats (but see Thurston and Reader 2001; Turton 2005; Törn et al. 2009) and grasslands and heaths (but see Bayfield 1979; Charman and Pollard 1993; Arnesen 1999, Roovers et al. 2004; Meadows et al. 2008). A major exception is the montane forests of the south-western United States which have been extensively studied (Dale and Weaver 1974; Cole 1978; Weaver and Dale 1978; Cole 1985; Cole 1987; Wilson and Seney 1994; Cole and Spildie 1998). Furthermore, across different habitats, recreational trampling of rare plant species has received research attention (Maschinski et al. 1997; Kelly et al. 2003; Kerbiriou et al. 2008).

Recreation impacts on vegetation include crushing, abrasion, introduction of exotic/invasive species, overall reduction of biomass- particularly of sensitive species and shifts in species composition (Liddle 1997; Cole 2004; Rooney 2005; Pickering and Hill 2007). Shifts in species composition include a shift to more non-native species, and/or to a community dominated by those species that can withstand trampling and physical disturbance (resistance or resilience) (Liddle 1997; Cole 2004; Rooney 2005; Pickering and Hill 2007).

The concepts of resistance and resilience are key elements to understanding the impacts of recreation on vegetation communities. In my study I adopted this community level view of resistance and resilience since I am interested in comparing different communities' responses to recreational impacts. I used two metrics for assessing resistance and resilience of boreal communities to ORV trails; 1) the vulnerability of the substrate to erosive forces (defined in section 1.4) and 2) changes in plant life form composition from trail edge to interior. I defined plant resistance to recreational impacts as the ability to withstand being "injured or impaired," and plant resilience to recreational impacts as the ability to "survive or regenerate" (Kuss and Hall 1991; Yorks et al. 1997). I defined tolerance as the ability of a plant to be highly resistant or resilient or the use of a combination of both strategies (Liddle 1997; Monz 2002). In my study resilience was not directly measured but was inferred by the amount of edge effect impact. Vegetation communities differ greatly in their resistance and resilience to recreation impacts, particularly trampling (Cole 1987, Liddle 1997; Yorks et al. 1997).

Open habitats such as tundra, heaths and bogs have low resilience and long recovery times following recreational (particularly vehicular) disturbance (Willard and Marr 1970; Greller et al. 1974; Bayfield 1979; Charman and Pollar 1993). Such habitats are ecologically sensitive and have limited capacity to recover from recreation disturbance due to their low productivity (Willard and Marr 1970, 1971; Greller et al. 1974; Liddle 1997). In such low productivity environments it has been estimated to take centuries for vegetation communities to recover from recreation impacts (Willard and Marr 1971; Webb 1983). Some systems may never fully recover to pre-disturbance conditions. Charman and Pollard (1993) investigated the recovery of vegetation following trampling by military vehicles 20 years previous in England. Bog sites showed very little recovery and one site was in succession towards a grassland community (Charman and Pollard 1993). This contrasted with grassland sites which showed little difference from un-trampled controls (Charman and Pollard 1993).

Wetter areas within a given habitat type are prone to more erosion and likely to be denuded of vegetation more quickly than drier areas, given similar types and amounts of use (Willard and Marr 1970; Radforth 1972; Liddle 1997; Törn, et al. 2009; Jorgenson et al. 2010). This means that within a given habitat, wetter areas are often more adversely affected than drier areas. Trampling has been shown to increase soil erosion and reduce vegetation cover disproportionately in wetter areas in a number of habitats (Willard and Marr 1970; Monz et al. 1996; Törn, et al. 2009). In general the relationship between trampling intensity and vegetation follows a curvilinear pattern (Liddle 1975a; Cole et al. 1987; Cole 1995b; Liddle 1997; Leung and Marion 2000), whereby at extremely high intensities of use a threshold has been reached and no further response is detectable (Figure 1.1). At this stage soil horizons have been completely eroded and only bedrock remains and vegetation cover has been reduced to zero. At very low intensities in tolerant plant communities (i.e., grass dominated) growth may be stimulated and overall biomass increased (Bayfield 1971; Kellomäki 1973 as cited in Liddle 1975b). However grass species biomass is reduced at higher intensities of use (Burden and Randerson 1972 as cited in Liddle 1975a). In more diverse communities, changes in overall species composition may occur. There is a shift from less tolerant dicotyledonous species to more tolerant monocotyledonous species (Liddle 1975a; Yorks et al. 1997). This shift is more pronounced in ecologically sensitive environments (Liddle and Thyer 1986 as cited in Yorks et al. 1997). For example in low productivity wetlands such as bogs, the amount of trampling by ATVs that reduced overall vegetation cover was extremely minimal (Ross, 1991). Ross (1991) found that only 20 ATV passes were required to reduce vegetation cover by 50% in bogs in Nova Scotia, Canada (Ross 1991). At 40 passes vegetation cover had been reduced to zero (Ross 1991). In open areas such as heaths, meadows and alpine, relative moisture is a key determinant of plant community tolerance to recreation impacts along with plant growth form (Kuss 1986). Low productivity environments, particularly those areas with higher moisture soil content are highly vulnerable to adverse recreation impacts.

1.6 Resistance and Resilience of Plant Communities to Recreation

There are numerous factors which influence the capacity for resistance and resilience to recreational trampling in plant communities. This study considers three influences: habitat type (environmental productivity), moisture regime and plant life form. Low productivity environments are believed to have lower resilience and resistance to recreational impacts than higher productivity environments (Liddle 1975b). Soil moisture compounds these impacts; generally the greater the soil moisture the greater the impact. More productive environments may or may not have higher levels of resistance and resilience compared to less productive environments. For example highly productive broad-leaved forest understory plants have been shown to have low resistance (Cole 1987; Yorks et al. 1997; Thurston and Reader 2001) but broad-leaved forest understory plants can have high resilience if recreational disturbance is intermittent (Cole 1995b). Adaptations of shade-tolerant understory plants (i.e., greater leaf area, thinner cuticles) make them less resistant to recreational trampling (Cole 1978; Cole 1995a). Interestingly when canopy densities are similar, deciduous forests may be more vulnerable to impacts than coniferous forests due to the relatively large unincorporated organic litter layer present in coniferous forests (Legg 1973 as cited in Kuss 1986). Thus, generally speaking, in coniferous or boreal forests there are fewer low growing understory plants and the soil is buffered from trampling by a thick organic layer. Cole (1995a; 1995b) found that vegetation community tolerance to recreational trampling was determined by the level of resilience rather than resistance. In general though, more productive environments have higher resilience than less productive environments and therefore higher tolerance.

Resistance and resilience to recreational impacts may be examined at the level of the individual, species, community or even ecosystem level. A common approach to examine plant resistance and resilience to recreational impacts has been to use plant life forms as indicators of community level response (Hall and Kuss 1989; Kuss and Hall 1991; Cole 1995a; Whinam and Chilcott 1999; Törn et al. 2006; Hill and Pickering 2009; Jorgenson et al.2010). The plant life form level can be used for monitoring the effects of disturbance because plant life form correlates with physiological and morphological traits that may be used to predict overall community response (McIntrye et al. 1995). Plant life form categories in my study were broader than those outlined in Raunkier's (1934) classification system (Table 1.1). I did this to make the results more comparable to general recreational studies of vegetation. Plant life form categories used in my study are outlined in Table 1.2. In terms of plant life form the most resistant and most resilient and therefore the most tolerant are consistently the graminoids (Cole 1995a; Yorks et al. 1997). Further generalization of sensitivity across plant life forms (least resistant or resilient) is more difficult to summarize since different researchers have different rankings, but generally chameaphytes and thallophytes (plants bearing a thallus- i.e., mosses and lichesns) are highly sensitive to recreation impacts (Cole 1995a; Yorks et al. 1997).

At the community level the most resistant communities are generally characterized by dry well-drained soil that resists compaction (i.e., gravel), are open (sun exposed), and graminoid dominated (Kuss 1986; Liddle 1997). In contrast the more sensitive communities are generally characterized by poorly drained soil or constantly wet and highly compactable soil (i.e., clay), a dense canopy or open low productivity habitat in which chamaephytes and/or thallopytes are dominant (Kuss 1986; Liddle 1997).

1.7 Consideration of Spatial Scale within Recreation Ecology

Within the field of recreation ecology it has been recognized that spatial scale aspects of human recreational impacts (i.e., All-Terrain Vehicle [ATV] trails) have been largely understudied (Cole 2004; Brooks and Lair 2005; Ouren et al. 2007). Brooks and Lair (2005) defined three categories of vehicular impacts with distinct spatial scales 1) direct effects, 2) indirect effects and 3) landscape effects. Direct effects occur within the confines of the trail itself (Brooks and Lair, 2005) through the loss of vegetation cover or erosion associated with rutting. The majority of research on ORV trails have been direct effects based (Liddle, 1997; Leung and Marion 2000; Buckley 2004; Ouren et al. 2007). Indirect effects occur in areas adjacent to trails (Brooks and Lair 2005) through increased sediment or nutrient loading of surrounding vegetation. Direct and indirect effects are conceptually analogous to the road ecology concepts of “road corridor” and “road-effect zone” outlined by Forman and Alexander (1998). As the name implies landscape effects are dispersed throughout the surrounding landscape (Brooks and Lair, 2005) i.e., habitat fragmentation, spread of invasive species. While all effects (direct, indirect and landscape) are influenced by specific environmental and ecological gradients and different land use regimes (Brooks and Lair, 2005), indirect and landscape effects can be more difficult to quantify. Nevertheless elucidating ORV impacts at the appropriate spatial scale of ecosystem response is crucial for efficient management of these impacts (Brooks and Lair 2005); particularly when many management decisions are at the landscape level. An understanding of ecological ORV impacts at multiple spatial scales allows for a clearer comprehension of the system as a whole. This holistic perspective allows for more effective adaptive management since management decisions can be tailored to a particular spatial scale where they are most easily implemented and most likely to be effective.

There may be different ecological impacts operating at discrete spatial scales; however there seems to be a disjunction of these different spatial processes from each other. Multiple ecological impacts from recreational trails occur at multiple spatial scales and rather than being mutually exclusive the processes may be cumulative with many smaller scale impacts translating into medium or large scale impacts (Brooks and Lair 2005).

Previous trampling and trail studies ranged from descriptive studies to before-after field and stimulated experiments (reviewed by Leung and Marion 2000; Cole 2004; Hill and Pickering 2009) and a standardized protocol has been put forth for manipulative direct (on-trail) trampling studies (Cole and Bayfield, 1993). These studies and protocol focused on the localized scale of the trail or area immediately adjacent to it. A notable exception is the methodology employed by Hall and Kuss (1989) where sample quadrats were placed at three discrete distances (1 m, 2 m, 10 m) from the trail; however 10 m quadrats were considered “un-impacted” controls (also see Naito 1969). Within recreation ecology there have been calls for more of this type of gradient study (Brooks and Lair 2005; Ouren et al. 2007). Gradient studies have several appealing aspects. Such studies avoid the problem of “untrue” controls, which is placement of control sites in areas impacted by indirect effects (Brooks and Lair 2005). Also gradient designs identify thresholds of ecological response which can later be incorporated into comparative (control vs. disturbed) study designs (Brooks and Lair 2005). Moreover data from these studies can inform modelling of landscape effects (Brooks and Lair 2005).

1.8 Regulatory Framework Governing All-Terrain Vehicles Use on the Island of Newfoundland

The vulnerability of ecologically sensitive environments to recreational impacts of off-road vehicles (ORVs) such as all-terrain vehicles (ATVs) has been recognized by the province of Newfoundland and Labrador policy makers. ORVs are regulated by different pieces of legislation within and outside of protected areas (i.e., parks and reserves). Provincial legislation governs the use of ORVs through the *Motorized Snow Vehicles and All-Terrain Vehicles Act* (Table A1). Outside of protected areas the *Motorized Snow Vehicles and All-Terrain Vehicles Regulations under the Motorized Snow Vehicles and All-Terrain Vehicles Act* (O.C. 96-240) in conjunction with the *Lands Act* lay out “approved areas” where ATV use is permitted. Under these regulations ecologically sensitive areas such as wetlands, bogs and barrens are not approved areas for ATV use (Table A1). The regulation does make allowances for hunters to use ATVs in unapproved areas; it allows hunters to cross unapproved areas up to five times to retrieve a hunt kill (Table A1). The use of ATVs within protected areas is regulated by different provincial legislation than described above (refer to Table A1 for a description of regulations governing ATV use in provincial parks).

In the province of Newfoundland and Labrador all human activities that take place within wilderness and ecological reserves are regulated under the *Wilderness and Ecological Reserves (WER) Act*. The discussion here will be limited to wilderness reserves since my study area contains a wilderness reserve. The legal functions under the WER Act are outlined in Table A1. The first legal function of a wilderness reserve is to provide the public with opportunities for outdoor recreation (Table A1). However, these recreational activities are intended to be relatively low impact such as hiking and canoeing (see Table A1 for a more complete description of permitted recreational activities). The number of people entering a reserve is regulated since all persons wishing to enter, for example, the Avalon Wilderness Reserve (study area) must first obtain an entry permit (*Wilderness Reserve Regulations* Section 4, 1997). Restrictions on human activities are designed to minimize the impacts on the environment. For example, campers are prohibited from erecting a tent (*Wilderness Reserve Regulations* Section 5.1, 1997). Instead the use of a pick-up truck camper located within 20 metres of the centre of any roadway within the reserve is permitted with the proper permit *(Wilderness Reserve Regulations* Section 5.2, 1997). Outboard motors up to 6 horsepower may be used on designated lakes (*Wilderness Reserve Regulations*, Section 16.4, 1997). Within the Avalon Wilderness Reserve the use of off-road vehicles such as all-terrain vehicles and snowmobiles is prohibited (Sections 7.1(i)(j), 16.1, 1997). Hunting of moose and small game in the adjacent non-reserve area is permitted under license (Department of

Environmental and Conservation 2013). As of 2004 caribou management area 65, the Avalon Peninsula, in which Avalon Wilderness Reserve is located, has been closed to caribou hunters (Department of Environment and Conservation 2013). For examples of other prohibited activities refer to Table A1.

The legal framework surrounding ORVs in the province of Newfoundland and Labrador recognizes the adverse environmental effects more intense forms of recreational activities have, particularly on sensitive environments. Both in protected and non-protected areas people's desire to use natural resources for recreational purposes and environmental protection are taken into account.

1.9 Recreational Trail and All-Terrain Vehicle Sales Statistics

In 2010 the first known study on Canada's national recreational trail system was commissioned by the National Trails Coalition (NTC). The NTC estimates there are over 278,000 kilometers of managed trails nationally (Norman 2010). Managed trails were defined as recreational trails managed or operated by a government agency or other incorporated or non-profit trail organization (Norman 2010). The managed trail system includes motorized and non-motorized trails. Motorized trails were defined as snowmobile, ATV or off-road motorcycle (ORM) trails (Norman 2010). Non-motorized trails were defined as walking/hiking, cycling, mountain biking, cross-country skiing and equestrian trails (Norman 2010). The amount of motorized trails is double the amount of non-motorized trails; specifically motorized trails comprise 66.4% of total kilometers of managed trails throughout Canada compared to 33.6% of non-motorized trails (Norman 2010). It is important to note that the statistics presented here only represent managed trails and do not take into account un-managed trails- i.e., trails constructed by private individuals and/or illegal trails. The actual number of trails in more rural provinces such as Newfoundland and Labrador are underreported. Thus the number and length of trails within the province and the entire country is likely much greater than the official statistics report.

As of 2010 the province of Newfoundland and Labrador had a total number of 7,440 km of managed trails (Norman 2010). This managed trail system includes 4,600 km of motorized single use trails (trails intended for only one type of recreational use), 1,602 km of non-motorized single use trails, 1,086 km of shared use motorized trails (trails used for more than one type of recreational use; for example ATVing in the summer and snowmobiling in the winter) and 152 km of shared use non-motorized trails (Norman 2010).

The trend of motorized trails outnumbering non-motorized trails is mirrored across Canada. Ninety-five percent of all recreational trails are rural (located away from a major population centre) and the majority of these are motorized (Norman 2010). Conversely the majority of urban trails are non-motorized trails (Norman 2010). When compared nationally, the province of Newfoundland and Labrador has a higher percentage of both single and shared ATV/ORM trails (Table 1.3). In Newfoundland and Labrador ATV/ORM trails make up over a quarter of all managed trails within the province, which is higher than the national average (Table 1.3).

The popularity of motorized recreation in Newfoundland and Labrador is reflected in retail sales. Data compiled from the Motorcycle & Moped Industry Council (MMIC) and the Canadian Off Highway Vehicle Distributors Council (COHV) (Scooter & All-Terrain Vehicle Annual Industry Statistics Report 2011) indicate the popularity of ATVs among residents of Newfoundland and Labrador. ATV sales have been on the rise over several years (Table 1.4). The population of Newfoundland and Labrador accounts for only 1.5% of the total Canadian population however in 2010 4.9% of all new ATVs sold in Canada were in Newfoundland and Labrador (Table 1.4). This number rose to 5.49% in 2011 which is quite high compared to more populous provinces in Atlantic Canada (Table 1.4). If these increasing trends of ATV sales continue, ATV use within the province can be expected to increase. Findings of this study could help to inform future ATV management discussions.

1.10 Objectives of Thesis

The broad objective of this study was to determine the level of ecological impacts that all-terrain vehicle (ATV) trails have in areas under different forms of legal protection. I focused on two different areas, a protected reserve (R) (the Avalon Wilderness Reserve [AWR]) and the adjacent surrounding non-protected, non-reserve (NR) area. I also sought to determine impact levels in high and low productivity habitat types and among habitats under different moisture regimes. The different habitats were categorized as forest (high productivity) or open (low productivity). The moisture regime within a habitat was categorized as dry or wet. The final objective of the study was to make recommendations to managers which may contribute to the reduction of ATV trail impacts within the Maritime Barrens Ecoregion (MBE). The MBE is an extensive ecoregion, covering over 20% of the island of Newfoundland and extends from the west coast through the central portion to the east coast of the island (Protected Areas Association of Newfoundland and Labrador 2008; Figure 1.2). Within these broader objectives, specific study objectives included: 1) to document the extent and location of ATV trails within AWR and adjacent lands (where logistically possible), 2) to document on-trail (direct) effects of wheel ruts on soil and bog substrate, 3) to document the level of ATV traffic on a subset of trails representative of the different areas of legal protection, habitat type and moisture level and 4) to investigate off-trail (indirect) effects- fine scale impacts on vegetation.

