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# Recreational Trail Impacts and their Spatial Influence on Species Diversity and Composition

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# RECREATIONAL TRAIL IMPACTS AND THEIR SPATIAL INFLUENCE ON SPECIES DIVERSITY AND COMPOSITION

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# Abstract

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This thesis quantifies the differences observed in floral communities exposed to varying degrees of long-term recreational trail use. The study was undertaken in a temperate deciduous forest located in Uxbridge, ON, Canada, which permits hiking, mountain biking and equestrian trail users. Vegetation exposed to trail impacts was sampled using transects which extended from the trail edge to 25m into the forest interior. The results demonstrated that trail-influenced environments experienced significant shifts in composition and reductions in species richness at distances beyond the influence of an edge effect. It was also established that types of recreational trail use do not disproportionately cause greater disturbance or result in greater exotic and invasive species coverage. Multiple regression analysis revealed that when choosing new trail routes, managers can mitigate changes to species composition by selecting areas with steep side-slopes and by avoiding areas with a south facing aspect.

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# List of Abbreviations

EDH - East Duffins Headwaters

ELC - Ecological Land Classification

GIS - Geographic Information System

IDH - Intermediate Disturbance Hypothesis

SD - Standard Deviation

TRCA - Toronto and Region Conservation Authority

# Chapter 1: Introduction to the Research Topic

In North America, the use of wildlands for recreation purposes has increased considerably since the 1960's. Consequently, increased degradation of these natural areas has also occurred. Already under pressure from other external influences such as increased habitat loss, pollution and climate change, the justification for allowing recreational activity on these lands has become questionable. Especially for conservation authorities and other agencies working to preserve biodiversity, it has become unclear to what extent recreational use compromises the underlying objectives that conservation lands first sought to achieve. This has left managers to determine if the disturbances previously considered to be acceptable on their lands should continue to be tolerated, and at what level.

In response to the many unknowns regarding recreational impacts, the field of recreation ecology has developed over the past several decades. The intention of the field is to provide natural area managers with the knowledge needed to predict and prevent recreational impacts in hopes that environmental conditions may be preserved (Liddle, 1975 a,b). Work began with Bates (1935) who was first to identify vegetation gradients adjacent to trails. Since then, several hundred studies have been conducted which examine recreational impacts on natural systems (Liddle, 1997; Cole, 2004).

Although recreational impacts can take many forms, researchers have primarily focused on the use of trails and trail systems because of their popular use and potential for significant impact (Washburne and Cole, 1983). The use of trails can cause a variety of impacts to the environment including wildlife, soil, water, air and vegetation. To date, impacts of recreational trails on vegetation have largely focused on direct and immediate disturbances. However, such studies do not account for changes in species composition which may occur after repeated exposure and little is known regarding how species diversity will change over the long term. The overall objective of this study was thus to compare the effects of different trail use types on floral composition and to do so on a much larger scale than previously studied. Additionally, this study aimed to determine if specific trail characteristics contribute to the combined effects on biodiversity and changes in species composition.

# Chapter 2: Background and Literature Review

## 2.1 Trail Impacts to Vegetation

The relationship between trails and their influence on vegetation is complex and multifaceted. These effects rarely occur in isolation and can even exacerbate or compensate for other changes. To simplify this complexity, descriptive information regarding impacts to vegetation from trails is best understood by discussing the nature of impacts, factors that influence the magnitude of impact, temporal impacts, spatial aspects and the relationship between trail impacts and biodiversity. This section summarizes the known impacts to vegetation with specific reference to trail-related disturbances.

## 2.2 The Nature of Impacts

Recreational impacts can be both direct and indirect in nature. Direct effects are those which occur immediately after direct contact is made between the trail user and the environment (e.g. loss of vegetation due to trampling). Models of these effects often contain positive feedback loops which can lead to indirect changes to the environment. On the other hand, indirect effects occur as a consequence of direct effects after repeated exposure (e.g. change in species composition as a result of increased competition from introduced species). Indirect effects characteristically occur in the long term and are often considered to be of greater significance because they can continue to occur even after recreational use has stopped.

The most immediate and pronounced effects of trail use are the physical alteration of floral vegetation caused by trampling. Trampling from users causes shearing, crushing and tearing of vegetation which can ultimately result in a reduction in plant height, stem length, leaf area, the number of flower heads per plant, the percentage of plants that flower as well as seed production (Holmes, 1979 a,b; Liddle, 1997). Abrasion to a plant via direct contact may lead to lesions which are exacerbated by wind and sand near the trail (Skidmore, 1966; Mackerron, 1976). Regardless of the alteration, the plant will subsequently need to produce wound tissue and new organs which requires the use of energy. If photosynthetic areas of the plant are significantly reduced as a result of physical alteration, food reserves which may have been required for future growth will also have to be sacrificed (Kendal, 1982). Typically, carbohydrate storage organs will gradually decrease and this is accompanied by a reduction in biomass (Weinmann, 1952). Responses can differ between plant types but in general, the plant will become more susceptible to adverse conditions as carbohydrates are reduced. Even if food reserves are not depleted, ethylene production induced by trauma will result in reduced growth (Cooper, 1972; Hiraki and Ota, 1975). Ironically, touching or bruising may also strengthen stems to mechanical stress which is advantageous to the plant growing in trampled areas, overall increasing its resistance and survival (Liddle and Moore, 1974).

In addition to trampling, trail users may also have less pronounced effects on floral vegetation. For example, shaking of a plant due to direct physical contact by the user will cause the stomata to open. The opening of the stomata ultimately leads to a higher transpiration rate as well as reduced water retention and reduced growth (Akers and Mitchell, 1984; Grace *et al.*, 1982).

In a concentrated setting such as a well used trail, trampling often causes defoliation or the complete removal of vegetation from an area (Hammitt and Cole, 1987). This occurs either because the plant is ripped out of the ground, its regenerative tissues are destroyed or because the loss of leaves reduces its ability to photosynthesize (Kuss, 1986). Typically, areas experiencing high levels of direct trampling will have low plant coverage and will be comprised of short, stunted plants, if any at all. In addition, species richness will be reduced and composition will shift to more tolerant species (Hartley, 1999).

The next observable change resulting from direct trail use occurs at the soil level. Soil compaction is caused when pressure exerted from the trail user depresses soil particles and reduces the interstitial pore space (Manning, 1979). This has a number of influences on vegetation. For example, soil compaction, which reduces soil aeration and water infiltration, may cause plants to grow fewer lateral roots and root hairs because cytoplasmic streaming within root hairs is reduced (Alessa and Earnhart, 2000). This may make it difficult for plants to attain water and nutrients as well as limit root penetration causing the plant to be less stable. The germination, emergence and establishment of plants are also influenced by soil compaction (ibid). A lack of proper incubation and moisture makes germination difficult due to the reduction of areas available for establishment. Even if the plant is successful at germinating, the removal of vegetation cover will subject the seedling to heat stress and the likelihood of death will increase. Moreover, if the environment receives additional stress from a lack of moisture, the effects of soil compaction can be exacerbated (Hammitt and Cole, 1987). Overall, recreation areas that experience soil compaction can support vegetation growth, but it is often limited.

Trampling also causes a loss of organic litter which affects some plants. Above ground, loss of soil litter can cause a shift in species composition towards species that prefer to germinate on mineral soils. This may occur because species which normally germinate on organic surfaces may no longer be able to establish themselves (Cole, 2004). Below ground, microbial populations may also be severely affected.

Microbial populations play an essential role in ecosystem functioning and contribute to an ecosystem's health by metabolizing nutrients, producing phytohormones as well as contributing to soil food webs (Perry and Amaranthus, 1990; Turkington *et al*., 1988; Curl and Truelove, 1986). As a result of trail use, loss of organic matter and root exudates may reduce carbon substrate availability for microbes (Zabinski and Gannon, 1997). Additionally, compaction can cause soils to lose their water retention capabilities and reduce soil aeration which can also affect the composition of microbial communities (Schimel and Parton, 1986; Parton *et al*., 1987; Wardle, 1992). If microbial changes do occur, they may suppress plant growth and change vegetative composition by influencing a species ability to colonize an area (Bever, 1994; Ingham, 1994; Chanway *et al.*, 1991; Turkington *et al*., 1988; Tranquillini, 1979).

Another effect of vegetation removal by trampling is that mineral soils tend to be quickly exposed. Increased trampling then loosens the soil making it vulnerable to erosion processes which are expedited by sunlight, rain and topography (Wilson and Seney, 1994). Since erosion is more or less irreversible, it has been given much attention by the scientific community. Generally, erosion causes higher stress to plants because it reduces water retention and nutrient accumulation capacities of the soil. The level of stress is dependent on the severity of erosion and the type of vegetative community it is affecting (Liddle, 1997).

Unlike direct impacts, the severity of indirect effects of recreational use has received less attention. However, this area of recreation ecology is growing and analysts have identified the loss of biodiversity and the spread of invasive species to be the most important indirect impact (Newsome *et al.,* 2002; Buckley *et al.,* 2003). This is because invasive species continue to spread and influence native composition even after disturbance has ceased (Meyerson and Griscom, 1999). Invasive species are defined as species which invade areas where they previously did not exist and are those which are able to persist and out-compete native species without disturbance.

Invasive species are able to out-compete native species because of a number of traits that the former species typically have. These include but are not limited to, rapid growth and short life cycle, prolific flowering, high seed production, efficient method of seed dispersal, staggered germination, and resistance to physical impacts. These species can become established in recreational areas due to two direct processes, through the alteration of habitat which can favour their establishment (e.g. decreased cover, increased light) and/or through the physical introduction by the recreationalist. Once established, invasive species can have several influences on the natural environment including the alteration of nutrient levels in the soil, changes to hydrology, modification of recruitment levels by shading and changes in biodiversity (Csurches and Edwards, 1998; Williams and West, 2000; Gordon 1998; Elton, 1977; Meyerson and Griscom 1999).