1.10.1 Study Rationale and Hypotheses

The study design incorporates aspects from landscape ecology, road ecology and recreation ecology. The hierarchical design is conceptually drawn from landscape ecology and the gradient design from road ecology. This study examines both direct effects (on-trail impact) measured by soil erosion and indirect effects (off-trail impact) measured by changes in the vegetation community away from trails. The intensity of trail use (amount of ORV/ATV traffic) is also considered. This allows for the better understanding of spatial aspects of ORV/ATV trails ecological impacts since both direct and indirect effects are considered.

I assumed a difference in ATV traffic volume on trails between the Avalon Wilderness Reserve (AWR) and the adjacent non-reserve (NR) area; as a protected area where ATV use is prohibited, AWR was expected to have lower ATV traffic volume on existing trails than the adjacent non-reserve. I intended that the AWR would act as a contrast to the higher ORV/ATV traffic NR but also that habitat types were similar. The study area (located within the MBE) was intended to be a proxy for the less accessible South Coast of Newfoundland also located within the MBE (Fig. 1.2), but where use of ATVs, particularly for hunting, is high.

I hypothesized that ecological impacts of ATVs as measured by on-trail erosion would be influenced by cover type (habitat), moisture level (micro-habitat) and intensity of use. I assumed that use thresholds are below the inflection point on the curvilinear use-impact curve due to the relative remoteness of the study area. Thus ecological impacts will be highly detectable. Low productivity habitats are believed to be highly susceptible to, and slow to recover from, recreation disturbance (Liddle 1975b). Open, low productivity habitats such as deserts, tundra, and alpine have been shown to be highly susceptible to erosion via recreational trails (Liddle 1997; Buckley 2004). Previous studies demonstrated that hydric and mesic sites are more susceptible to erosion than more xeric sites (Radforth 1972; Burton 1974 as cited in Liddle 1975b; Jones 1978 as cited in Yorks et al. 1997; Lagocki 1978 as cited in Liddle 1997). Thus, I predicted that 1a) on-trail erosion would be greater in open habitats than in forested habitats; 1b) on-trail erosion would be greater in wet sites than dry sites; and 1c) on-trail erosion would be greater on high traffic trails than on low traffic trails.

Recreation impacts on vegetation both on- and off-trail include crushing, abrasion, introduction of exotic/invasive species, overall reduction of biomass (particularly of sensitive species) and shifts in species composition (Liddle 1997; Cole 2004; Rooney 2005; Pickering and Hill 2007). Shifts in species composition may include a shift to more non-native species, and/or to a community dominated by those species that can withstand trampling and physical disturbance (resistance or resilience) (Liddle 1997; Cole 2004; Rooney 2005; Pickering and Hill 2007).

My second hypothesis is that ecological impacts of ATVs as measured by off-trail impacts (indirect edge effects) on vegetation will be influenced by cover type (habitat), moisture level (micro-habitat) and intensity of use. Open habitats such as tundra, heath and bogs have low resilience and long recovery times following recreational (particularly vehicular) disturbance (Willard and Marr 1970; Greller et al. 1974; Bayfield 1979; Charman and Pollard 1993). Wetter areas within a given habitat type have shown more erosion and are likely to be denuded of vegetation more quickly than drier areas (Willard and Marr 1970; Radforth 1972; Liddle 1997; Törn et al. 2009; Jorgenson et al. 2010) given similar types and amounts of use. I predicted that 2a) off-trail vegetation impacts (changes in species composition) would be greater (appear further from the trail) in open habitats than in forested habitats; 2b) off-trail vegetation impacts (changes in species composition) would be greater (appear further from the trail) in wet sites than dry sites; and 2c) off-trail vegetation impacts (changes in species composition) would be greater (appear further from the trail) on high traffic trails than low traffic trails.

Table 1.1: Notes on Life Form Classification. Table lists primary classes only. Life form classifications and descriptions follow Clapham, Tutin and Moore (1987).

Life Form Class	Description
Phanerophytes	Woody with buds above 25 cm above the soil surface
Chamaephytes	Woody or herbaceous with buds below 25 cm but above the soil surface
Hemicryptophytes	Mostly herbaceous (rarely woody) with buds at the soil surface
Geophytes	Herbaceous with buds below the soil surface (i.e. plants with bulbs, corms or rhizomes)
Helophytes	Marsh plants
Hydrophytes	Water plants
Therophytes	Plants which pass the unfavorable season as seeds (i.e. annuals)

Table 1.2: Life form categories used in this study.

Life Form Category	Description
Trees	Woody plants >4.5 metres in heights at maturity, some species may attain tree size in other regions of North America but classified as shrubs using regional province of Newfoundland and Labrador classification
Shrubs	Woody plants <4.5 metres in height at maturity, often possessing multiple stems
Herbs	Non-woody vascular plants excluding ferns
Ferns	Ferns
Mosses	Mosses
Lichens	Lichens
Graminoids	Grasses and grass allies

Species were assigned life form categories based on field guides: Ryan (1978); Johnson and colleagues (1995); Farrar (1995) and Scott and Black (2008).

Table 1.3: National and Regional Managed All-Terrain Vehicle (ATV) and Off-Road Motorcycle (ORM) Trails by Percentage of Total Kilometres (km) as of 2010. Single use trails are trails designated for one type of recreational use only. Shared use trails are designated for multiple types of recreational use. Percentage of total km of motorized trails for Canada includes Newfoundland and Labrador trails.

	Percentage (%) of Single Use ATV Trails	Percentage (%) of Shared Use ATV/ORM Trails	Total
Newfoundland and Labrador	13.4	12.5	25.9
Canada	11.6	9.9	21.5

Table 1.4: Retail All-Terrain Vehicle (ATV) Sales by Province in Percent (%) of Units Sold Nationwide. Percentages are based on the Motorcycle & Moped Industry Council (MMIC) member companies comprise approximately 95% of new on-road motorcycles and scooters sold in Canada (Scooter & All-Terrain Vehicle Annual Industry Statistics Report 2011). Canadian Off Highway Vehicle Distributors Council (COHV) member companies comprise approximately 90% of new ATVs and off road motorcycles sold in Canada (Scooter & All-Terrain Vehicle Annual Industry Statistics Report 2011).

Province	Percent of population as of July 1st 2009	Retail ATV sales as a percent of total units sold nationwide		
		2009	2010	2011
Newfoundland and Labrador	1.5	4.42	4.9	5.49
New Brunswick	2.2	3.72	4.09	3.76
Nova Scotia	2.8	2.20	2.28	2.82

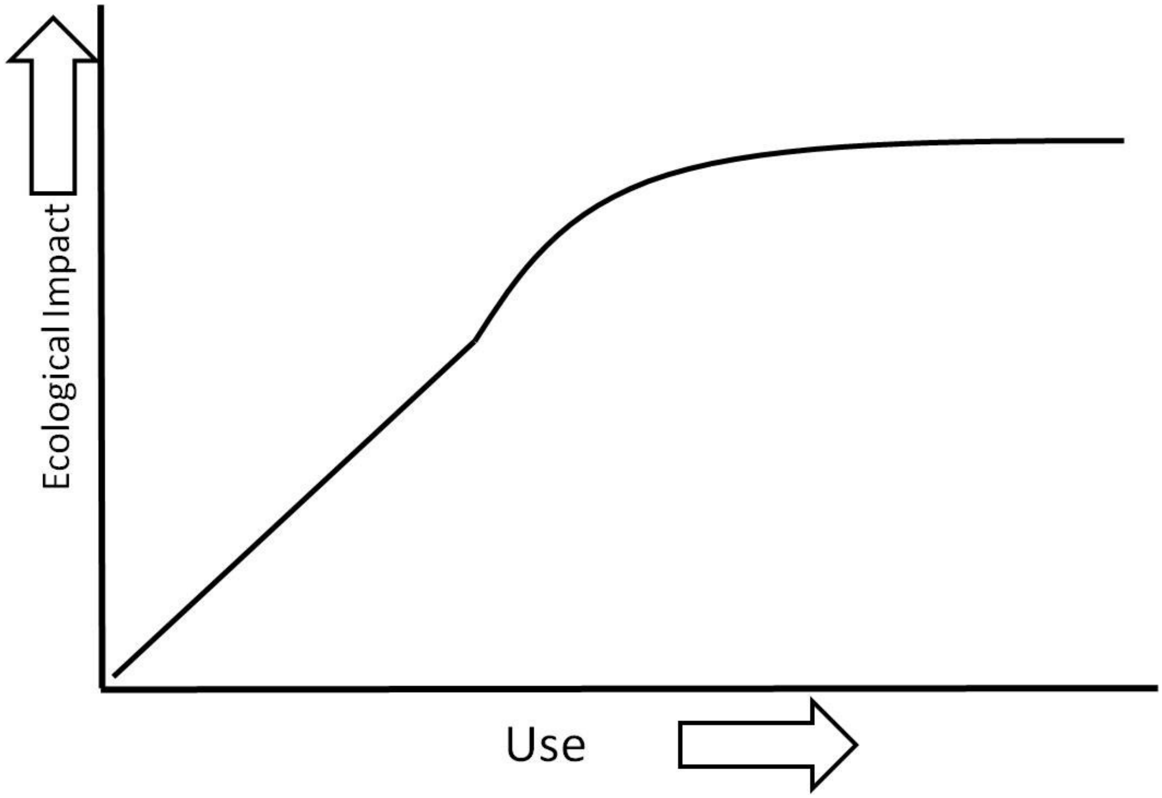

Figure 1.1: A generalized form of the curvilinear use-effect relationship, adapted from Quinn and Chernoff (2010). This figure describes the relationship between direct ecological impacts and intensity of recreational use. Direct ecological impact increase proportionally with the increase in recreational use until an inflection point where further increase in use does not result in further direct ecological impact response.

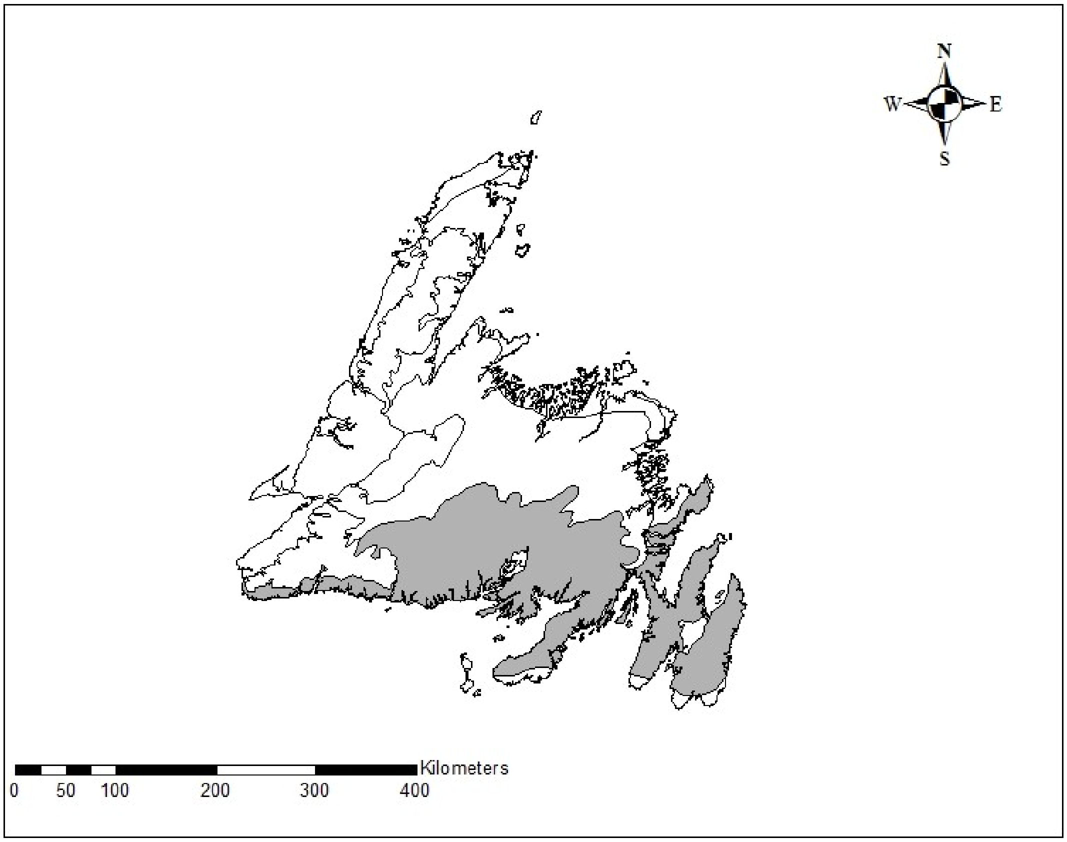

Figure 1.2: Map of the island of Newfoundland, Canada. Location of the Maritime Barrens Ecoregion is shaded grey.

2 Chapter 2: Methods

2.1 Study Area

The Maritime Barrens Ecoregion (MBE) is a specific ecological and phytogeographical division of the island of Newfoundland and is considered a part of the southern boreal zone (Damman 1983). These divisions or ecoregions were created to reflect regional differences in climate on the island (Damman 1983). The climate is marine with characteristic cool summers (mean temperature ranges from 13 to 16^{o}C) and mild winters (mean temperature ranges from -3 to -8^{o}C) (Damman 1983; Protected Areas Association of Newfoundland and Labrador 2008). Annual precipitation is over 1250 mm in most of the ecoregion and fog cover is frequent (Damman 1983). During winter, snow cover may be intermittent due to precipitation falling as rain or snow (Damman 1983). As its name suggests, the Maritime Barrens landscape is dominated by nutrient poor environments; heaths, bogs and fens (Damman 1983).

Forest stands occur in more sheltered areas such as valleys (Damman 1983) with deeper soil and better growing conditions. Prior to European settlement forests were widespread in this ecoregion, the current landscape structure was created by the deliberate setting of fires by Europeans to clear the land for agriculture (Protected Areas Association of Newfoundland and Labrador 2008). With the arrival of the railway in Newfoundland in the late 19th century fires became more widespread and numerous (Protected Areas Association of Newfoundland and Labrador, 2008). For example, in 1904 it was recorded that two million acres of forest burnt throughout the island (Protected Areas Association of Newfoundland and Labrador 2008). Forests are now restricted to remnant patches within this ecoregion due to competition with ericaceous dwarf shrubs, poor growing conditions and herbivory pressure from the hyperabundant non-native moose (*Alces Alces*) (Damman 1983; McLaren et al 2004; Gosse et al 2011).

Among the dwarf shrubs *Kalmia augustifolia* L. (sheep laurel) is the dominant heath species, however *Rhododenron canadense* (L.) Torr. (rhodora) and *Vaccinium augustifolium* Ait. (wild blueberry) are also abundant (Damman 1983). In more exposed areas *Empetrum nigrum* L. (black crowberry) and *Vaccinium vitis-idaea* L. (partridgeberry or lingonberry) are prominent (Damman 1983). Other dominant ground cover includes members of the lichen genus *Cladonia* spp., mosses *Pleurozium schreberi* (Brid.) Mitt. and *Sphagnum* spp. (in wetter areas) (Damman 1983; Meades 1983).

Interestingly this ecoregion has relatively few blanket bogs. Blanket bogs are limited to ridge tops due to high moisture surplus in the springtime (Damman 1983). Common bog types in the ecoregion include basin and slope bogs which are characterized by being relatively shallow (rarely exceeding 2 m) (Meades 1983). Basin bogs develop on flat terrain and are ombrotrophic (the main source of nutrients is from rainwater) (Meades 1983). Slope bogs are largely ombrotrophic, with slopes ranging from 5%-20% and are often located amongst forested areas (Meades 1983). In forest patches dominant trees are black spruce (*Picea mariana* Mill.) and balsam fir (*Abies balsamea* L.) (Protected Areas Association of Newfoundland and Labrador 2008). Forest floor mosses are dominated by *Dicranum* spp. and *Rhytidiadelphus* spp. (Damman 1983).

My study area was located in the MBE on the east central section of the Avalon Peninsula, 47^{o}6’ N, 53^{o}15’W (Damman 1983; Parks and Natural Areas Division 2006), Newfoundland. This included the Avalon Wilderness Reserve (AWR) and the adjacent area shown in Figure 2.1. Directly off Route 10 is Horse Chops Road, an unpaved road that penetrates deeply into the reserve (Figure 2.1). Horse Chops Road is privately owned historically by the Newfoundland Light and Power Company (currently Newfoundland Power Incorporated a Fortis company) and its construction predates the founding of the reserve (Avalon Wilderness Reserve Management Plan, 1986; Department of Environment and Conservation 2012). The road was constructed by the company to provide access to dams in the area (Avalon Wilderness Reserve Management Plan 1986). The road is suitable for 4-wheel drive vehicles and ATVs only; this makes it an ideal entry point for two reasons. First it allows relative ease of access for field work and second there is an increased probability of discovering illegal trails since a road easily usable by ATVs is already in place.

2.2 Study Design

My study design incorporated a hierarchical 3x2 factorial design. The three factors were: Legal Status, Habitat and Micro-Habitat. Each factor has two levels. The legal status levels are Non-Reserve (NR) and Reserve (R). The habitat levels are Forest (F) and Open (O). The micro-habitat levels are Dry (D) and Wet (W).

Aerial photographs were used to identify potential ATV trails. This was necessary since a comprehensive map of ATV trails across the island of Newfoundland does not exist (Personal Communication R. Noseworthy, Newfoundland T’Railway Council

2010) and trails within the AWR are illegal and thus not mapped. I assigned treatments using a random stratified sample created with a Geographic Information System (GIS). All geo-processing was done in ArcGIS version 9.3 (ERSI, 2008). Stratification was based on three data sources: 1) 2008 air photo imagery (55 cm ortho-photos obtained from the Department of Natural Resources [DNR]); 2) 2003, 25 m Earth Observation for Sustainable Development (EOSD) land cover data (obtained from Natural Resources Canada [NRCan] and based on Landsat Imagery) and 3) 75 m Topographic Relative Moisture Index (TRMI) created from a Digital Elevation Model of the Province (Skinner 2011). All treatment levels assigned from GIS random stratification were verified at the beginning of the field season (May 2010); where necessary new sampling locations were chosen or factors reassigned. For example a trail that had been classified as open/wet by the GIS that was in fact open/dry would be reassigned. Such discrepancies were due to the coarse spatial scale on the input data layers. The majority of trails discovered were not visible on the aerial photos. Thus preliminary site selection via GIS random stratification could not be entirely applied. I mapped all trails using a handheld Geographic Positioning System (GPS) GARMIN 76 and walked the entire length of each trail. There were a total of 28 trails, with lengths ranging from 29-1415 m (mean 459 m, SD±413.06). The total length of Horse Chops Road is 68 km (see Table 2.1 for further trail details).