### 2.3 Factors That Influence the Magnitude of Impact

Wherever recreational use is present along a trail, some degree of environmental impact is inevitable. However, unless the impact is occurring in an area that is highly fragile or of great conservation priority, it is typically undesirable for land managers to prohibit all recreational use. Thus, the challenge is to reduce impact to an acceptable limit by managing the factors that have the greatest influence on the magnitude of impact. These include the frequency of use, season of use, the environmental conditions and type of recreational use.

#### 2.3.1 Frequency of Use

The area of recreation ecology which has received the most attention is the relationship between frequency of use and the magnitude of impact. This relationship differs depending on the impact criteria, direct impacts such as trampling or soil compaction as well as indirect impacts to species composition or biodiversity. The frequency-impact relationship for direct impacts is said to be asymptotic (Figure 1A) (Cole, 2004). At first, high increases in impact occur with small increases in frequency. As use intensity increases, the rate of increase in impact decreases. Interestingly though, areas that are heavily used but receive varying amounts of frequency may have similar impact levels. The opposite has been observed in areas where use is light (Frissell and Duncan, 1965). This response likely occurs because of an initial sharp decline in vegetation coverage as sensitive species are eliminated by trampling followed by a slower attrition of more resilient individuals with increasing levels of use. Cole (1995) demonstrated that over a two year period, vegetation cover decreased curvilinearly with increased trampling intensity. This suggests that direct impacts on established trails likely experience little change despite fluctuations in the frequency of use.

Unlike direct impacts, vegetation and biodiversity are thought to exhibit a curvilinear (humped-back curve) relationship with increasing use or impact (Figure 1B). This assumption derived from Connell's Intermediate Disturbance Hypothesis (IDH) which suggests that biodiversity will be highest when disturbance is intermediate. Disturbance can be defined as 'events that disrupt ecosystem community or population structure and change the availability of resources or the physical environment' (Pickett and White, 1985). Connell (1978) theorized that competitive species would dominate at low levels of disturbance and would be replaced by disturbance-tolerant species at high levels. This is also accompanied by a reduction in species richness which occurs because there are typically fewer species that are resistant to impacts than original occupants. At intermediate levels of disturbance, however, large numbers of both communities can coexist creating higher diversity and a change in species composition. Moreover, disturbed sites are open to immigration and can be quickly filled by invasive species or by species that were previously extirpated from the area. The edges of trails are thought to exhibit these conditions since they are disturbed by trampling and are often exposed to sources of seeds and propagules which are transported by trail users (Liddle, 1997). As is further discussed below, a community's change in diversity is also dependent on the resistance and resilience of its individuals. Consequently, researchers such as Hall and Kuss (1989) have previously found trail-influenced environments to be more diverse than control/baseline conditions while others to be less diverse (Boucher *et al.*, 1991). Thus, designing management recommendations for impact mitigation requires thorough understanding of how specific vegetation types will respond to impact.

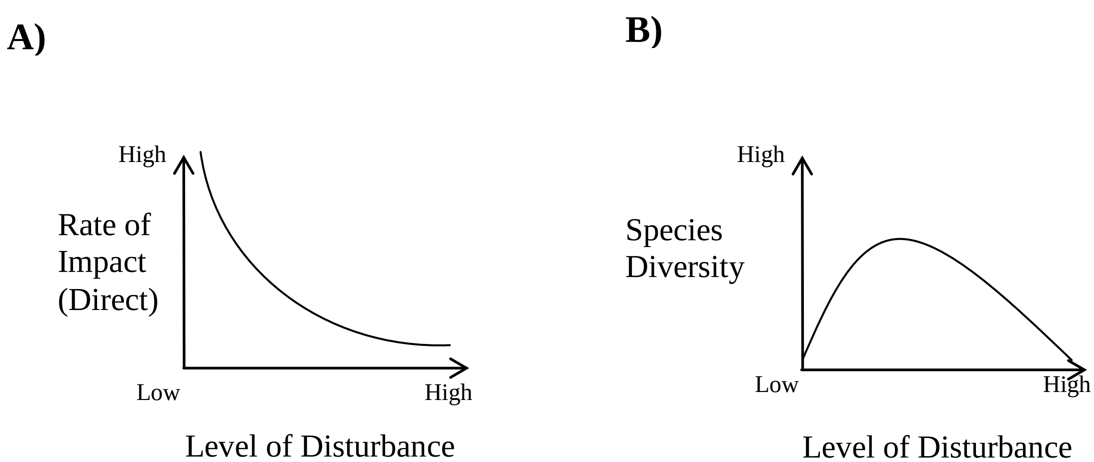


**Figure 1. Disturbance Relationships.** A) The relationship between disturbance and rate of direct impacts such as trampling and soil compaction is represented as an asymptote. B) The relationship between disturbance and species diversity is represented as a humped-back model of the Intermediate Disturbance Hypothesis (modified from Connell, 1978).