2.3 Data Collection Field Methods

2.3.1 Erosion

To obtain a simplified measure of the vulnerability of the substrate to erosive forces and compare this to off-trail impacts I examined the physical degradation of soil on trails through displacement (i.e. gully depth) (Snakin et al. 1996; Meyer 2002). Gully or rut formation is a common indicator of soil loss on ORV trails (Meyer 2002). Rut depth was used as a proxy for erosion. On longer trails (> 300 m) I took rut depth measurements every 100 meters. On shorter trails (<300 m) I took depth measurements at the beginning, midpoint and end of a trail. Longer trails accounted for 12 and shorter trails accounted of 16 of the total 28 trails (Table 2.1). I used a measuring tape to take measurements which were rounded to the nearest millimeter. For each trail I took an average of the total amount of erosion.

2.3.2 Traffic

I deployed magnetic Off-Highway Vehicle (OHV) counters (G3 OHV counters manufactured by TRAFx Research Ltd.) on a subset of trails. Funding limited the number of counters to four. I rotated counters throughout the study period to obtain replicates for all factors. Counters were deployed at 16 of 28 (just over half) of the trails; however one counter failed to start so data from only 15 trails were collected (Table 2.1). Counters were deployed May 25-August 30, 2010. The average length of time a counter was placed at any one trail was 19 (± 10 SD) days. I could not deploy counters on all trails, nor during the entire season. Thus, I also noted what was at the end of each trail (the "destination") as a potential proxy for traffic intensity. Destinations were lakes in 15 cases (which may have higher traffic to access fishing in early spring before I deployed counters), domestic cutovers in 3 cases (which may have zero to low traffic if no longer in active use), campsites in 6 cases, circling back to the road in 2 cases and overgrowth in 2 cases (Table 2.1).

2.3.3 Vegetation

To assess changes in off-trail vegetation species composition, line transects were established at the midpoint of each trail and run 50 metres at right angles away from the trail. Every 5 meters starting from the edge (beside wheel ruts or a visible path), 1 m^2 quadrats were laid. Quadrats were placed with the lower left corner at the appropriate metre demarcation. Within each quadrat all vascular species were identified and their percent cover estimated. Non-vascular species were categorized broadly as either moss or lichen. Their presence/absence was recorded and their percent cover estimated. See Figure 2.2 for graphical illustration of the vegetation sampling schematic. Plant life form classifications used in this study are presented in Table 1.2. Complete species lists are given in Tables B 1-5.

2.4 Statistical Analysis

2.4.1 Addressing Spatial Autocorrelation among Sampling Locations

Autocorrelation, or lack of independence among samples, is a common problem of ecological data. There are two types of autocorrelation: temporal and spatial (Legendre 1993), only the latter will be discussed here. Autocorrelation may be positive or negative. Positive autocorrelation among samples predicts homogeneity, conversely negative autocorrelation predicts heterogeneity. In the case of spatial autocorrelation, it is geographical distance that influences the homogeneity or heterogeneity of samples. In ecological systems positive spatial autocorrelation is usually present particularly at small spatial scales (Legendre 1993) i.e., samples that are close together are more likely to be similar. Positive autocorrelation over short distances may be due to migration or dispersal among individuals or to similar environmental conditions (Sokal and Oden 1978a).

Spatial autocorrelation has important implications for statistical analysis. If positive spatial autocorrelation is present there is a lack of independence among samples since the value of one sample can predict the value of a neighboring sample (Sokal and Oden 1978b; Legendre 1993; Diniz-Filho et al. 2003). Thus positive autocorrelation increases the tendency for Type I error or rejection of the null hypothesis when it is true (Legendre 1993; Diniz-Filho et al. 2003). Therefore spatial autocorrelation is an important consideration for researchers conducting field experiments or investigations. There are several analytical tools for measuring spatial autocorrelation within data; below I outline the techniques I used and the reasoning behind them.

2.4.2 Spatial Statistics: Join Counts

For nominal or categorical data join count statistics are appropriate (Cliff and Ord 1973 1981; Sokal and Oden 1978b; Fortin and Dale 2005). Join count statistics test if a join or edge connecting sampling points or localities is random based on the categorical variable(s) specified (Sokal and Oden 1978b; Fortin and Dale 2005). The test was first developed to map spatial patterns of disease in humans based on a county system (Fortin and Dale 2005). Each county would be assigned a colour variable, Black (B) or presence of disease or White (W) or absence of disease (Fortin and Dale 2005). This nomenclature is still in use today. Although referred to as “colour” any nominal variable may be used and join count statistics have also been extended beyond the strict binary case to k categories (Cliff and Ord 1973; 1981; Sokal and Oden 1978b; Fortin and Dale 2005). Spatial closeness or adjacency is defined by joins between “counties” or lattice data or a set of points (Cliff and Ord 1973; Fortin and Dale 2005). The degree of connectedness may be established if samples are spatially contiguous by neighborhood rules based on chess board moves (see Cliff and Ord 1973, 1981; Sokal and Oden 1978b for more detailed explanation). If samples are not spatially contiguous, different connectivity algorithms may be used: i.e., nearest neighbor network (Fortin and Dale 2005).

I used join count statistics to determine the level of spatial autocorrelation among my sample locations. I analyzed my sample points using join count statistics because I had previously categorized my sample sites into nominal categorical variables: NR/R, F/O and D/W. I used a binary weighing matrix to define the strength of connections between sample points. The choice of weighting matrix is important because it defines the strength of links between sets of points (Cliff and Ord 1973; 1981). When describing a weighting matrix, Cliff and Ord (1973) used a county system analogy. A weighting matrix defines the strength of links (railway and roads) between counties (or sets of points) (Cliff and Ord 1973). A binary weighting matrix is the simplest form of weighting matrix and is much less flexible than other weighting structures (Cliff and Ord 1973). A more generalized weighting structure is approximate if an investigator has *a priori* assumptions or knowledge about the strength of connections between sampling locations (Cliff and Ord 1973). To use the county analogy from Cliff and Ord (1973) again, a generalized weight structure would allow a researcher to account for size of counties and natural barriers between counties. I used nearest neighbor connectivity to define the degree of spatial closeness between my sample locations. I ran the join count statistic under non-free sampling assumption. Non-free sampling (or sampling without replacement) is the typical assumption for ecological data (Sokal and Oden 1978b). Non-free sampling assumes that each county (sample unit) has the same probability a priori of being W or B but that assignment is constrained (Cliff and Ord 1973; 1981). The assignment of one county to a particular color (category) affects the colour type of other counties (Cliff and Ord 1973; 1981). The alternative assumption is free sampling. Free sampling (or sampling with replacement) assumes the assignment of colours is independent for each county (Cliff and Ord 1973; 1981). Free sampling is often not applicable to biological data since it assumes knowledge of the parent distribution (of colour types) upon which sampling units are drawn (Sokal and Oden 1978b).

2.4.3 Join Count Analysis

I applied join count statistics to examine spatial autocorrelation among trails based on experimental factors (NR/R, F/O, D/W) using R 14.1 Statistical Software (2011). Sites were not chosen completely at random, sampling locations were constrained by requiring the presence of a pre-existing trail. Thus I used join count statistics to address spatial independence of trails. Join count statistics test if a join or edge connecting sampling points or localities is random, based on categorical variable(s) specified (Sokal and Oden 1978b; Fortin and Dale 2005). I used GPS coordinates (point data) taken at the beginning of each trail to act as a location of each trail. I tested for the presence of spatial autocorrelation between and among (i.e., at different levels within one factor) for cover type and moisture regime (F/O and D/W). Protected status (NR/N) was not tested for spatial autocorrelation for two reasons: 1) it was a legal rather than biological factor and therefore I was less concerned with possible confounding effects and 2) by definition trails within one of the two conditions (NR or R) will be close together and likely positively autocorrelated. I used a binary weighting matrix to define the strength of connections between sample points. The choice of the weighting matrix is important because it defines the strength of links between sets of points (Cliff and Ord 1973; 1981). I choose to use this simple and rigid weighting structure because I did not have *a priori* assumptions about the strength of connections between sampling locations. A more generalized weighting structure is appropriate if an investigator has *a priori* assumptions or knowledge about the strength of connections between sampling locations (Cliff and Ord 1973). I used nearest neighbor connectivity to define the degree of spatial proximity between my sample locations. I ran the join count statistic under non-free sampling assumption. Non-free sampling (or sampling without replacement) is the typical assumption for ecological data (Sokal and Oden 1978b).

2.4.4 Erosion

To test the relationship between erosion depth and the legal status, habitat and micro-habitat, I analyzed the data using Generalized Linear Models (GzLMs) and the information theoretic approach (corrected Akaike's Information Criterion (AICc) to account for low ratio of sample size to parameters) in R 14.1 Statistical Software (2011). All GzLMs had a Gaussian distribution and used the identity link. The response variable (erosion depth) was normalized via log transformation prior to analysis to allow for the fitting of GzLMs. Predictor variables included the *a priori* factors as well as the *post hoc* variable of destination (i.e., trails that end in lakes compared to all other types of endpoints). AIC allows for the selection of the 'best' model from a suite of competing models based on the principle of parsimony (Burnham and Anderson 2002). Because of the low ratio of sample size (n) to parameters (K), it was more appropriate to use the corrected AIC; AICc (Burnham and Anderson 2002). Thus this approach aided in the determination of which factor(s) were most influential in erosion depth. Only variables cover type and moisture were used in model selection. An interaction only model could not be fit due to sample size restrictions.

2.4.5 Traffic

Differences in ATV traffic level among and between factors (NR/R, F/O, and D/W) were tested with a nonparametric Kruskal-Wallis ANOVA in R 14.1 Statistical Software (2011). A nonparametric Kruskal-Wallis ANOVA is an appropriate test for comparing two or more non-normally distributed groups (Zar 1999).

2.4.6 Vegetation

2.4.6.1 Multi-Response Permutation Procedure (MRPP)

To test if my experimental groups (i.e. NR/R, F/O, D/W) were truly capturing differences in vegetation community composition, I performed a Multi-Response Permutation Procedure (MRPP) using PC-ORD version 6.0 (McCune and Mefford 2011). Multi-response permutation procedure is a type of permutation (a nonparametric statistical technique) that tests for a difference among two or more groups in one or more dimensions (Mielke and Berry 2007). The MRPP I used is an exact test, that is, exact p-values were calculated. In an exact test, a test statistic is calculated from observed (real) data linked to a suite of groups and the data are permutated over all possible combinations of those groups (Mielke and Berry 2007). Under the null hypothesis all combinations (partitions) of groups have an equal chance of occurrence (Mielke and Berry 2007). To clarify, the response(s) are coupled with the predictor (i.e., group) and it is the predictors (groups) that are permuted, not the individual responses for each group (Mielke and Berry 2007). The groups are compared using a weighted distance function (Mielke and Berry 2007); which may be specified by the researcher. For very large datasets where the calculation of exact p-values may be difficult "resampling" permutations may be used (Mielke and Berry 2007). In a resampling permutation only a subset of all possible permutation is examined and exact p-values are approximated (Mielke and Berry 2007). In my case I could apply an exact test which is preferable since exact p-values may be calculated. Another appealing feature of permutations is that they are "distribution-free." They do not make any assumptions about the distribution of the underlying population because p-values are calculated from observed data and tested against randomization (Mielke and Berry 2007). For full mathematical workings of the MRPP see Mielke (1984); for an ecological example see Biondini and colleagues (1985) and for tabular and illustrative graphical explanations see Mielke and Berry (2007).

I used a natural weighting as recommended by Mielke (1984) and a Sørensen (Bray-Curtis) distance metric to measure the difference in ecological distance between factors. In order to easily compare different multivariate techniques based on a distance measure, the distance measure used should be the same among all the tests (Clarke 1993). Thus I choose the same metric I used for MRPP. I also rank transformed the distance matrix. As ecological community heterogeneity increases, distance metrics can suffer a loss of sensitivity (McCune and Grace 2002). Rank transforming a distance matrix helps to correct for this and makes the results comparable to Non-Metric Multidimensional Scaling (Clarke 1993; McCune and Grace 2002). When the distance matrix is rank transformed in the PC-ORD package the null hypothesis is changed from "average within-group distance no smaller than expected by chance" to "no difference in average within-group rank of distances" (McCune and Mefford 2011). I tested for differences within and among the three factors (NR/R, F/O and D/W) using un-pooled data and pooled data. I pooled data by summing abundances across quadrats at the same position along the line transect within a given habitat (e.g., all quadrats across trails in a forest site at the 5 m mark were pooled). Thus, when data were pooled relative spatial position was maintained. Pooling was preferable since the multidimensional scaling algorithm performs poorly above 100 samples (Clark 1993).

By using MRPP I test for real ecological differences between my experimental groups and thus this test provides a robust validation for those groups. Moreover I compare un-pooled and pooled data using MRPP and thus lay the groundwork for further investigative techniques.

2.4.6.2 Non-Metric Multidimensional Scaling (NMDS)

To examine community gradients I used Non-Metric Multidimensional Scaling (NMDS) ordination and Polar (Bray-Curtis) ordination using PC-ORD version 6.0 (McCune and Mefford 2011). NMDS is considered a robust test for detecting ecological patterns since it makes no assumptions about underlying gradients and thus falls into the category of a “free ordination” technique (Minchin 1987; Clarke 1993; Peck 2010). Non-metric multidimensional scaling (NMDS) is an ordination technique that reduces high dimensionality data to a lower dimension (Clarke 1993; Borg and Groenen 2005). Like other ordination techniques this reduction in dimensionality allows the researcher to visually interpret data structure. The NMDS algorithm constructs a plot based on the distances between objects (Borg and Groenen 2005). Thus the relative proximity of points on a NMDS plot should correspond to actual distances (or (dis)similarities of predictors). For example if proximities represented dissimilarities, points further apart would be more dissimilar than points closer together (Kruskal and Wish 1978). NMDS is a useful tool for ecologists since the distance matrix is constructed from biologically meaningful data and a wide range a similarity coefficients are available (Clarke 1993). NMDS also considers the rank-order (rather than absolute value) of samples (Clarke 1993) which add robustness when considering subjective scales. NMDS uses an iterative process to determine the optimal solution (Kruskal and Wish 1978; in other words, a model that best fits to the data. The goodness-of-fit is measured by the amount of noise or error present in the data which in NMDS is termed “stress” (Kruskal and Wish 1978). The concept of stress can be thought of in terms of “badness-of-fit” since high stress is an indicator of a poor final solution (Kruskal and Wish 1978; Clarke 1993; Borg and Groenen 2005).

Kruskal and Wish (1978) use a highly informative landscape analogy to describe the conceptual framework behind NMDS. The starting point of the configuration may be specified but is often unknown. The starting configuration may be generated by use of a random number table (McCune and Mefford 2011). This is analogous to a blindfolded parachutist jumping from a plane which is flying at night (Kruskal and Wish 1978). Where the person lands on the ground (i.e., ordination space) is the starting point

(Kruskal and Wish 1978). Now on the ground the person heads downhill step by step. Each step is analogous to one iteration and heading downhill is seeking to minimize stress (Kruskal and Wish 1978). At each stop (or run) the person feels around for the strongest downhill direction and takes a step in that direction (Kruskal and Wish 1978). Eventually that person comes to a location where the terrain no longer goes downhill and stops; this is the final configuration/solution (Kruskal and Wish 1978).

To ordinate sites, a Sørenson (Bray-Curtis) dissimilarity matrix was calculated on species and life form abundance data. Data were not transformed prior to analysis; NMDS algorithm does not assume linearity among variables (McCune and Grace, 2002). I followed the general guidelines for conducting NMDS recommended by McCune and Grace (2002) and Peck (2010). It is important to note that there exists no firm statistical criterion for choosing the correct dimensionality (Kruskal and Wish 1978). I performed NMDS in a step-down from 1 through 6 dimensions, 3 separate times for each comparison. Each step-down run had a random starting configuration. The iteration maximum was 250, the stability criterion was <0.00001. Appropriate dimensionality was determined through the inspection of scree plots, Monte Carlo tests (250 iterations) of each dimensionality and inspecting the final stress of each dimension. Once an appropriate dimensionality is decided upon a number of runs must be conducted to avoid local stress minima (Clarke 1993). Upon determining the appropriate dimensionality I performed NMDS 5 times with the above parameters; each run had a random starting configuration. Of the 5 runs the starting coordinates of lowest stress solution were used as the starting coordinates for the final solution. The final solution was inspected against previous runs and if no discrepancies were found was considered the global optima. Plots had axes rotation making principal axes orthogonal.

NMDS allowed for the examination and comparison of overall plant species and life form community homogeneity between habitat types while making no assumptions of underlying ecological gradients. Thus it allows for the assessment of the inherent levels of community homogeneity between habitat types.

2.4.6.3 Polar (Bray-Curtis) Ordination

To test for the presence of the experimental gradient (i.e., off-trail/edge effect) I performed a polar (Bray-Curtis) ordination using PC-ORD version 6.0 (McCune and Mefford 2011). In contrast to NMDS, polar ordination is a "guided ordination" technique that assumes the presence of an ecological community gradient (Beals 1984; Peck 2010). Polar (or Bray-Curtis) ordination is similar to the above techniques in that it calculates a distance matrix (Beals 1984). Like NMDS the choice of distance metric may be specified by the researcher and a number of different measures are available (Beals 1984; McCune and Mefford 2011). Historically one of the main appeals of this test was speed since relatively simple calculations are required to perform this ordination (Beals 1984; McCune and Grace 2002). Indeed, in the original paper by Bray and Curtis (1957) ordination scores were found using a compass. Today with modern computers, computational speed is not as much of an issue. However polar ordination has another appealing feature: the selection of two reference points or poles. These reference points or poles (may also be referred to as endpoints) may be based on real or synthetic samples and either objectively or subjectively chosen (Beals 1984; McCune and Grace 2002). All other samples are projected onto each axis based in relation to the two reference points (Beals 1984). I used the Sørenson (Bray-Curtis) distance index and chose the subjective method to select reference points because I wished to test for the presence of a defined (by the researcher) community gradient. I selected the closest sample to the trail (i.e., the physical edge) as the first reference point and the furthest sample from the trail (i.e., the interior) as the second reference point.