### 2.3.2 Season of Use

Although not typically studied, the season of use can also influence the type and magnitude of impact. During the spring, plants that are not fully established can be more susceptible to trampling and other disturbances than later in the growing season (Liddle and

Grieg-Smith, 1975). Researchers have also noted that during seasons of high precipitation, trails can become saturated with water and are more susceptible to disturbance by heavy users such as equestrians (Liddle, 1997). Furthermore, when trails are wet and muddy, hikers and mountain bikers are more likely to widen the trail to avoid puddles which consequently may cause increased vegetation loss and soil disturbance (Cole, 1987).

### 2.3.3 Environmental Conditions

The environmental characteristics of an area including the topography, soil and vegetation type can play a vital role in the magnitude of impacts. This area of recreation ecology has received much attention because it allows for the avoidance of locations prone to high impact and the development of preventive measures (Price, 1983). Avoiding or manipulating certain environmental features to mitigate potential impacts is particularly important in places where recreation and environmental conditions must be maintained or where management action is infrequent.

Characteristics of topography have been well documented and several studies have reported a strong positive relationship between trail slopes and soil loss (Leung and Marion, 1996; Cole and Bayfield, 1993). The cause of the increased erosion is primarily due to the higher velocity of runoff on steep slopes. However, soil disturbance caused by feet, tires or hooves is also likely to be higher on steep slopes due to slippage (Newsome *et al.,* 2004). The topography can also alter the microclimate of an area. Aspect refers to the direction to which a slope faces. In the northern hemisphere south-facing slopes are exposed to more sunlight and are typically dryer than north-facing slopes due to higher levels of evapotranspiration. Evapotranspiration causes north-facing soils to be moister, and although not always the case, it may also cause vegetative communities to be typically more productive and have greater vegetation coverage (Bennie *et* *al.*, 2006).

Trails in areas with high water tables can also increase the magnitude of impacts as a result of being constantly wet (Bayfield, 1973). If trails do not have proper drainage, eroded and muddy tread surfaces will likely occur. Soil texture can contribute to this problem since soils with fine and homogeneous textures have been found to hold more water (Bratton *et al.*, 1979). A wet trail tread often results in users seeking to avoid muddy areas and in effect causes increased trail width and vegetation loss.

The vegetation type as well as certain physical vegetation characteristics can influence the magnitude of impact. Studies have also shown that particular vegetation types can tolerate more impact than others, in some cases more than 30 times as much use (Cole, 1995). Vegetation durability is a product of both its resistance, the ability to tolerate use, and its resilience, the ability to recover from damage (Sun and Liddle, 1993a; Cole, 1993; Cole and Bayfield, 1993). Generally speaking, resistance decreases with erectness. Plants with large thin leaves and tall stems are particularly vulnerable to trampling compared to low shrubs and grasses (Liddle, 1997). However, shrubs are considered to be less resilient and if damaged, will take longer to recover (Cole, 1995). The most tolerant and resilient vegetative species are grasses which explains why impact magnitude can vary so widely between meadows and other areas such as forests which are dominated by forbs and woody species (Dale and Weaver, 1974). Trail use is likely most heavily influential in forests since tree saplings seldom survive high levels of trampling. In the event that existing over-story is reduced and saplings are consequently not able to regenerate and reproduce over-story conditions, a shift in species composition is likely to occur (Hammitt and Cole, 1987). The area can become devoid of trees and many of the species native to that area will likely be replaced by shade intolerant species, though this would be an extreme case (Brown *et al.*, 1977). Since every ecotype is unique, a detailed understanding of how floral communities exposed to trail use within particular ecotypes is needed.

### 2.3.4 Type of Use

The type of trail use also plays an important role in the type and magnitude of impact. Although many studies have been published on impacts of user groups, most of them have examined user groups individually (Gower, 2008; Davies and Newsome, 2009; Bright, 1986; Cole, 2004; Bayfield, 1971). Though these studies may point to sources of impacts, statistically valid comparisons cannot be made between them to compare user group severity of impact. This is because either their environments were not homogeneous or data was not collected in a way that would allow such comparisons to be made. Thus, within this section, only studies that compare user group impacts were examined.