Polar ordination allowed for the investigation of the experimental gradient (i.e., the off-trail impacts) within the different habitat types and thus allowed for the assessment of the strength of indirect (off-trail) impacts among the different habitat types.

Table 2.1: General Trail Characteristics of All-Terrain Vehicle (ATV) Trails discovered along Horse Chops Road

	Factor	Trail Number	Length (m)	Distance from the beginning of Horse Chops Road (km)	Trail Destination
Non-Reserve	Forest/Dry	1*	1140	3.7	Lake
	Forest/Dry	2*	241	5.0	Circle back to Horse Chops Road
	Forest/Dry	3	526	5.2	Lake
	Open/Wet	4*	634	6.0	Domestic Wood Cutting
	Forest/Wet	5*	224	6.5	Domestic Wood Cutting
	Open/Wet	6*	724	6.6	No Clear Destination
	Open/Dry	7*	653	8.0	Lake
	Open/Wet	8*	1415	8.1	Lake
	Open/Wet	9	660	10.0	Lake
	Forest/Dry	10*	98	10.1	Lake
	Forest/Dry	11	166	10.1	Lake
	Open/Wet	12	185	10.2	Circle back to Horse Chops Road
	Forest/Dry	13	69	11.1	Lake
	Forest/Dry	14	246	11.1	Lake
	Open/Wet	15	1070	11.2	Lake
	Forest/Wet	16	239	11.6	Lake
Reserve	Forest/Wet	17*	163	11.6	No Clear Destination
	Open/Wet	18*	225	11.9	Lake
	Open/Wet	19*	1233	13.9	Campsite
	Open/Dry	20*	128	13.9	Lake
	Open/Dry	21	64	14.0	Campsite
	Forest/Dry	22*	706	19.2	Campsite
	Forest/Dry	23	100	19.8	Domestic Wood Cutting
	Forest/Dry	24*	83	19.8	Campsite
	Forest/Dry	25*	684	20.0	Lake
	Forest/Wet	26	105	20.1	Campsite
	Forest/Dry	27	29	20.8	Campsite
	Open/Dry	28*	1030	21.4	Lake

*Denotes a traffic counter was deployed at this trail. The traffic counter for trail 8 failed to start, no data were collected

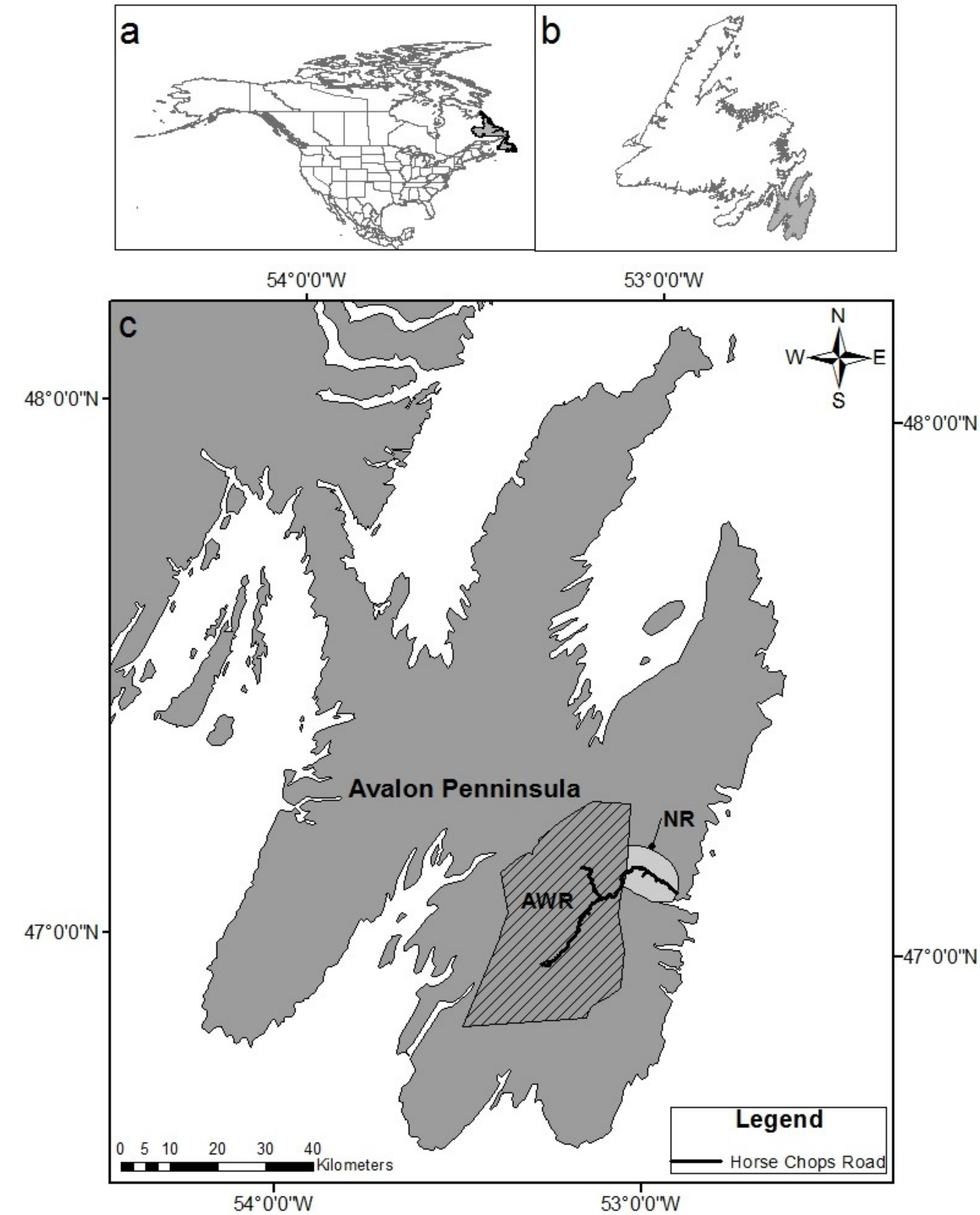

Figure 2.1: Map of the general study area location on the southeast Avalon Peninsula, island of Newfoundland, Canada. a. **The location of the province of Newfoundland and Labrador (shaded grey) within North America.** b. **the island of Newfoundland, the Avalon Peninsula is shaded in grey.** c. **Detail of the Avalon Peninsula. The Avalon Wilderness Reserve (AWR) is crosshatched. The Non-Reserve (NR) portion of the study area is shaded light grey. Horse Chops road is the black line.**

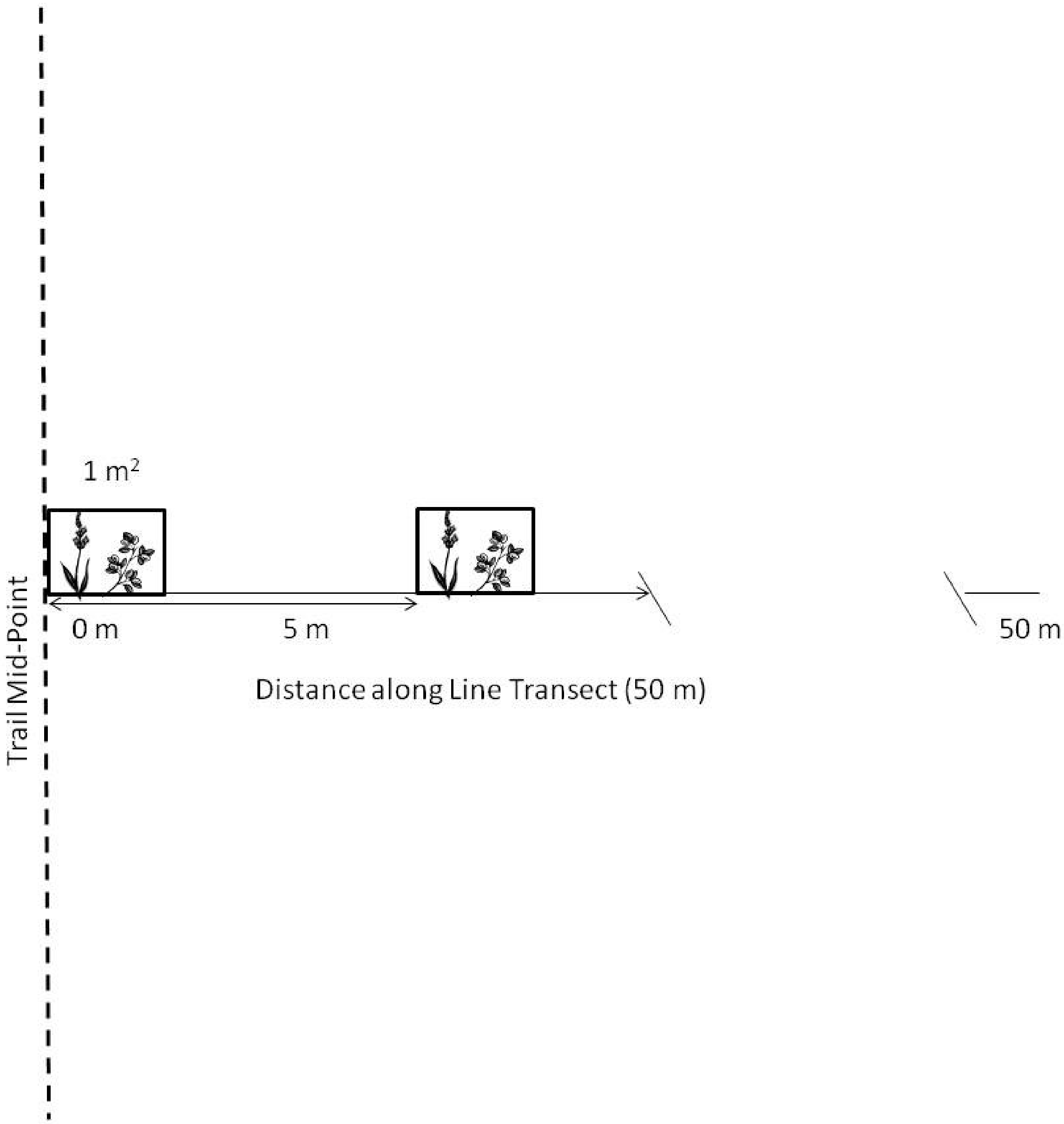

Figure 2.2: Vegetation Sampling Schematic. Dashed line indicates an ORV/ATV trail. Arrows indicate direction of transects and boxes represent quadrats.

3 Chapter 3: Results

3.1 Spatial Autocorrelation

In general there was not a high degree of positive autocorrelation among sites of the same treatment factor (Table 3.1). Among the experimental factors F-F had the highest amount of positive spatial autocorrelation (Table 3.1). F/D-F/D and W-W also had a notable level of positive autocorrelation (Table 3.1). Therefore I am confident that my study sites are reasonably spatially independent.

3.2 Erosion

The amount of erosion differed significantly by cover type, moisture level and habitat type. Forested trails had significantly less erosion than open trails ($t_{(26, 27)}$=20.394, p<0.001; Figure 3.1a and Table 3.2). These results support hypothesis 1a; on-trail erosion was predicted to be highest on open trails. Dry trails had significantly less erosion than wet trails ($t_{(26, 27)}$=21.077, p<0.001; Figure 3.1b and Table 3.2).These results support hypothesis 1b; on-trail erosion was predicted to be higher on wetter trails. Forested and dry trails had significantly less erosion compared to all other habitat types ($t_{(24, 27)}$=16.931, p<0.001; Figure 3.1c and Table 3.2). In other words, trails in dry forested habitats had significantly less rutting than trails in other habitat types. The other habitat types did not differ significantly in erosion amount from one another; wet forested trails ($t_{(24, 27)}$=0.435, p=0.667, dry open trails $t_{(24, 27)}$=0.385, p=0.704 and wet open trails $t_{(24, 27)}$= 1.129, p=0.270; Figure 3.1c and Table 3.2). In other words, trails in wet forested, dry open (heath), and wet open (bog) habitats had similar amounts of rutting. Trails within the AWR showed a significant difference in erosion level compared to trails outside AWR

($t_{(26, 27)}$=20.332, p<0.001) but this difference became highly non-significant when habitat type was accounted for ($t_{(23, 27)}$=0.301, p=0.766; Table 3.2). Trails that ended in lakes showed a significant difference in erosion level compared to trails that ended in other destinations (i.e., camp sites, wood cutting areas), $t_{(26, 27)}$=21.606, p<0.001 but became highly non-significant when habitat type was taken into account ($t_{(23, 27)}$=1.377, p=0.182; Table 3.2). These results indicate that trail destination is not a predictor of erosion. Overall erosion depth ranged from 3.25-21.5 cm, mean=11.21 cm, S.D. =4.83 cm. Models that included moisture level only and cover type only ranked as top models in model selection (i.e., had Δ_i<2; Table 3.3). The moisture only model had slightly more weight of evidence (w_i= 0.4357) compared to the cover only model (w_i=0.3985) (Table 3.3).

3.3 Traffic

All trails had vehicular traffic. Traffic counts per day ranged from 0.2-13.6 (mean=3.18 and S.D. =3.62). There was no difference in the amount of traffic on forested trails compared to open trails (Table 3.4) or on dry trails compared to wet trails (Table 3.4). There was no difference in the amount of traffic on non-reserve trails compared to reserve trails (Table 3.4) or among the different habitat types (Table 3.4). This indicates that during the time period of counter deployment ORV/ATV users showed no preference for a particular habitat type and utilized the trails within the AWR and outside the AWR at similar intensities. Therefore protected status did not influence use. I predicted direct effects on trails (i.e., rutting) would be greater with increasing use (hypothesis 1c) and indirect effects on trails (i.e., edge effects) would be greater with increasing use (hypothesis 2c). However hypotheses 1c and 2c could not be tested directly since there were no significant differences in traffic volume (intensity of use) among the various experimental factors. Anecdotal evidence for early spring use before counter deployment by destination is presented in Appendix C.

3.4 Vegetation

3.4.1 Multi-Response Permutation Procedure Results: Unpooled Data

Table 3.5 shows MRPP results for species and life form levels of analysis based on unpooled data. Unpooled data included all quadrat samples (see sample size [n] in Table 3.5). MRPP analysis confirmed that all experimental groups/factors were significantly different from one another both in community species composition and life form composition (Table 3.5). The T statistic is a measure of groups separation, the more negative the value, the greater the separation (McCune and Grace, 2002). Separation trends were similar at both the species and life form level of analysis; with most separation between groups among the habitat types ($T=-44.917$, $p<0.001$ [species], $T=-38.886$, $p<0.001$ [life form]) and least separation between reserve and non-reserve ($T=-10.058$, $p<0.001$ [species], $T=-11.046$, $p<0.001$ [life form]). A is the chance-corrected within group agreement, a measure of group homogeneity (McCune and Grace, 2002). If $A=1$ all items within the group are identical (McCune and Grace, 2002). If $A=0$, heterogeneity within groups is equal to chance (McCune and Grace, 2002). Chance-corrected within group agreement, A, did not differ markedly between the species and life form level analyses (Table 3.5). Factors had similar levels of community homogeneity at species and life level analyses.

3.4.2 Multi-Response Permutation Procedure Results: Pooled Data

MRPP analysis confirmed that all pooled experimental groups were significantly different from one another both in community species composition and life form composition (Table 3.6). Separation trends were similar at both the species and life form level of analysis; with most separation between groups among the habitat types (T = -16.200, p<0.001 [species], T = -17.655, p<0.001 [life form]) and least separation between reserve and non-reserve (T = -7.413, p<0.001 [species], T = -7.221, p<0.001 [life form]). Greatest dissimilarity in species composition among the habitat types confirmed the assumption of community level differences between the experimental factors. Least dissimilarity in species composition between the NR and R confirms the assumption that these two areas differ little ecologically and that the main difference is in protect status. Chance-corrected within group agreement, A, did not differ markedly between the species and life form level analyses (Table 3.5; Table 3.6). Factors (NR/R, F/O, D/W and All Habitat Types) had similar levels of community homogeneity at species and life form level analyses (Table 3.5; Table 3.6).

3.4.3 Non-Metric Multidimensional Scaling Results: Species Level Analysis

NMDS ordination showed a distinct separation between cover types (Figure 3.2a). Open samples were more clustered than forested samples (Figure 3.2a). NMDS ordination yielded a 2-dimensinal solution that explained 90.2% of the total variation. The final stress was 12.05516 after 49 iterations (Monte Carlo stress test p=0.004). Among the forested samples there is clear separation between the edge (For_1) and interior samples (For_10, For_11). NMDS ordinations showed a distinct separation between moisture levels (Figure 3.2b). NMDS yielded a 3 dimensional solution explaining 82.2% for the total variation. For ease of interpretability only a 2 dimensional solution is shown in Figure 3.2b; the first two axes explain 63.4% of the total variation. The final stress for the 3 dimensional solution was 12.130 after 30 iterations (Monte Carlo stress test p=0.004). Among the wet samples there is a clear separation between the interior samples (Wet_9, Wet_10, Wet_11) and the edge and mid-way samples (Figure 3.2b). Among the dry samples, the closest to the trail (Dry_1) is distinctly separated from the rest of the samples. There was not a strong segregation between samples based on protected status. NMDS ordination of samples based on protected status yielded a 3 dimensional solution explaining 76.2% of the total variation. For ease of interpretability only the 2 dimensional solution is shown in Figure 3.2c; which explains 63.5% of the total variation. The final stress was 13.475 after 28 iterations (Monte Carlo stress test p= 0.0359). NMDS ordination of all habitat types showed a clear clustering of O/W samples while other habitat types were much more scattered. Ordination yielded a 3 dimensional solution explaining 83.2% of the total variation. For ease of interpretability only the 2 dimensional solution is shown in Figure 3.2d, which explains 72.2% of the total variation. Final stress was 13.658 after 63 iterations (Monte Carlo stress test p=0.004).