Generally, the types of impacts to vegetation and soil were demonstrated to be similar between hiking, mountain biking and equestrian use. However, significant differences have been found in their severity in terms of floral composition and trail degradation (Torn *et al.*, 2009). For the most part, comparative studies have focused on impacts to soil such as compaction and erosion (Cole and Spildie, 1998; Wilson and Seney, 1994). These studies have found that per capita, horses and mountain bikes cause more impact than hikers (ibid). A study by Weaver and Dale (1978) concluded that equestrian trails were significantly deeper than hiking trails. In addition, the amount of erosion was also found to differ between the user groups. To date, only one study has compared the three groups potential to cause erosion (Wilson and Seney, 1994). This experimental study demonstrated that equestrians were the only group to cause significantly more sediment yield than control sites and that mountain bikes caused no more erosion than hikers. It should be noted, however, that this study had several methodological limitations. Most notably, several trails had significant differences in sediment yield, prior to the start of the experiment (Pickering *et al.*, 2010).

Comparative impact studies to vegetation are also uncommon in the literature. Researchers have typically reported on vegetation coverage, trail width, species richness and invasive species (Torn *et al.*, 2009). Similar to direct impacts, equestrians were generally found to have the most influence. Nagy and Scotter (1974) reported that horses destroyed eight times as much coverage and created more bare ground than hikers when trampling through undisturbed vegetation. In a rare study that included mountain biking, Thurston and Reader (2001) demonstrated that hiking and mountain biking did not influence vegetation coverage or expose mineral soils differently. However, methodological shortcomings may have caused this indifference since the study provided little opportunity for investigation of braking, accelerating or turning. The study potentially did not exert impacts that would normally be observed on established mountain biking trails (Pickering *et al.*, 2010). The study by Thurston and Reader (2001) is also the only known study to compare mountain bikers against other users by their influence over species richness. Similar to the user groups' influence on vegetation cover, it was reported that impacts to species richness did not differ between the groups. It is important to note that this was an experimental study through undisturbed vegetation with only 500 passes. Such an experiment would unlikely take into account indirect impacts and species interactions that may occur in the long term (Pickering *et al.*, 2010). Despite their popularity, the effects of mountain bikers are still relatively unknown and have rarely been compared with other trail use types (Marion and Wimpey, 2007).

Although horses may have a high potential to introduce species through their manure (Campbell and Gibson, 2001), it remains uncertain which user group is likely to introduce the most species (Lonsdale, 1999). There has been report of hikers acting as vectors, with note that socks and shoes contribute to the transport of seeds and propagules (Mount and Pickering, 2009). Interestingly though, no studies exist that document introduced species on mountain biking trails or have examined mountain bikes as vectors of these species (Pickering *et al.,* 2010). This is surprising because of the known association between motor vehicle tires and seed transport (Mount and Pickering, 2009). Mountain bike tires may carry and transport seeds in the same manner that motor vehicles do. Moreover, mountain bikers could transport seeds and propagules similar to hikers, via their shoes and clothing.

With the exception of specialized mountain bike trails that include jumps and other technical features, recreational trails are commonly used by all three user groups (i.e. multi-use trails). Interestingly however, studies which examine the effects of these users cumulatively are rare and are also limited to direct impacts of soil loss. White *et al*., (2006) found indicators of soil loss to be similar between multi-use, hiking, and mountain bike trails. Further research of multi-use trails under different impact criteria would be particularly valuable for management purposes.

Although there is some evidence that equestrians have a greater impact than hikers and mountain bikers on trails, the majority of comparative research has been limited to direct impacts. Furthermore, it is largely unknown how other impact criteria such as species diversity and species composition will be influenced by different user groups. Researchers, protected area managers and some user groups agree on the need for more research on the comparative impacts of hiking, mountain biking, equestrian and multi-use trails (Cole and Spildie, 1998; Marion and Wimpey, 2007; Newsome *et al.,* 2008). Further understanding and quantification of how trail users influence vegetation and species diversity may allow managers to make better decisions pertaining to the conservation of floral communities.

### 2.4 Temporal Patterns of Impact

Despite the fact that time can play an integral role in the mitigation of impacts, there is still undoubtedly a lack of research on temporal patterns of impact. The understanding of how disturbance varies through time is essential for assessing impact magnitude and for management to decipher when mitigation measures should be taken. Mitigation success is then partially time-dependent due to the fact that areas exposed to recreational impacts typically have a 'life history' and go through stages of development, dynamic equilibrium and recovery (Cole, 2004). The life history theory is based on evidence observed from campsites with impacts thought to be similar to those of trails (Marion and Cole, 1996). During the abovementioned stages, the magnitude and type of impact can vary and mitigation efforts must thus reflect this.

Generally, recreational impacts increase rapidly during the first season of use and trail side vegetation exhibits a sharp reduction in the total percent coverage and species richness (Marion and Cole, 1996). This is known as the development phase. After the initial disturbance, the rate of impact decelerates until equilibrium is reached, usually a few years later, depending on the resilience of the area (Cole, 2004; Connell, 1978). As trails become more established, however, the total coverage can potentially exceed what existed there in the first year of use. This does not suggest that increased use causes a higher percent coverage in a linear fashion but can be explained by a change in the species occupying these areas (Marion and Cole, 1996). Thus, the impact magnitude for criteria such as vegetative composition can be influenced by when impacts are assessed. Direct impacts will generally remain stable if the frequency of use and the characteristics of use do not radically change (Fish *et al.*, 1981; Cole, 1991). However, abnormal behaviour such as rerouting a trail or creating mountain bike jumps can cause significant increases in impact despite the stage in the trails' life history. Indirect impacts such as species diversity may fluctuate in severity depending on the frequency of introduction and invasiveness of introduced species.

The recovery phase is the final stage in the life history of an area influenced by recreational impacts. This stage represents the time between the absence of recreational use and when environmental parameters have returned to pre-disturbance conditions. Since recovery is a slower process than deterioration, this stage is usually longer than the development phase (Eagen *et al.*, 2000). The length of recovery, however, fluctuates greatly depending on the type and magnitude of impact. For example, a study by Cole and Monz (2002) found that alpine grasslands which were trampled 1000 times had a faster recovery rate than flora in a neighboring forest trampled fewer than 100 times. Based on this example, it is evident that the ecosystem type also plays a significant role in the rate of recovery. When under similar environmental conditions, however, areas that are more frequently used will take more time to recover (Cole, 2004).

Since the magnitude of impact can vary significantly depending on the stage that the affected environment is in, it is essential to use a method of analysis that matches the objectives to be achieved. If attempting to quantify immediate or user intensity impacts, an experimental approach would be appropriate and has typically been used in the past. On the other hand, an analytical/comparative approach may be used to assess impacts that occur in the long term, allowing species to adjust and composition to change. This may be of concern to conservation agencies because of the interest in determining if floral communities are resistant to trail-related disturbances and if significant changes occur after extended periods of use.

## 2.5 Spatial Patterns of Impact

Although not typically studied, spatial patterns of impact are essential in determining the degree of impact and in creating strategies for managing them. The intensity of impact heavily depends on the location and spatial scale chosen for analysis (Cole, 2004). For example, if observing impacts solely on the trail tread, species richness will likely be severely reduced compared to undisturbed vegetation. However, if observing vegetation at the trail edge, species richness could potentially be even higher than undisturbed areas, as supported by the Intermediate Disturbance Hypothesis (Connell, 1978). Thus, the magnitude of impact is largely reliant on the chosen scale of analysis and should appropriately be taken into consideration when reporting results that quantify the magnitude of impact.

Of particular importance to the quantification of spatial impacts is the chosen measure of impact. In the case of trails, the study of impacts on vegetation has primarily been limited to experimental studies. Such studies typically assess immediate and direct disturbances caused by trail users such as vegetation coverage reduction caused by trampling (Weaver and Dale, 1978). This has led to the assumption that impacts are concentrated alongside the path with little impact observed off the trail. While the trail itself will reveal maximum impact, limiting the study area to a few feet beyond the trail edge ignores the potential that indirect changes may be occurring in areas previously thought to be undisturbed. Such indirect changes may include shifts in species composition and changes to the the richness of floral communities. Although the influences/causal factors that affect floral biodiversity have been documented, the spatial extent of trail impacts is relatively unexplored and has raised the hypothesis that spatial impacts occur both directly and indirectly.

Changes to biodiversity occurring beyond the trail edge are suspect for several reasons.

First, studies of other types of disturbances have found shifts in species composition to penetrate well into the interior of forests. For example, Buckley *et al*., (2002) found skid trails that measure 10m in width to influence species composition and diversity out to 100m from the trail. Secondly, trails can act as conduits for introduced species which may persist and invade adjacent habitats (Lonsdale, 1999). The spatial extent to which this occurs is dependent on both the competitive ability of native species and the degree of disturbance which require further investigation in the context of recreational trails. The placement of trails and proximity to one another may also cause impacts to occur over large distances. While the adjacent vegetation of one trail is unlikely to penetrate and change the entire composition of a forest stand, several trails in close proximity to each other potentially could, if the outer reaches of these impacts overlap. Moreover, at the landscape level, impacts can increase over time due to a lack of recovery on rerouted trails further exacerbating the effects. Lastly, the spatial extent to which trampling occurs in the long term remains unknown. Although most users typically do not stray from the trail, some users may venture off the trail to take a shortcut, view flora or fauna, to pass other trail users, avoid muddy areas or fallen trees, pick mushrooms, and partake in geocaching or simply to explore (Bayfield, 1971; Dale and Weaver, 1974; Bayfield, 1973; Bright, 1986; Davies and Newsome, 2009).