3.4.4 Non-Metric Multidimensional Scaling Results: Life Form Level Analysis

NMDS ordination showed a distinct separation between cover types (Figure 3.3a); indicating distinct vegetation community types. Open samples were more clustered than forested samples (Figure 3.3a); indicating a more homogeneous community in open habitats along the experimental gradient. NMDS yielded a 2 dimensional solution explaining 97.0% of the variation. The final stress was 6.196, achieved after 53 iterations (Monte Carlo stress test p=0.004).There is also a clear separation between edges samples (i.e., forest quadrats 1, 2 and 3) and interior samples (i.e., forest quadrats 9, 10, 11). For the open cover type, edges samples were more tightly clustered with mid-way samples but there is a clear separation of interior samples (open quadrats 9, 10, 11). This indicates a slower rate of species turnover (with distance from trail) in open habitats compared to forested habitats. NMDS ordination showed a distinct separation between moisture levels (Figure 3.3b); indicating distinct vegetation community types. Wet samples were more clustered than dry samples (Figure 3.3b); indicating a more homogeneous community in wet sites along the experimental gradient. NMDS yielded a 2 dimensional solution which explained 92.0% of the variation. The final stress was 10.336 achieved after 45 iterations (Monte Carlo stress test p=0.004). There is a clear separation between edge samples (dry quadrats 1, 2, 3) and interior samples (dry quadrats 9, 10, 11). For the wet samples edges samples are clustered with mid-way samples however there is clear separation of interior samples (wet quadrats 10, 11). This indicates a slower rate of species turnover (with distance from trail) in wet sites compared to dry sites. NMDS ordination of protected status yielded a 3-dimensional solution explaining 96.9% of the variation; only the 2-dimensional solution is shown for ease of interpretability, and this explains 88.7% of the variation (Figure 3.3c). The final stress for the 3 dimensional solution was 5.359 after 70 iterations (Monte Carlo stress test p=0.004). Separation between reserve and non-reserve samples is clear (Figure 3.3c). There is some separation between AWR edge samples (quadrats 1, 2 and 3) and interior samples (quadrats 9, 10, 11) (Figure 3.3c). Only interior samples (quadrats 8-11) segregate in the non-protected samples (Figure 3.3c). Both the reserve and non-reserve showed detectable species turnover (with distance from the trail) however turnover was more pronounced in the reserve. NMDS ordination of all habitat types yielded a 3 dimensional solution explaining 94.5% of the variation. Only the 2 dimensional solution is shown in Fig. 3.3d for ease of interpretability, which explains 78.9% for the variation. Final stress for the 3 dimensional solution was 8.211 after 69 iterations (Monte Carlo stress test p=0.004). The Open/Wet habitat type is highly clustered but interior samples (quadrats 10 and 11) clearly separate (Figure 3.3d). The Open/Dry habitat type is more scattered, showing no clear separation between edges and interior samples. The Forest/Dry habitat type is highly scattered but there is still visible separation between edge samples (quadrats 1, 2, 3) and interior samples quadrats 9, 10, 11) (Figure 3.3d). The Forest/Wet habitat type is also highly scattered and there is unclear separation between edge and interior samples. Upon comparison of all habitat types using NMDS ordination only Forest/Dry and Open/Wet showed clear community gradients. NMDS ordination results do not support hypotheses 2 a and 2b since strongest edge (species turnover gradient) was detected in forested and dry samples.

3.4.5 Polar Ordination Results

There was a strong detectable life form community gradient for edge to interior seen in Forested, Open, Dry, Wet, Forest/Dry and Open/Wet habitats. In polar ordination endpoints were quadrat 1 (edge) and quadrat 11 (interior). Polar ordination of forested samples explained 85.8% of the total variation (Figure 3.4a). Polar ordination of open samples explained 80.8% of the variation (Figure 3.4b). Polar ordination of dry samples explained 81.5% of total variation (Figure 3.4c). Polar ordination of wet samples explained 76.8% of total variation (Figure 3.4d). Polar ordination of F/D samples explained 79.7% of total variation (Figure 3.4e). Ordination of Forest/Wet samples only explained 50.1% of the total variation (Figure 3.4f). It should be noted that F/W habitat type had the smallest number of samples of the various types (n=4). Interestingly, the amount of variation explained in polar ordination of O/D samples was only 16.3% (Figure 3.4g). Polar ordination of O/W samples explained 88.3% of the total variation (Figure 3.4h).

Across the various habitat types, life form groups that showed the strongest association with the edge were shrubs, graminoids and mosses (Table 3.7). Life form groups that had the weakest association with the edge were ferns and lichens (Table 3.7).

3.5 Vegetation Results Summary

MRPP results statistically validated the appropriateness of the experimental groups; there are real ecological differences between groups. MRPP results justified the decision to pool data as unpooled and pooled data showed similar trends in overall community homogeneity at both the species and life form levels of analysis. NMDS ordination showed similar patterns at both the species and life form level of analysis. NMDS results in a comparison of forested and open habitats, forested sites were more heterogeneous than open sites. There was also a sharper edge (a clearer separation between edges and interior samples) in forested sites compared to open sites. Dry sites were more heterogeneous than wet sites and had a sharper edge. Wet sites showed clustering of edge and mid-way samples however there was a clear separation from interior samples. Upon comparison of all four habitat types bogs showed the highest degree of homogeneity. Polar ordination detected the strongest experimental gradient (trail impact) in bog sites followed by forest sites. The weakest gradient was detected in heath sites. The strong community gradient open sites may be driven by bogs sites given the weak gradient detected in heath sites. The strong community gradient of dry sites may be driven by forest sites given the weak gradient detected in heath sites. The strong community gradient of wet sites may be driven by bog sites given the weak gradient of wet forested sites. The weak gradient of wet forest may be a function of small sample size.

Table 3.1: Summary of join-count analysis results of same habitat type (i.e. same colour, BB). Asterisks denote p-values ≤ 0.05 significance level indicating presence of positive spatial autocorrelation.

	Sample Size (n)	Observed Joins BB	Expected Joins BB	Std. Error	Join Count Autocorrelation BB	p-value
Forest: Forest						
Neigh 1	16	8	5.71429	1.5740	1.8219	0.0342*
Neigh 2	16	15	11.7460	3.6639	1.7000	0.0446*
Neigh 3	16	22	17.7778	5.7010	1.7683	0.0385*
Neigh 4	16	27	22.8571	6.6646	1.6048	0.0543
Neigh 5	16	30	27.6190	7.4944	0.8697	0.1922
Open: Open						
Neigh 1	12	5	3.1426	1.3508	1.5979	0.0550
Neigh 2	12	8	6.4603	2.9439	0.8974	0.1848
Neigh 3	12	13	9.7778	4.4507	1.5274	0.0633
Neigh 4	12	15	12.571	5.2803	1.0569	0.1453
Neigh 5	12	17	15.190	5.9818	0.7399	0.2297
Dry: Dry						
Neigh 1	17	8	6.4762	1.5657	1.2178	0.1116
Neigh 2	17	14	13.3122	3.7154	0.3568	0.3606
Neigh 3	17	19	20.1481	5.8271	-0.4756	0.6828
Neigh 4	17	22	25.9048	6.7846	-1.4991	0.9331
Neigh 5	17	28	31.3016	7.6138	-1.1965	0.8843
Wet: Wet						
Neigh 1	11	5	2.6190	1.2395	2.1386	0.0162*
Neigh 2	11	7	5.3836	2.6635	0.9904	0.1610
Neigh 3	11	11	8.1481	4.0005	1.4258	0.0770
Neigh 4	11	14	10.476	4.7623	1.6148	0.0532
Neigh 5	11	17	12.659	5.4039	1.8675	0.0309*
Forest/Dry: Forest/Dry						
Neigh 1	12	6	3.1429	1.3508	2.4583	0.0070*
Neigh 2	12	10	6.4603	2.9439	2.0630	0.0196*
Neigh 3	12	12	9.7778	4.4507	1.0533	0.1461
Neigh 4	12	13	12.571	5.2803	0.1865	0.4260
Neigh 5	12	15	15.190	5.9818	-0.0779	0.5310
Forest/Wet: Forest/Wet						
Neigh 1	4	1	0.2857	0.2378	1.4648	0.0715
Neigh 2	4	1	0.5873	0.4715	0.6010	0.2739
Neigh 3	4	1	0.8889	0.6805	0.1347	0.4464
Neigh 4	4	1	1.1429	0.8272	-0.1571	0.5624
Neigh 5	4	1	1.3812	0.9482	-0.3912	0.6522
Open/Dry: Open/Dry						
Neigh 1	5	1	0.4762	0.3691	0.8622	0.1943
Neigh 2	5	1	0.9788	0.7392	0.0246	0.4902
Neigh 3	5	1	1.4814	1.0723	-0.4650	0.6790
Neigh 4	5	1	1.9048	1.2998	-0.7936	0.7863
Neigh 5	5	1	2.3016	1.4880	-1.0670	0.8570

	Sample Size (n)	Observed Joins BB	Expected Joins BB	Std. Error	Join Count Autocorrelation BB	p-value
Open/Wet: Open/Wet						
Neigh 1	7	2	1.0000	0.6667	1.2247	0.1103
Neigh 2	7	2	2.0556	1.3636	-0.0476	0.5190
Neigh 3	7	5	3.1111	1.9996	1.3358	0.0908
Neigh 4	7	6	4.0000	2.4103	1.2882	0.0988
Neigh 5	7	8	4.8333	2.7517	1.909	0.02813*

Neigh=Neighbour

Table 3.2: All explanatory models fitted for the response variable erosion depth

Model	Factor	β	Std. Error	t value	Deviance		95% C.I.	P value
					Null df=27	Residual df=26		
Cover Type	Forest (intercept)	0.98196	0.04815	20.394	0.9950	0.9645	0.8876- 1.0763	<0.001
	Open	0.06661	0.07355	0.906			-0.0775 0.2108	0.373
Moisture	Dry (intercept)	0.98145	0.04656	21.077	0.99491	0.95836	0.8902-1.0727	<0.001
	Wet	0.07398	0.07429	0.996			-0.0716 -0.2196	0.328
					Null df=27	Residual df=24		
Habitat	Dry Forest (intercept)	0.96950	0.05726	16.931	0.99491	0.94436	0.8573-1.0817	<0.001
	Wet Forest	0.04985	0.11453	0.435			-0.1746 -0.2743	0.667
	Dry Open	0.04061	0.10559	0.385			-0.1663-0.2476	0.704
	Wet Open	0.10654	0.09434	1.129			-0.0786- 0.2914	0.270
					Null df=27	Residual df=26		
Reserve	Non-Reserve	1.02392	0.05036	20.332	0.99491	0.98910	0.9252- 1.1226	<0.001
	Reserve	-0.02888	0.07391	-0.391			-0.1737 0.1160	0.699
					Null df= 27	Residual df=23		
Habitat + Reserve	Dry Forest/Non-Reserve (intercept)	0.97960	0.06735	14.545	0.99491	0.94066	0.8476-1.1116	<0.001
	Wet Forest	0.05792	0.11981	0.483			-0.1769-0.2927	0.633
	Dry Open	0.04505	0.10866	0.415			-0.1679-0.2580	0.682
	Wet Open	0.10337	0.09676	1.068			-0.0862-0.2930	0.296
	Reserve	-0.02422	0.08060	-0.301			-0.1822-0.1338	0.766
Habitat + Reserve (re-level)	Dry Forest/ Reserve (intercept)	0.95537	0.07496	12.745	0.99491	0.94066	0.8085- 1.1023	<0.001
	Wet Forest	0.05792	0.11981	0.483			-0.1769- 0.2927	0.633
	Dry Open	0.04505	0.10866	0.415			-0.1679- 0.2580	0.682
	Wet Open	0.10337	0.09676	1.068			-0.0863- 0.2930	0.296
	Non-Reserve	0.02422	0.08060	0.301			-0.1338- 0.1822	0.766
					Null df=27	Residual df=26		
Destination	Lake (intercept)	1.05523	0.04884	21.606	0.99491	0.93030	0.9595-1.1520	<0.001

Model	Factor	β	Std. Error	t value	Deviance		95% C.I.	P value
	Other	-0.09632	0.07168	-1.344			-0.2368- 0.0441	0.191
					Null df=27	Residual df=23		
Habitat + Destination	Dry Forest/ Lake (intercept)	1.01307	0.06451	15.704	0.99491	0.87241	0.8866-1.1395	<0.001
	Wet Forest	0.08470	0.11526	0.735			-0.1412-0.3106	0.470
	Dry Open	0.03887	0.10368	0.375			-0.1643-0.2421	0.711
	Wet Open	0.10779	0.09263	1.164			-0.0738-0.2893	0.256
	Other	-0.10455	0.07592	-1.377			-0.2534-0.0442	0.182
Habitat + Destination (re-level)	Dry Forest/ Other	0.90851	0.07157	12.694	0.99491	0.87241	0.7682-1.0488	<0.001
	Wet Forest	0.08470	0.11526	0.735			-0.1412-0.3106	0.470
	Dry Open	0.03887	0.10368	0.375			-0.1643-0.2421	0.711
	Wet Open	0.10779	0.09263	1.164			-0.0738-0.2893	0.256
	Lake	0.10455	0.07592	1.377			-0.0442-0.2534	0.182

Table 3.3: Model selection results for models that predict log-transformed erosion levels (cm) for ATV trails (n=28) found within the Avalon Wilderness Reserve and outside the reserve in the adjacent surrounding area.

Model	Log-Likelihood	K	AIC_c	Δi	ωi
Moisture	7.516059	3	-8.0321	0.0000	0.4357
Cover Type	7.426769	3	-7.853537	0.1786	0.3985
Moisture + Cover Main Effects	7.716453	4	-5.693776	2.3383	0.1353
Global	7.722125	5	-2.716977	5.3151	0.0305

Table 3.4: Differences in traffic level among Non-Reserve (n=7) and Reserve (n=8), forested (n=8) and open (n=7), dry (n=9) and wet (n=5) areas and all habitat types (forest/dry n=6, forest/wet n=2, open/dry n=4 and open/wet n=3) tested using Kruskal-Wallis ANOVA.

Comparison	χ^2	df	p-value
Forest: Open	0.8602	1	0.3537
Dry: Wet	0.0538	1	0.8166
Non-Reserve: Reserve	0.1647	1	0.6849
All Habitat Types	1.4692	3	0.6894

Table 3.5: Multi Response Permutation Procedure comparisons of species and life form community composition among and between factors. The T statistic is a measure of groups separation, the more negative the value, the greater the separation (McCune and Grace, 2002). A is the chance-corrected within group agreement, a measure of group homogeneity (McCune and Grace, 2002). If A=1 all items within the group are identical (McCune and Grace, 2002). If A=0, heterogeneity within groups is equal to chance (McCune and Grace, 2002). Data are unpooled.

			Species			Life Form		
		n	T	A	p	T	A	p
Cover Type	Forest	120	-35.473	0.100	<0.001	-34.169	0.095	<0.001
	Open	124						
Moisture	Dry	140	-29.107	0.082	<0.001	-24.437	0.068	<0.001
	Wet	104						
Reserve	Non-Reserve	130	-10.058	0.028	<0.001	-11.046	0.031	<0.001
	Reserve	114						
All Habitat Types	Dry Forest	85	-44.917	0.221	<0.001	-38.886	0.188	<0.001
	Wet Forest	35						
	Dry Open	55						
	Wet Open	69						

Table 3.6: MRPP comparisons of species and life form community composition among and between factors. The T statistic is a measure of groups separation, the more negative the value, the greater the separation (McCune and Grace, 2002). A is the chance-corrected within group agreement, a measure of group homogeneity (McCune and Grace, 2002). If A=1 all items within the group are identical (McCune and Grace, 2002). If A=0, heterogeneity within groups is equal to chance (McCune and Grace, 2002). Data are pooled.

		n	Species			Life Form		
			T	A	P	T	A	P
Cover Type	Forest	11	-11.456	0.375	<0.001	-10.439	0.317	<0.001
	Open	11						
Moisture	Dry	11	-9.351	0.311	<0.001	-9.003	0.274	<0.001
	Wet	11						
Reserve	Non-Reserve:	11	-7.413	0.241	<0.001	-7.221	0.219	<0.001
	Reserve	11						
All Habitat Types	Dry Forest	11	-16.200	0.482	<0.001	-17.654	0.480	<0.001
	Wet Forest	11						
	Dry Open	11						
	Wet Open	11						

Table 3.7: Correlation results of Polar Ordination life form analysis. Negative correlation indicates association with the edge. Correlations over 0.7 are bolded for emphasis.

Habitat Type	Pearson's r							Kendall's tau						
	Trees	Shrubs	Herbs	Ferns	Gramin	Moss	Lichen	Trees	Shrubs	Herbs	Ferns	Gramin	Moss	Lichen
Forest	0.18	**-0.964**	-0.579	-0.213	-0.638	**-0.769**	-0.603	0.33	**-0.855**	-0.455	-0.081	-0.486	-0.636	-0.661
Open	-0.008	-0.693	-0.638	-0.234	**-0.843**	**-0.837**	0.346	0.110	-0.382	-0.600	-0.231	**-0.891**	-0.477	0.382
Dry	0.255	**-0.970**	**-0.722**	-0.062	**-0.789**	**-0.709**	0.178	0.127	**-0.855**	-0.455	-0.065	**-0.771**	-0.418	0.127
Wet	-0.146	-0.564	-0.283	0.087	**-0.948**	**-0.872**	-0.372	-0.150	**-0.818**	-0.130	-0.085	**-0.709**	**-0.745**	-0.359
Forest/Dry	0.189	**-0.962**	-0.590	-0.007	-0.656	**-0.871**	-0.623	0.127	**-0.917**	-0.382	-0.070	**-0.756**	-0.673	-0.561
Forest/Wet	-0.156	**-0.978**	0.095	-0.083	0.096	0.028	0.332	-0.101	**-0.964**	-0.018	-0.185	-0.019	-0.208	0.370
Open/Dry	0.320	0.514	-0.319	0.014	**-0.801**	0.544	0.492	0.114	0.294	-0.164	-0.182	-0.278	0.330	0.382
Open/Wet	-0.127	**-0.860**	-0.560	-0.248	**-0.923**	**-0.856**	-0.442	0.022	-0.527	-0.527	-0.217	-0.564	**-0.881**	-0.110

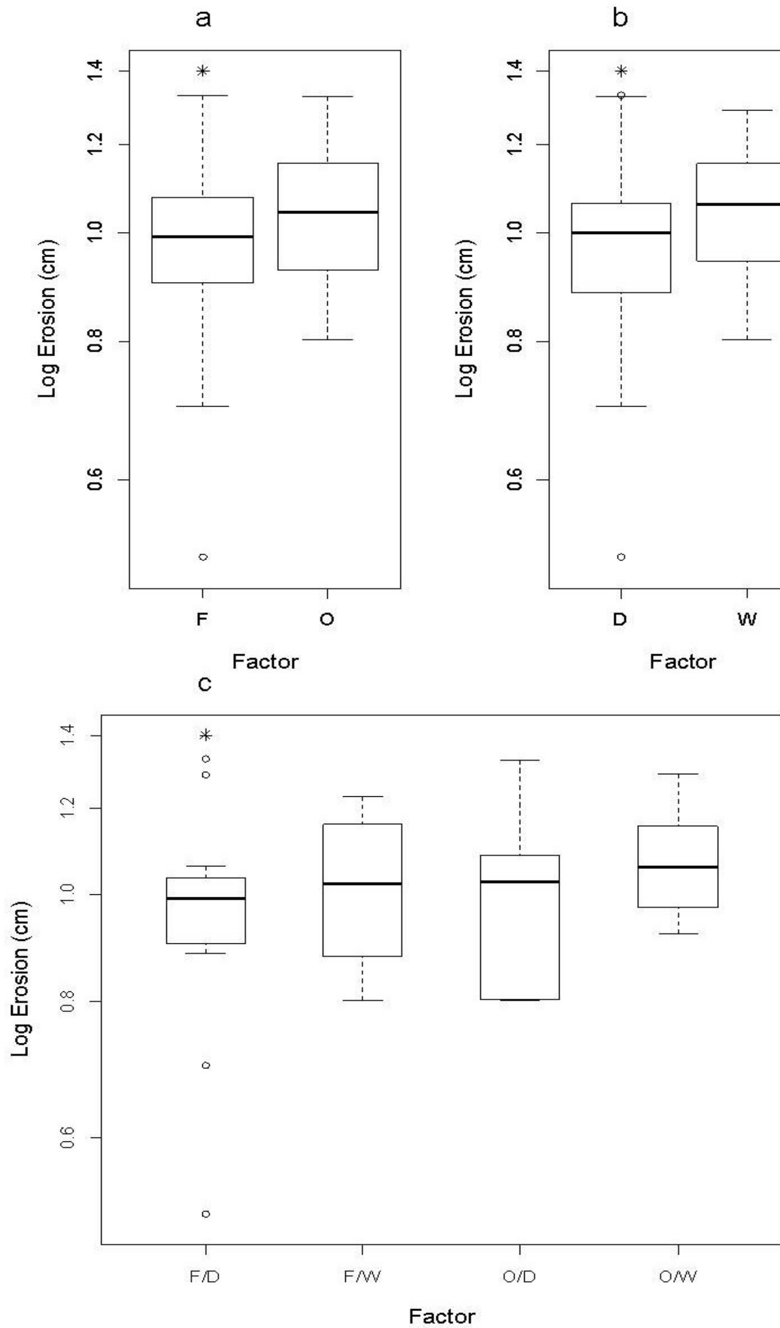

Figure 3.1: Box and whisker plots of erosion depth (cm) for the different habitat types. Horizontal bar is the median, box is the interquartile range, whiskers are the highest and lowest extremes, circles are outliers (1.5-3 box lengths from either end). Asterisks denote significance at p<0.001. Panel a: factor F is forest (n=16), factor O is open (n=12). Panel b: factor D is dry (n=17), factor W is wet (n=11). Panel c: factor F/D is dry forest (n=12), F/W is wet forest (n=4), O/D is dry open or heath (n=5) and O/W is wet open or bog (n=7).

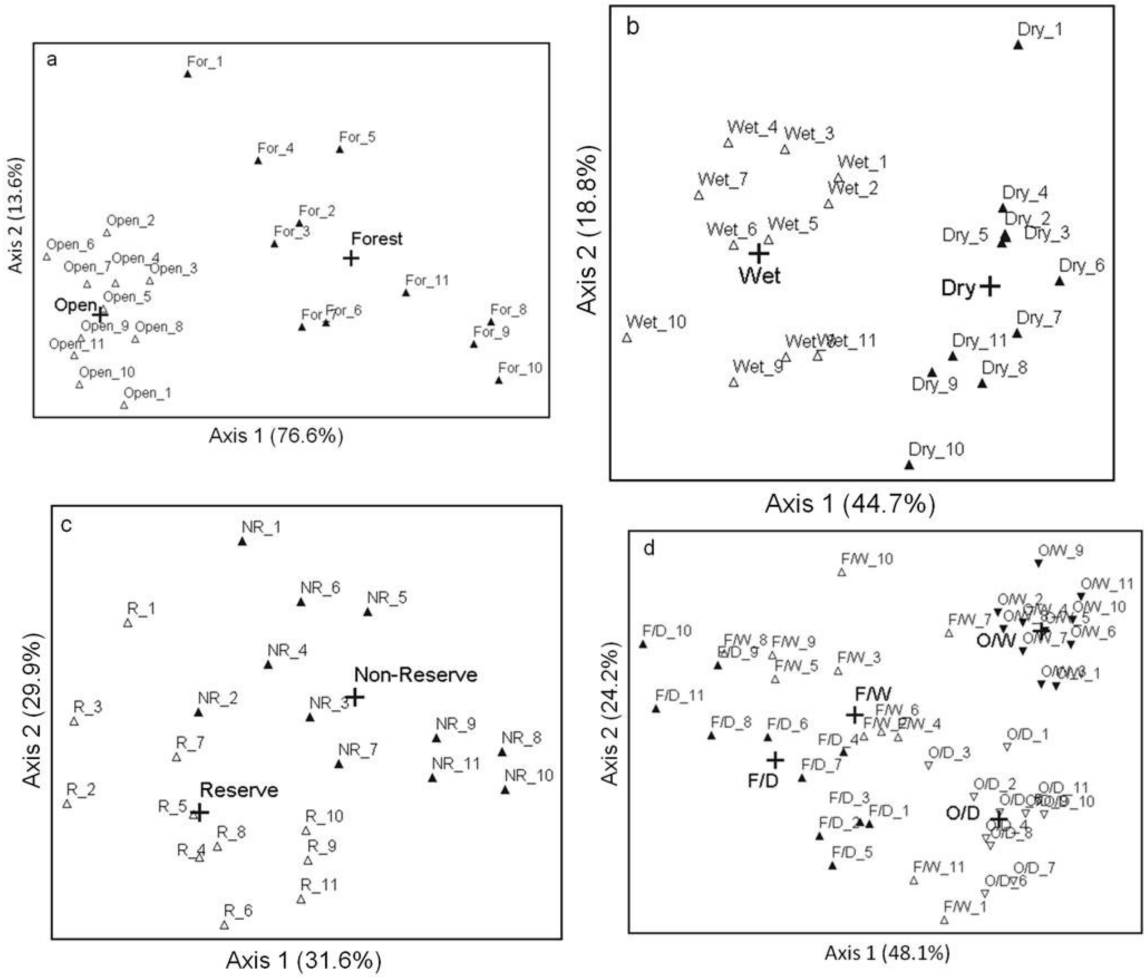

Figure 3.2: NMDS ordinations of species level community analysis among factors. Triangles denote quadrats, numbering indicates their spatial position along the line transect. The number 1 denotes the closest position to the trail (i.e. the edge) and the number 11 denotes the further position from the trail (i.e. the interior). Crosses denote group centroids. Triangles that are close together have more similar life form assemblages than triangles that are further apart. Panel a: comparison of forest samples (closed triangles ▲) to open samples (open triangles △). Panel b: comparison of dry samples (closed triangles ▲) to wet samples (open triangles △). Panel c: comparison of non-reserve samples (closed triangles ▲) to reserve samples (open triangles △). Panel d: comparison of forested dry samples (closed triangles▲), forested wet samples (open triangles △), open dry samples (inverted open triangles▽) and open wet samples (inverted closed triangles ▼).

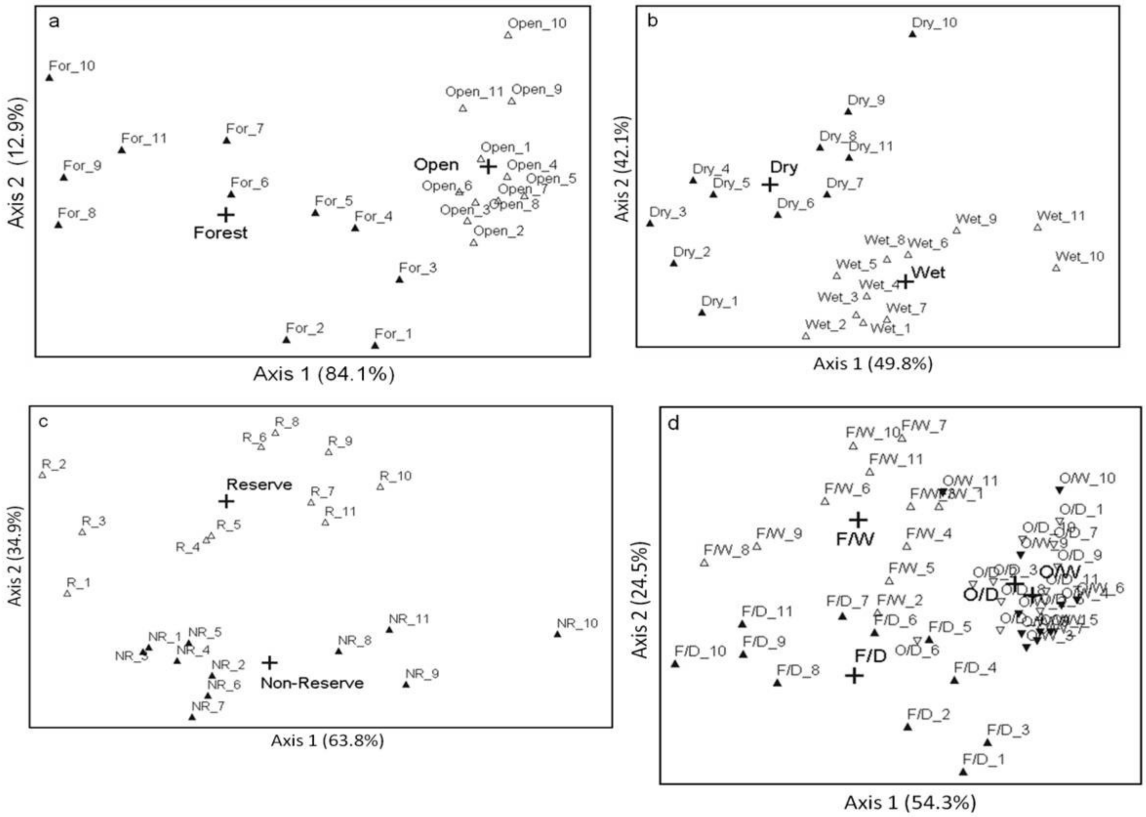

Figure 3.3: NMDS ordinations of life form community level analysis among factors. Triangles denote quadrats, numbering indicates their spatial position along the line transect. The number 1 denotes the closest position to the trail (i.e. the edge) and the number 11 denotes the further position from the trail (i.e. the interior). Crosses denote group centroids. Triangles that are close together have more similar life form assemblages than triangles that are further apart. Panel a: comparison of forest samples (closed triangles ▲) to open samples (open triangles △). Panel b: comparison of dry samples (closed triangles ▲) to wet samples (open triangles △). Panel c: comparison of non-reserve samples (closed triangles ▲) to reserve samples (open triangles △). Panel D: comparison of forested dry samples (closed triangles▲), forested wet samples (open triangles △), open dry samples (inverted open triangles▽) and open wet samples (inverted closed triangles ▼).

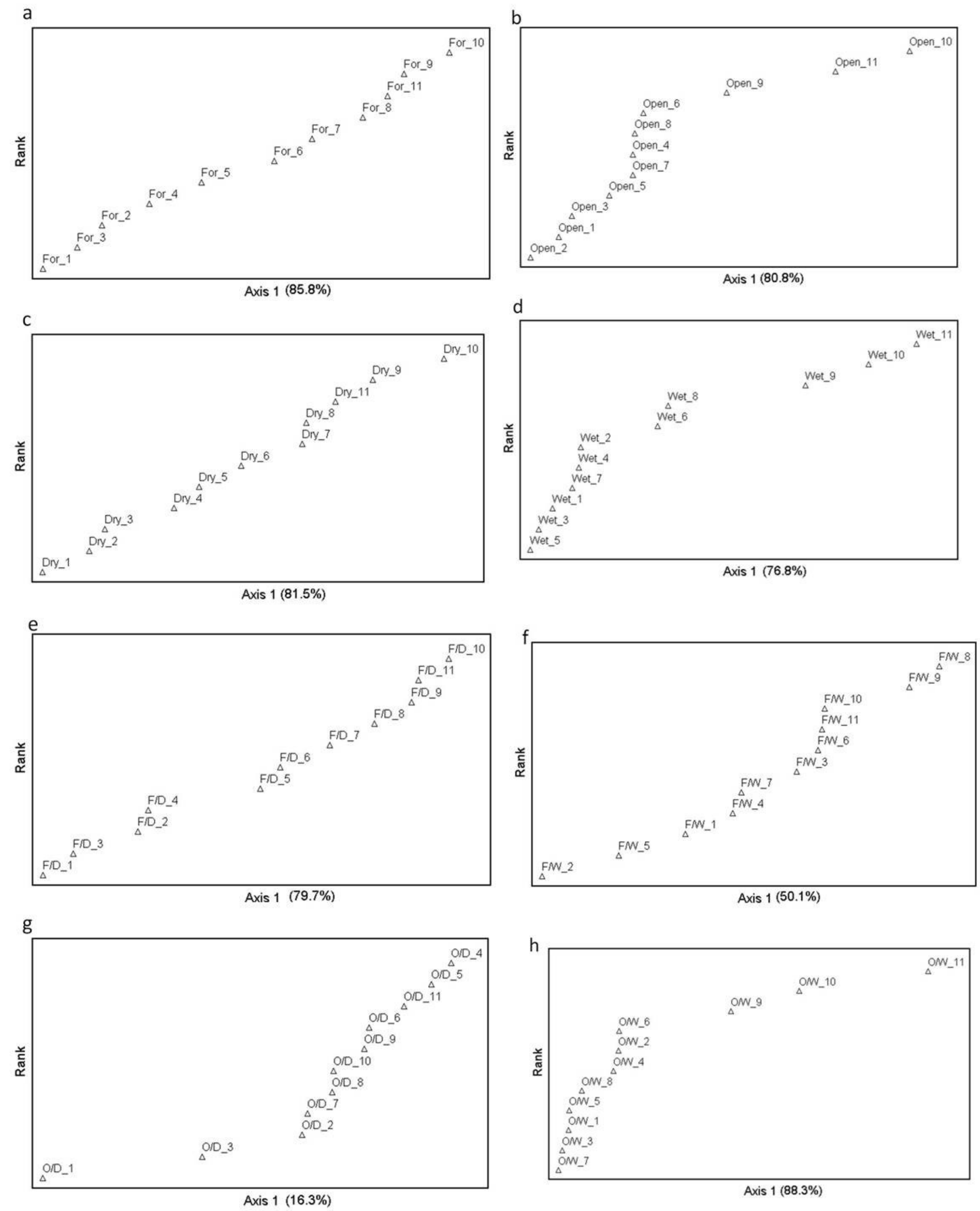

Figure 3.4: Polar ordinations of life form community level analysis within a factor. Triangles denote quadrats, numbering indicates their spatial position along the line transect. The number 1 denotes the closest position to the trail (i.e. the edge) and the number 11 denotes the further position from the trail (i.e. the interior). Panels: a-h shows the following factors: a) Forest, b) Open, c) Dry, d) Wet, e) Forest Dry, f) Forest Wet, g) Open Dry and h) Open Wet.

4 Chapter 4: Discussion

Three principal findings emerged from my study. First, the level of legal protection did not influence the amount of vehicular traffic trails received. Second, on-trail or direct impacts of ORV/ATV trails were greater on wet forested, heath and bogs sites as compared to dry forested sites. Third, off-trail or indirect impacts of ORV/ATV trails were greatest on bog sites. These findings are discussed below followed by suggestions for future research.

4.1 Concepts of Resistance and Resilience

In my study I adopted a community level view of resistance and resilience to facilitate comparisons of different communities' responses to recreational impacts. I used two metrics for assessing resistance and resilience of boreal communities to ORV trails; 1) the vulnerability of the substrate to erosive forces and 2) changes in plant life form composition from trail edge to interior. I defined plant resistance to recreational impacts as the ability to withstand being "injured or impaired," and plant resilience to recreational impacts as the ability to "survive or regenerate" (Kuss and Hall 1991; Yorks et al. 1997). I defined tolerance as the ability of a plant to be highly resistant or highly resilient or a combination of both strategies (Liddle 1997; Monz 2002). In my study resilience was not directly measured but was inferred by the amount of off-trail edge effect impacts on vegetation.

4.2 Intensity of Use

During the period of the traffic counter deployment there were no significant differences in traffic volume between and among trails within the various habitat types or between the trails in areas under different legal protection. While unexpected, given that ORV use is prohibited within the AWR, and unrestricted outside the reserve, these findings do serendipitously control the predictor variable of intensity of use (i.e., traffic volume). That said, I do recognize that intensity of trail use could have differed in the past, particularly when trails were first made; however focusing on the time frame of the field season allows for the further elucidation of on-trail and off-trail impacts across a larger sample of trails without the effect of variation in intensity of use. This is an intriguing finding since several studies have demonstrated an influence of intensity of recreational activity on on-trail (i.e., amount of erosion) impacts particularly at low levels of use (Weaver and Dale 1978; Iverson et al. 1981; Meadows et al. 2008). However due to the curvilinear-use impact relationship, several studies have found on-trail soil erosion to be a poor predictor of level of use (Dale and Weaver 1974; Cole 1992; Olive and Marion 2009) further illustrating the value of using traffic counters to understand the potential influence of intensity of use. Given that in this study intensity of use was statistically equivalent across all trails, other drivers such as cover type, moisture level, plant life form and environmental productivity could be more clearly delineated with respect to on and off trail impacts.

4.3 On-Trail Impacts: Resistance

Comparisons of levels of erosion among forested and open and dry and wet trails confirmed predictions that: 1a) erosion would be higher in open habitats; and 1b) erosion would be higher at wetter sites. Ruts were significantly deeper on open trails compared to forested trails. For areas that experience repeated trampling events site characteristics may be important determinants of community resistance and resilience (Cole 1995a). Direct comparison of this result to previous literature is somewhat difficult since I am unaware of comparable ORV/ATV recreation studies which compared soil erosion in a boreal forest and heath (barrens) and bog environments. A possible driver of habitat resistance is the relative soil moisture level. AIC_c analysis indicates that both cover type and moisture level were important drivers of on-trail rut formation, although moisture had a slightly higher weight. This agrees with the findings of previous researchers (Weaver and Dale 1978; Kuss 1986; Wilson and Seney 1994; Liddle 1997).