A possible explanation as to why researchers have not assessed spatial impacts beyond a few metres of the trail could be due to limitations in the analysis of these effects. Specifically, researchers have quantified changes in diversity with increasing distance from the trail using separate transects as replicates (Dale and Weaver, 1974; Hall and Kuss, 1989; Weaver and Dale, 1978). This method typically finds that composition does not change with distance and has led researchers to believe that impacts are confined to the trail edge. A possible explanation for this may be due to the method used which allows for the same species to be counted more than once at a specific distance. Thus, uncommon species may be undervalued in an analysis of richness and dominant species may distort the results because of reoccurring presence. Analysis is thus needed whereby the number of species found in separate replicates is summed for each distance without accounting for the same species twice. Such analysis will provide a better quantification of the spatial influence on a community scale and is conducted in this study.

In the literature, researchers have called for a better understanding of the spatial extent of recreational impacts (Liddle, 1997). Such knowledge would be of assistance to those wanting to preserve biodiversity and species of high conservation priority. The fact that trail impacts can persist even after recreational use has stopped raises a higher cause for concern. Therefore, assessment of impacts beyond the trail edge appears to be a next logical step in the understanding of trail impacts on vegetation and is one of the major components of this study. Quantifying impacts spatially should increase the ability to draw useful conclusions about the ecological significance of impacts and may result in wiser judgments regarding recreational impact mitigation.

### 2.6 Trail Impacts and Biodiversity

Cumulatively, the effects of trails can influence floral diversity. In its simplest form biodiversity can be measured as a count of the number of species that occupy an area, known as species richness, and the proportional abundance of species, known as evenness. Changes to diversity occur because species greatly differ in their ability to tolerate disturbance. More specifically, they vary in their ability to resist injury, in their ability to recover from injury and in their ability to survive in disturbed conditions (Dale and Weaver, 1974; Cole, 1982). Therefore, in the presence of disturbance, some species may increase in abundance and others decrease, while some can be removed entirely (Zabinski *et al*., 2000). In the event that disturbance can remove the entire population of a species from an area, species richness will be reduced.

The diversity of vegetation can also be significantly influenced by the introduction of invasive species. Introduced species that are characteristically well-adapted to periodic disturbances and are capable of surviving in the altered conditions can potentially fill the void created by trail impacts (Lonsdale, 1999). Therefore, the richness of introduced species may be increased and the richness of indigenous species may consequently decrease (Benninger-Truax *et al*., 1992). Shifts in species composition may also influence the evenness of a community. This is likely to occur if introduced species can successfully colonize an area and reduce the dominance of indigenous species.

Generally speaking, recreation areas characteristically have vegetation that is less abundant and have different species composition compared to undisturbed areas (Cole, 1982, 1993; Luckenbach and Bury, 1983). Trail-influenced environments can be less diverse (Boucher *et al*., 1991), but they may also be more diverse (Hall and Kuss, 1989). The amount of change to biodiversity is likely a combination of both the level of habitat disturbance and the degree to which invasive or opportunistic species are able to establish themselves in the disturbed environment. Although the avenues for these changes are roughly understood and documented, the magnitude of these changes are relatively unknown (Liddle, 1997; Cole, 2004). Specifically, it is unclear what the spatial extent of these changes is, which users are likely to have the most impact on biodiversity and how the diversity of different environments will react to trail impacts, among others.

# Chapter 3: Objectives

The overall goal of this study was to improve the understanding of trail use types and their relationship with floral composition after extended periods of exposure. As discussed in the literature review, hikers, mountain bikers and equestrians create similar types of impacts when traversing trails. However, the severity and spatial extent of impact to vegetation compared among these trail use types remains largely unknown. To determine whether recreational trail use types influence floral composition on a large community scale, vegetation was sampled beyond the trail edge. By establishing the spatial extent and severity of impact users have on floral composition, the respective trail use types were then compared. The second aim of the study was to determine if specific trail characteristics contributed to these impacts.

Therefore, the specific objectives of this study were to: i) determine if floral communities adjacent to trails differ among trail use types and baseline areas; ii) determine if a relationship exists between distance from the path and invasive species, vegetation coverage, species richness, species diversity and species sensitivities; iii) investigate the influence specific trail characteristics may have on vegetation composition; iv) investigate the relationship between user groups and invasive species; and, v) suggest improvements to trail system planning for the conservation of species diversity.

The study results were used to further understand and quantify the impacts recreational trail users have on floral composition in temperate deciduous forests. With this knowledge, managers may have a better understanding of the potential impacts caused by users. In addition, it would also allow them to make justified decisions regarding trail system designs and which users should be permitted on their lands depending on conservation objectives.

# Chapter 4: Research Methodology

## 4.1 Description of Research Methodology

The data collection methodology was determined using best known practices in the literature on vegetation sampling, environmental monitoring and recreational impact analysis (Cole, and Bayfield, 1993; Kershaw and Looney, 1985; Liddle, 1997). The research took an analytical/comparative approach as opposed to an experimental approach in attempt to capture long term effects. Unlike most user impact studies, it was possible to compare user impacts that occurred on different trails because these areas were determined to be homogeneous and conceivably free of bias. Thus, differences in impact between trail use types will provide justification for further comparisons between these groups. The remainder of this chapter describes the methods used to determine homogeneity of baseline conditions and user affected environments.