Although organic soil is highly vulnerable to erosion by trampling disturbance, it is most so when wet (Bryan 1977). The open sites have more waterlogged/water saturated soil/bog substrate. Rut depth was significantly deeper on wetter trails compared to drier trails. This result corresponds to findings of other recreation studies that demonstrated that wetter sites generally have higher levels of erosion than drier sites (Bryan 1977; Weaver and Dale 1978; Wilson and Seney 1994). Drier soil has greater capacity to bear a moving load (Marshall and Holmes 1979 as cited in Kuss 1986); conferring more resistance to vehicular impacts than wet soil. Dry forested trails had significantly less erosion compared to trails on other habitat types.

Interestingly there was no significant difference in erosion level between wet forested trails, dry open trails and wet open trails. This was an unexpected result since previous heavy tracked vehicle studies from the Canadian tundra indicate that wet sites are more susceptible to rutting than well-drained sites (Bellamy et al. 1971; Bliss and Wein 1972). In an ORV study of Australian saltmarshes, Kelleway (2005) found rut depth was greater in wetter areas. A possible explanation for my result may be the high inherent resistance of the system to erosion. Monz and colleagues (2013) recognize that the curvilinear relationship is not always applicable. Ecosystem responses to recreational use may be linear or exponential; highly resistant systems may exhibit a flat (zero slope) relationship with increasing use (Monz et al. 2013). The majority of Newfoundland soils are podzolic typical of boreal regions (Roberts 1983). The island was glaciated during the Wisconsin glaciation (7,000-10,000 years ago) and thus the majority of soil parent materials are morainal deposits – a mixture of glacial till, clay, silt, sand pebbles, and boulder-sized rocks (Roberts 1983). The soils in my study area are characterized by coarse podzolic soils with thin till and exposed bedrock (Roberts 1983). Thin till and exposed bedrock were particularly evident at heath sites. Thus these soils lack a thick organic layer and are relatively erosion resistant which may account for my findings.

4.4 Off-Trail Impacts: Resistance and Resilience

I do recognize that by not identifying graminoids, lichens and mosses down to species underestimates the biodiversity of these groups however broad patterns of community diversity are still taken into account by using life form classification for these groups. Vegetation community composition showed similar trends at both the species and life form levels of analysis. This could indicate that neither one nor several species were driving community response. Alternatively a single or several species or life form group could dominate across large areas. In either case generalities can be made at the life form level. This is advantageous since the vegetation community response can be considered at a broader scale.

Life form community gradients did not support hypotheses 2a and 2b. I predicted that 2a) off-trail vegetation impacts (changes in species composition) would be greater (appear further from the trail) in open habitats than in forested habitats and 2b) off-trail vegetation impacts (changes in species composition) would be greater (appear further from the trail) in wet sites than dry sites. In comparison of overall life form community homogeneity between habitat types (NMDS), forested trails showed sharper edges (more separation between edge and interior samples) than open trials. Interestingly, dry trails had sharper ecological edges than wet trails. Dry forested trails had the sharpest edge (strongest separation of samples) compared to the other habitat types. Investigation of the trail effect (the experimental gradient) within habitat types (Polar ordination), showed heath trails had the softest edge (lowest amount of species turnover from edge to interior) whereas bog trails had the sharpest edge (highest degree of species turnover from edge to interior) compared to all other habitat types.

The NMDS results concur with studies on the influence of canopy and sensitivity forest understory vegetation to trailside alteration (Cole 1978; Kuss 1986; Thurston and Reader 2001). Alteration in light conditions can provide the opportunity for sun loving exotics to invade (Pardenes and Jones 2000). In sunny open areas light is not a barrier to invasion by sun loving exotics (Pardenes and Jones 2000). That said, work done by Hansen and Clevenger (2005) has shown grasslands to be more susceptible to invasion by exotics than forests. The NMDS results provide information on the overall level of plant species and life form community homogeneity. The pattern between forest and open sites may also be a reflection of habitat productivity. Open sites or heaths and bogs are lower productivity environments compared to forest sites and thus could be expected to be more homogeneous in their overall community structure. Bogs sites are likely driving the wet and dry trail comparison given the small sample size of wet forested sites. When all four habitats were compared the high productivity dry forest showed the heterogeneity whereas low productivity bogs sites showed a high degree of clustering, indicating high homogeneity. Viewed in this way, a high degree of community homogeneity would be expected in a nutrient poor, acidic and waterlogged area.

A given habitat's ability to sustain recreational use depends upon its relative resistance, resilience or combination of those two strategies - tolerance (Monz 2002). Both plant morphological characteristics and habitat characteristics (i.e., relative productivity) play a role in determining an ecosystem's relative resistance, resilience and ultimately tolerance of human recreational use. Work by Cole (1995b) suggests that plant morphology is the most important determinant of community resistance and resilience for a single short-term trampling event. Bernhardt-Römermann and colleagues (2011) suggest that resistance is determined by morphological adaptations driven by background anthropogenic disturbance. On the other hand, resilience is determined by plant growth rate which is related to climatic conditions (Bernhardt-Römermann et al. 2011); which relates directly to the productivity of the system.

4.4.1 Habitat Productivity

My study demonstrated the sensitivity of bog vegetation to recreational impacts of ATV/ORV trails beyond the trodden ground. The vulnerability of vegetation communities in open low productivity habitats is recognized in the recreation literature (Willard and Marr 1970; Greller et al. 1974; Wilshire 1983; Ross 1991; Chapman and Pollard 1993). Recreation studies that have examined multiple vegetation or habitat types have focused on "on-trail" impacts (Cole 1995b; Arnesen 1999; Roovers et al. 2004). However few studies have examined recreational "off-trail" impacts (but see Naito 1969; Bayfield et al 1981; Hall and Kuss, 1989). The high community gradient on bog trails was expected since it corresponds to past studies, and indicates the high sensitivity of bog vegetation to vehicular disturbance (Ross 1991; Chapman and Pollard 1993). Unexpectedly the weakest community gradient was found on heath trails.

Lichen dominated heath has been found to be highly sensitive to on-trail recreation impacts (i.e., direct trampling) (Willard and Marr 1970; Greller et al. 1974; Liddle 1997; Arnesen 1999). Indirect edge effects have been demonstrated in a heath community in England, Angold (1997) detected an edge effect as far as 200 m away from a dual lane carriageway (i.e., four-lane divided highway). Angold (1997) found a decrease in lichen abundance near the road, including members of the genus *Cladonia*. I predicted dry heath would respond with a stronger gradient given lichen abundance has been documented to decrease near roads (Glenn et al. 1993) likely due to their sensitivity to air pollution (Ferry et al 1973 as cited in Angold 1997) and direct trampling (Willard and Marr 1970; Bell and Bliss 1973). Perhaps stronger off-trail effects were not found due to the low intensity of traffic. Angold (1997) found that edge effects in heath were strongly correlated to the amount of traffic on the nearby road. Results from my study demonstrate high resistance in heaths and low resistance in bogs to indirect recreational vehicular impacts. Given the fact that both habitats are low productivity environments community resistance may be driven by morphology.

4.4.2 Morphology

Life form groups that had strong association with the trail edge were graminoids, mosses and shrubs; conversely life form groups that had a weak association with the trail edge were lichens and ferns. Strong graminoid presence at trailside agrees with findings of other researchers (Liddle and Greig-Smith 1975b; Hall and Kuss 1989). Graminoids have a high tolerance to trampling disturbance; morphological characteristics such as tough stems/tissues convey trampling resistance (Cole 1995a; Yorks et al. 1997). Mosses are another group that show strong edge associations in a variety of habitat types. As a taxonomic group, bryophytes are relatively tolerant of trampling with the notable exception of members of the genus *Sphagnum* (Studlar 1983; Cole 1985; Liddle 1997). Morphological traits such as small size and compact growth form convey trampling resistance (Cole 1995a; Yorks et al. 1997). Studlar (1983) noted that given sufficient moisture, some species can exploit disturbed ground. Within MBE mosses (excluding *Sphagnum*) may make an important contribution in conferring overall community resistance to recreational vehicular trampling impacts particularly in wet areas.

The finding that shrubs had a strong association to the edge was interesting since direct trampling studies have indicated low resistance and resilience to trampling of shrubs (Yorks et al. 1997); in particular the chamaephytes (Cole 1995a). In this study vegetation sampling began directly beside the trail so vegetation was not directly trampled. Shrubs as a life form group were able to exploit the nearby disturbed but un-trampled area. This result is similar to that reported by Naito (1969) who found differences in alpine plant communities under a decreasing gradient of human trampling. Within the MBE forest floor moss species may contribute to overall community resistance (via morphology) and resilience by exploiting nearly disturbed ground. Moreover shrubs may be an important contributor to overall community resilience, particularly in forested habitat.

Across habitat types, ferns showed the weakest association with the edge. Ferns with upright brittle stems have been recognized as highly sensitive to direct trampling (Liddle 1997; Yorks et al. 1997; Hill and Pickering 2009). Low resistance to trampling has been reported for *Pteridium aquilinum* (bracken fern) (Littlemore and Barker 2001; Pickering and Hill 2009). Littlemore and Barker (2001) found high resilience in bracken fern in the following two years after trampling. My study provides evidence of continued sensitivity of ferns away from the trailside under semi-continuous summer ORV/ATV use. This is an interesting finding since although the bracken fern has been reported to be highly resilient, in my study system the timeframe of the disturbance regime appears to not allow for recovery. Finally, lichens as a life form group showed a weak association with the edge. Lichen sensitivity in alpine and heath environments has been well recognized (Willard and Marr 1970; Bell and Bliss 1973; Bayfield et al. 1981). Evidence of sensitivity beyond the trailside is comparable with results from Bayfield and colleagues (1981). Bayfield and colleagues (1981) investigated walking path impacts on lichen (*Cladonia* spp.) dominated heath in Scotland. They found detectable small amounts of damage on the heaviest use paths up to 50 metres from the path (Bayfield et al. 1981). However, on lightly used paths structural damage to lichen beyond 1 metre from the path was low (Bayfield et al 1981). This suggests that within the MBE community resilience of lichen dominated heath is low, whereas resistance is high.

4.5 Overall Community Resistance and Resilience

All habitat types were vulnerable to on-trail and off-trail impacts. There was no one "super tolerant" community. Broadly speaking forested communities were less resistant (strong edge) but more resilient in drier stands (less erosion) compared to heath and bog. Heath communities were more resistant (softer edge) but less resilient (more erosion). Bog communities were neither resistant nor resilient. Roovers and colleagues (2004) found that wet mesophilous forests were more resilient than heath to hikers. Heather species showed limited recovery following trampling in contrast to herbaceous forest species (Roovers et al. 2004). Gallet and Rozé (2001) found dry heath to be more resistant to pedestrian traffic than mesophilous heath which was more resilient. This provides support for the concept that higher productivity habitats are better able to tolerate trampling (Liddle 1975b).

Low resistance of bog vegetation to direct trampling in similar habitats has been shown by other researchers. In a study of Canadian bogs in Nova Scotia Ross (1991) found that only 40 passes of an ATV were needed to reduce vegetation cover to zero. In a study of a Tasmanian fen, Whinman and Chilcott (1999) found an increase in grass cover and decreases in herb cover six weeks following experimental trampling of 30 passes by hikers. Low resilience and ultimately limited recovery of vegetation is indicated by work done by Charman and Pollard (1993). In several English bogs, they found no, or limited recovery or succession towards a grassland community twenty years after use by military vehicles (Charman and Pollard 1993). Forbes (1992) examined floristic species richness and biomass in wet meadows in the Canadian arctic, twenty years following trampling by tracked vehicles. Overall species richness was reduced and the increase in tolerant graminoid biomass did not offset losses from sensitive non-herbaceous plants (Forbes (1992). Vehicular disturbance studies of arctic wetlands indicate that the resulting dominant species-poor tolerant graminoid community is self-perpetuating and natural (unassisted) recovery is limited (Forbes 1992; Forbes 1993; Forbes 1998; Forbes 1999).

The finding of relatively high resistance of heath to indirect ORV/ATV vehicle impacts is consistent with human foot traffic studies of heath (Bayfield et al 1981; Gallet and Rozé 2001; Roovers et al. 2004). Törn and colleagues (2006) found re-vegetation of Finnish subalpine heath following short-term light use (25-150 passes) by hikers. In contrast Bayfield (1979) found that montane heath communities recovered slowly from human trampling (40-240 passes) but recovery of most communities was nearly complete after 8 years. In a review of experimental human (pedestrian/hiking) trampling Hill and Pickering (2009) compared 65 studies of various vegetation types. Heath along with herb-fields was ranked as the lowest resilient habitat type (Hill and Pickering 2009); note that bogs were not one of the habitats assessed in their review due to the lack of studies. Hill and Pickering (2009) rank forest understory as having higher resilience than heaths but below communities such as grasslands.

My study showed the low resistance of boreal forest understory to indirect recreational vehicle impacts. Vulnerability of forest understory vegetation to direct recreation impacts is well recognized (Dale and Weaver 1974; Kuss 1986; Cole 1987; Cole 1995a; Cole 1995b; Thurston and Reader 2001). The majority of the recreation research on forests has focused on temperate or montane forests (Liddle 1997; Yorks et al. 1997; Hill and Pickering 2009). Roovers and colleagues (2005) illustrated evidence of temperature deciduous forest resilience to continuous hiking use. They found no difference in the distribution of plant life forms on the trampling path centre and control vegetation 10 metres from the path, six years after path closure (Roovers et al. 2005). Coniferous (boreal) may be more resistant than temperate forest due to the high amount of unincorporated organic litter (Legg 1973 as cited in Kuss 1986). In the experimental removal of ground, understory and humus layer in a pristine boreal spruce forest in Finland, Hautala and colleagues (2008) found that recovery was dependent upon the type of disturbance. If the humus layer is removed recovery is much slower than removal of the ground cover or understory layers (Hautala et al. 2008). However removal of the humus layer can leave lasting scars on the forest floor (Hautala et al. 2008). Interestingly, in all treatments (removal of ground cover, understory and humus) understory (vascular plants) completely recovered after 4 years (Hautala et al. 2008). Ground cover (bryophytes and lichens) did not recover to control levels at any treatment level (Hautala et al 2008). This is similar to my findings of the high resilience of shrubs and low resilience of lichens, as indicated by edge association. Implications from this work for my study indicate that although ORV/ATV traffic leaves long lasting scars on the forest floor, boreal forests have relatively high resilience compared to heaths and bogs.

4.6 Considerations of Spatial Scale and Trail Proliferation

Key to this study was the consideration of scale albeit at the localized level. Cole (2004) recognizes that spatial scaling of recreation impacts is essential to assess impact levels themselves and to create subsequent management strategies and argues that impacts may be greater at a local scale (i.e., the individual species level) and much smaller at greater scales (i.e., the landscape level). If this were the case, impacts at smaller extents would not be relevant to the wider population across the landscape (Cole 2004). Thus impacts would be limited to the local scale and overall landscape integrity would not be threatened (Cole 2004) assuming impacts were not permitted to proliferate. However, it is recognized that proliferation of impacts is often widespread. In regard to recreational trails specifically, impacts proliferate in several ways: 1) new trail creation, 2) widening or "braiding" of existing trails and 3) little or no recovery or deterioration of abandoned/closed trails over time (Cole 2004; Ouren et al. 2007).

New trail creation (particularly into areas lacking trails) is highly problematic since this permits access into previously inaccessible areas and could lead to ever increasing human impact (Buckley 2004; Cole 2004). Furthermore trails can be created through light traffic (Weaver and Dale 1978) but have substantial ecological impact given the curvilinear-use impact relationship (Liddle 1997; Cole 2004; Quinn and Chernoff 2010).

The widening of trails (through use) increases the impacts both on-trail and off-trail (Leung and Marion 1996, 2000; Cole, 2004; Ouren et al. 2007). At high intensities of use, open habitats such as meadows have wider trails than more closed habitats such as forests (Dale and Weaver 1974). Finally lack of recovery or deterioration of trails over time is also an important mode for proliferation of impacts. Impacts may be magnified through time, for example ever increasing natural erosion on existing ruts or changes in plant species composition in subsequent growing seasons (i.e., Bayfield 1979; Charman and Pollar, 1993; Jorgenson et al. 2010).

4.7 Conclusion

This study provides valuable information about different habitat responses under continuous recreational use. This is important for management decisions since recreational vehicle use is unlikely to stop within the wider MBE. This study's use of two distinct spatial scales (on and off-trail) attempted a broader consideration of spatial scale that is currently lacking in many recreational studies.

This study highlights the vulnerability of several boreal habitats to direct and indirect ATV/ORV trail impacts with particular consideration of habitat type, intensity of use, plant life form and primary productivity as key predictors. Plant life form morphology is an important determinant of community resistance; whereas habitat primary productivity is an important determinant of community resilience to recreational vehicle impacts. As a whole the MBE is relatively resistant to recreational vehicle erosion due to the stoniness of the soils and the lack of a thick organic layer.

4.8 Future Research

Monz and colleagues (2010) state that the ecological impacts of motorized recreation are greater than the impacts of non-motorized recreation; due to the vehicle's ability to travel long distances (spread impact) and torque applied to the ground. It is these impacts dispersed over large distances that make ORV impacts challenging to study. Previous ORV studies and vegetation trampling studies in general have largely focused on the trail surface or the area immediately adjacent to it. This study incorporates a gradient design to examine impacts into the interior of various habitat types and within a protected area and non-protected area (intended as a surrogate for traffic intensity). This gradient of distance feature could be incorporated into the design of more classic trampling studies; not only ORV studies. The rigor of experimental trampling studies at the local scale combined with data at the habitat patch scale in a number of different habitat types would allow for inferences about human recreational trails at a wider spatial extent. Examining impacts at multiple scales would aid in assessments of "impact creep." Impact creep is the gradual cumulative increase in impacts associated with increasing use and proliferation (Pickering and Hill 2007). Recent work by Arp and Simmons (2012) has combined time-series aerial photography with current ground series to assess the proliferation of ORV impacts. Given the applied nature of recreational ecological research, funding limitations are an obstacle to the advancement of this field (Monz et al. 2010). Extending low-cost landscape predictive models combined with field studies can contribute cost-effective innovation to the recreation ecology field.

5 Chapter 5: Management Implications

Plant communities whose tolerance to recreational vehicle impacts is via high resistance and low resilience could withstand periods of intense use, but once the impact threshold has been reached (i.e., damage has occurred) they will require periods of recovery with no recreational use (Cole 1995a; Gallet and Rozé 2001). Communities whose tolerance to recreational vehicle impacts is via low resistance and high resilience could withstand brief periods of intense use at regular intervals (Cole 1995a; Gallet and Rozé 2001). As long as the activity remained discontinuous the community would have an opportunity to recover. If one is managing for a high resistance and low resilience system, then regulating the number of recreational users is important. In this type of system if use is kept low (below the impact threshold) then the system could sustain continual use (i.e., throughout the year). If one is managing for a low resistance and high resilience system the impact threshold is likely to be exceeded even at low levels of use. Here regulating *when* recreationists use the system is important (i.e., permitting limited seasonal use).

Within the MBE management of ORV/ATV use may be adjusted for different habitat types. Management recommendations are summarized in Box 1. Firstly, wet or waterlogged areas should be avoided since my work showed either greater resistance (dry forest) or greater resilience (heaths) in drier areas to ATV impacts. Secondly, my work showed bogs have low resilience and resistance to ORV/ATV impacts, since they are damaged by even low amounts of use, the 5 passes by hunters permitted under the

Motorized Snow Vehicles and All-Terrain Vehicles Act may cause lasting damage. Any use of bogs by recreational vehicles should be discouraged. Thirdly, my work showed the resistance of heaths to indirect ATV impacts. Management efforts should focus on keeping ATV use below the impact threshold given the low resilience of the habitat. Properly managed heaths could support low and relatively continuous ATV use. However due to the openness of the habitat there is greater risk for trail proliferation which magnifies impacts. Fourthly, my work showed that dry boreal forest have low resistance to ATV impacts but may be more resilient due to the higher productivity of the habitat. Management of forests should focus on season of use; for example avoiding use of forest trails during snowmelt when soils are water saturated. Forests are also more closed environments therefore risk of trail proliferation may be less compared to more open heath. In summary heath and forest habitats can tolerate low levels of ATV use but their mechanisms of tolerance differ. Bogs cannot tolerate ATV use.

Box 1: Summary of Management Recommendations

Recommendations to Managers

- Avoid the proliferation of new trails in all habitat types mentioned in this study. New trails impact previous unaffected or minimally impacted areas. Encourage the use of clearly pre-existing trails, i.e., those already denuded of vegetation or where ruts are clearly visible.
- Encourage ORV/ATV riders to drive in dry areas since wet or waterlogged soil is more greatly impacted given equal use than dry soil.
- Avoid driving in bogs since they are a highly sensitive habitat, lacking in ability to withstand and recover from even light recreational use.
- Promote the use of existing dry forest trails or heath (barrens) trails. Dry forested communities are vulnerable to ATV impacts but can recover better following periods of use. Heath (barrens) may tolerate higher levels of traffic than forests before being indirectly damaged but once damaged will be unlikely to recover fully before the next cycle of use. Moreover as intensity of use increases open habitats such as heaths are more susceptible to proliferation of impacts (i.e., through trail widening and multiple braiding of trails). Management of heaths should take this into consideration.

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A. Appendix A: Summary of Regulations Governing the Use of Off-Road Vehicles on the Island of Newfoundland

Table A 1: Summary table of regulations governing the use of off-road vehicles on the island of Newfoundland.

Act/Regulation	Direct Quote from Statue	Comments
Motorized Snow Vehicles and All-Terrain Vehicles Act	Under the Act a vehicle is defined as “all motorized vehicles designed and constructed for travel on or immediately over land, water, snow, ice, marsh, swampland, and other natural terrain, including four-wheel drive or low-pressure-tire powered vehicles, low-pressure-tire motorcycles and related two-wheel vehicles, snowmobiles, amphibious machines, ground effect or air-cushioned vehicles, but does not include a motor vehicle” (Section 2(j), 2005).	Definition of a vehicle
	Under the Act an all-terrain vehicle is defined as “a wheeled or tracked motorized vehicle, excluding a 2 wheeled vehicle, designed or adapted for off-road use” (Section 2(a), 2005).	Definition of an all-terrain vehicle
Motorized Snow Vehicles and All-Terrain Vehicles Regulations	Under the Regulations “a person shall not use or operate an all-terrain vehicle outside an approved area” (Section 5.1, 2005). Under the Regulations approved areas included: “forested lands underlain by mineral soil… a trail constructed under licence issued under the *Lands Act*, beaches unless otherwise prohibited by the minister, abandoned railway corridors, highways abandoned… forest access roads…roads constructed under licence issued under the *Lands Act,* and any other road constructed for the purpose of providing vehicular access to resources…lands when snow-covered and frozen below the ground surface…” (Section	Legal definition of approved areas

Act/Regulation	Direct Quote from Statue	Comments
	2(c) (i), (ii), (iii), (iv), (vi), (vii), 2005).	
Motorized Snow Vehicles and All-Terrain Vehicles Regulations	The exceptions for hunters operating an ATV in unapproved areas are as follows: "…a person who holds a big game licence and, as permitted by the licence, has killed a moose, caribou or bear may use or operate an all-terrain vehicle outside an approved area for the purpose of transporting the animal from the place where it was killed... a person other than the licence holder may use or operate an all-terrain vehicle for the purpose of transporting an animal from the place where it was killed, but the licence holder shall remain in the immediate area… a person shall not use or operate an all-terrain vehicle under subsection (1) where an approved area may reasonably be used for the purpose... a person shall not use or operate an all-terrain vehicle more than 5 times to and from the place where the animal was killed and, when travelling from the place where the animal was killed, a portion of the animal shall be on the all-terrain vehicle or on a trailer being towed by the all-terrain vehicle… where more than one all-terrain vehicle is used or operated in relation to the transporting of a single animal under subsection (1) the total number of trips for all the all-terrain vehicles shall not exceed 5" (*Motorized Snow Vehicles and All-Terrain Vehicles Regulations*, Sections 5.1(1), 5.1(2), 5.1(3), 5.1(5), 5.1(6), 1999).	Statue defines exceptions of ATV use in unapproved areas for hunters
Provincial Parks Act	The *Provincial Parks Act* regulates the use of ORVs within provincial parks in the province of Newfoundland and Labrador. Under the *Provincial Parks Regulations under the Provincial Parks Act* (O.C. 97-510) an off-road vehicle is defined as	Regulation of ORV within provincial parks in the province of Newfoundland and Labrador

Act/Regulation	Direct Quote from Statue	Comments
	similarly to the definition in the *Motorized Snow Vehicles and All-Terrain Vehicles Act*. Within provincial parks ORV use is conditional (see below).	
Provincial Parks Regulations	ORVs may not be operated within provincial parks except if the individual is a park employee performing their duties, has a permit from the minister, or is within a designated use area of the park (*Provincial Parks Regulations* Section 9.1 (a), 9.1(c), 9.1(d), 2009).	Conditions of ORV operation within provincial parks on the island of Newfoundland
Provincial Parks Regulations Federal Parks Regulations	The major exception to this is the Newfoundland and Labrador T'Railway Provincial Park where the use of off-road vehicles is permitted (*Provincial Parks Regulations* Section 11.1, 2013). Snowmobile use is also permitted in designated areas in the Main River Waterway Provincial Park and Gros Morne National Park (*Provincial Parks Regulations* Section 9.3, 2009 and *Canada National Parks Act* Section 24.3, 2000).	Exceptions of ORV use within provincial and federal parks on the island of Newfoundland
Wilderness and Ecological Reserves Act	Under the *Wilderness and Ecological Reserves Act* (WER Act) Wilderness Reserves are defined as "…areas of the province that are subject to no or little human activity (WER Act, Section 4, 1980). The WER Act states four functions of wilderness reserves, 1) "to provide for the continued existence of those areas as large wilderness areas to which people may come and in which they may hunt, fish, travel and otherwise experience and appreciate a natural environment;" 2) "to allow within those areas undisturbed interactions of living things and their environment;" 3) "to preserve those large areas that may be necessary for the continued survival of a particular species; or" 4) "to protect areas with primitive or extraordinary	Text describing the legal functions of a wilderness reserves within the province of Newfoundland and Labrador

Act/Regulation	Direct Quote from Statue	Comments
	characteristics" (WER, Section 4 (a),(b),(c),(d), 1980).	
Wilderness and Ecological Reserves Act	For example prohibited activities within a wilderness reserve include: construction of a structure or reconstruction of a structure or construction of a "…road, path or track", or engaging in the "cutting or logging of trees, agriculture, mining…" WER Act, Section 24.1 a(i)(ii), 1997). Also prohibited is the use of motorized vehicles or equipment and landing aircraft (WER Act Section 24.2 (a)(b), 1997).	Examples of prohibited activities within wilderness reserves within the province of Newfoundland and Labrador
Wilderness Reserve Regulations	Legal recreational activities within the reserve include: hiking, canoeing, boating on designated lakes, camping, wildlife and bird viewing, hunting and angling (*Wilderness Reserve Regulations*, Sections 5, 10, 11, 16.4, 1997; Department of Environment and Conservation, 2013).	Examples of recreational activities permitted in wilderness reserves in the province of Newfoundland and Labrador
Wilderness Reserve Regulations	Newfoundland Power and Newfoundland and Labrador Hydro companies may receive permission to use all-terrain vehicles and snowmobiles to access existing facilities but routes are defined by the managing agency of the reserve (*Wilderness Reserve Regulations* Section 25, 1997).	Exception to ATV use within wilderness reserves in the province of Newfoundland and Labrador

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B. Appendix B: Species Lists, Species Codes and Life Form Categories

Below are species lists encountered in this study. Plant field guides included: Ryan (1978), Johnson and colleagues (1995) and Scott and Black (2008).

Table B 1: Species names and codes that were classified under the life form of trees

Common Name	Scientific Name	Code	Family
Balsam Fir	*Abies balsamea* (L.) Mill.	abi bal	Pinaceae
Larch	*Larix laricina* (DuRoi) K. Koch	lar lar	Pinaceae
Black Spruce	*Picea mariana* (Mill.) B.S,P.	pic mar	Pinaceae

Table B 2: Species names and codes that were classified under the life form of shrubs (woody) plants

Common Name	Scientific Name	Code	Family
Speckled Alder	*Alnus rugosa* (DuRoi) Spreng.	aln rug	Corylaceae
Serviceberry	*Amelanchier* spp.	ame spp	Rosaceae
Bog rosemary	*Andromeda glaucophylla* Link.	and gla	Ericaceae
Chokeberries	*Aronia* spp.	aro spp	Rosaceae
Newfoundland Dwarf Birch	*Betula michauxii* Spach	bet mic	Corylaceae
Leatherleaf	*Chamaedaphne calyculata* (L.) Moench	cha cal	Ericaceae
Black Crowberry	*Empetrum nigrum* L.	emp nig	Empetraceae
Common Strawberry	*Fragaria virginiana* Duchesne	fra vir	Rosaceae
Snowberry	*Gaultheria hispidula* (L.) Muhl.	gau his	Ericaceae
Common Juniper	*Juniperus communis* L.	jun com	Pinaceae
Sheep Laurel	*Kalmia augustifolia* L.	kal aug	Ericaceae
Bog Laurel	*Kalmia polifolia* Wang.	kal pol	Ericaceae
Northern Honeysuckle	*Lonicera villosa* (Michx.) R.&S.	lon vil	Caprifoliaceae
Sweetgale	*Myrica gale* L.	myr gal	Myricaceae
Mountain Holly	*Ilex mucronata* (L.) M.Powell, Savol., & S.Andrews	nem muc†	Aquifoliaceae
Shrubby Cinquefoil	*Potentilla fruticosa* L.	pot fru	Rosaceae

Common Name	Scientific Name	Code	Family
Three-Toothed Cinquefoil	*Potentilla tridentata* Ait.	pot tri	Rosaceae
Pin Cherry	*Prunus pensylvanica* L. f.	pru pen	Rosaceae
Choke Cherry	*Prunus virginiana* L.	pru vir	Rosaceae
Rhodora	*Rhododendron canadense* (L.) Torr.	rho can	Ericaceae
Labrador Tea	*Rhododendron groenlandicum* Oeder	led gro†	Ericaceae
Northern Dwarf Raspberry	*Rubus arcticus* L. subsp. acaulis (Michx.) Foeke	rub acr	Rosaceae
Raspberry	*Rubus idaeus* L.	rub ide	Rosaceae
Dwarf Raspberry	*Rubus pubescens* Raf.	rub pub	Rosaceae
Willows	*Salix* spp.	sal spp	Salicaceae
Meadowsweet	*Spiraea latifolia* (Ait.) Borkh.	spi lat	Rosaceae
Canadian Yew	*Taxus canadensis* Marsh.	tax can	Taxaceae
Blueberry	*Vaccinium angustifolium* Ait.	vac ang	Ericaceae
Marshberry/Small Bog Cranberry	*Vaccinium oxycocus* L.	vac oxy	Ericaceae
Bilberry (tundra)	*Vaccinium uliginosum* L.	vac uli	Ericaceae
Lingonberry/Bog Cranberry/Partridgeberry	*Vaccinium vitis-idaea* L.	vac vit	Ericaceae
Wild Raisin	*Viburnum nudum* L. var. *cassinoides* (L.) Torr. & A. Gray	vib cas	Caprifoliaceae

† nem muc based on former classification *Nemopanthus mucronata* L. (Trel.)

† led gro based on former classification *Ledum groenlandicum* Oeder

Table B 3: Species names and codes that were classified under the life form of herbaceous (herbs) plants

Common Name	Scientific Name	Code	Family
Pearly Everlasting	*Anaphalis margaritacea* (L.) Clarke	ana mar	Compositae
Dragon's Mouth	*Arethusa bulbosa* L.	are bul	Orchidaceae
Bluebead/ Corm lily	*Clintonia borealis* (Aiton.) Raf.	cli bor	Liliaceae
Goldenthread	*Coptis groenlandica* (Oeder) Fern.	cop gro	Ranunculaceae
Canadian Bunchberry (crackerberry)	*Cornus canadensis* L.	cor can	Cornaceae
Round-Leaved Sundew	*Drosera rotundifolia* L.	dro rot	Droseraceae
Fireweed	*Chamerion angustifolium* L.	epi ang†	Onagranceae
Blue Flag Iris	*Iris versicolor* L.	iri ver	Iridaceae
Fall dandelion	*Leontodon autumnalis* L.	leo aut	Compositae

Common Name	Scientific Name	Code	Family
Twinflower	*Linnaea borealis* L.	lin bor	Caprifoliaceae
Canadian Mayflower	*Maianthemum canadense* Desf.	mai can	Liliaceae
Cloudberry/Bake-Apple	*Rubus chamaemorus* L.	rub cha	Rosaceae
Pitcher Plant	*Sarracenia purpurea* L.	sar pur	Sarraceniaceae
False Solmon's Seal	*Smilacina trifolia* syn. *Maianthemum trifolium* (L.) Sloboda {racemosa]	smi tri	Liliaceae
Rough-Stemmed Goldenrod	*Solidago rugosa* Ait.	sol rug	Compositae
Bog Goldenrod	*Solidago uliginosa* Nutt.	sol uli	Compositae
Common Dandelion	*Taraxacum officinale* Weber	tar off	Compositae
Tall Meadow Rue	*Thalictrum pubescen*s Pursh	tha pub	Ranunaculaceae
Starflower	*Trientalis borealis* Raf.	tri bor	Primulaceae

† epi ang based on former classification *Epilobium angustifolium* L.

Table B 4: Species names and codes that were classified under the life form of ferns

Common Name	Scientific Name	Code	Family
Bracken Fern	*Pteridium aquilinum* (L.) Kuhn	pte aqu	Dennstaedtiaceae
Marsh Fern	*Thelypteris palustris* (Schott)	the pal	Thelypteridaceae

Table B 5: Other life form categories and codes not identified to species or genus

Life Form	Code	Comments
Mosses	moss	This category also includes other bryophytes such as liverworts. Mosses were however most prominently sampled. Commonly sampled were members of the genus *Dicranum*. *Pleurozium schreberi* (Brid.) Mitt was also common along with other unidentified members of the family Hylocomiaceae.
Lichens	lichen	Prominently sampled were members of the genus *Cladonia*.
Graminoids	gramin	Well represented were members of the Cyperaceae (sedges) including *Carex* spp. and members of Poaceae (grasses)

B.1 References

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C. Appendix C: Supplemental Off-Highway Vehicle Counter Results

I wished to further investigate the destination effect, that is whether trails ending in lakes were preferentially travelled (have higher traffic volume) than trails that did not end in lakes. I deployed 4 G3 OHV counters manufactured by TRAFx Research Ltd. I placed 2 counters on trails that ended in lakes and 2 counters that ended in other destinations (see Table 2.1 for trail details). Counters were deployed from May 3 2011-June 1 2011. I chose this time period to coincide with the beginning of the angling season (Newfoundland and Labrador Angler's Guide 2011-2012). One of the counters placed at a non-lake destination trail failed to start. Due to the small sample size of traffic counts obtained in spring 2011 rigorous statistical analysis is not possible. However this data does provide anecdotal evidence that trails ending in lakes are travelled more frequently than trails that ends in other destinations. Trails that ended in lakes had a mean counts per day of 0.30 whereas no traffic was logged for trails ending in other destinations (Table E1). Traffic volume measured in spring 2011 was much lighter (mean 0.30) compared to the volume measured in summer 2010 (mean 3.18). This may indicate low continuous use throughout the summer season rather than a peak time at the start of the angling season.

Table C 1: Traffic Counter Volume from Summer 2010 and Spring 2011, counts are standardized as counts per day

Counts per day Summer 2010		
Maximum		13.6
Minimum		0.2
Mean		3.18
Standard Deviation		3.62
Counts per day Spring 2011		
Lake Trails		
Maximum		0.32
Minimum		0.29
Mean		0.30
Standard Deviation		0.03
Other Trails	No Data	0

C.1 References

Newfoundland and Labrador Angler's Guide 2011-2012. Department of Fisheries and Oceans. www.nfl.dfo.mpo.gc.ca. Last Accessed on June 8, 2012. Pp. 40.