For the purposes of this study, ‘impact’ is used to describe the difference between baseline conditions and user affected environments. Thus, impact does not imply a positive or negative result but simply that a difference exists. A primary objective of this study was to examine trail impacts resulting from long term exposure to recreational use. The plant communities studied have been subject to recreational use for at least 20 years and have had ample time to equilibrate to continued use. Since the study did not attempt to assess the change in impact over time, it assumes that impact has already taken place. It is important to note that this is not a mechanistic study of impacts but rather a comparison between areas with changes that can be attributed to a distinct source of impact.

### 4.2 The Study Location

The location for this study is an assemblage of properties managed by the Toronto and Region Conservation Authority (TRCA), commonly referred to as the East Duffins Headwaters (EDH). They are located in a rural landscape between Uxbridge and Pickering, Ontario, Canada, on the Oak Ridges Moraine (Figure 2). The Walker Woods Tract and Pleasure Valley Forest properties were chosen for this study based on the criteria that they have large naturally forested areas, the presence of hikers, mountain bikers, equestrian users and their known use frequencies.

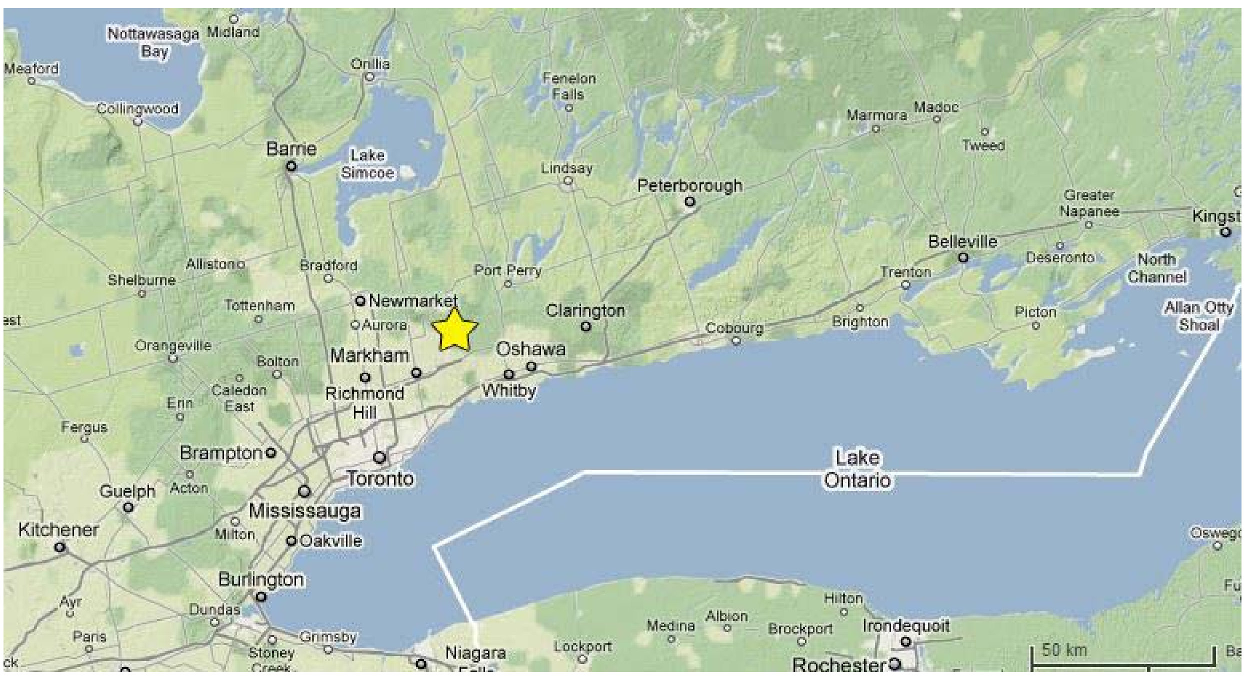


**Figure 2. Map of Study Area.** The star indicates the location of the study area within Southern Ontario, Canada (http://maps.google.ca/).

#### 4.2.1 Biotic and Abiotic Features of the East Duffins Headwaters

The topography of the EDH consists of rolling till deposits which, combined with various soil and moisture conditions, result in a large variety of vegetation communities. The majority of these properties are covered by mature forests which are classified as the Great Lakes-St.

Lawrence Mixed Forest Zone (MTRCA, 1982; Rowe, 1972). Although the watershed does not include any of the Carolinian Zone there is a significant number of Carolinian flora species, including Bitternut Hickory (*Carya cordiformis*) and Black Cherry (*Prunus serotina*) (MTRCA, 1982). Some areas in the EDH are designated as provincially significant for their geological features and diverse communities. The properties surrounding these lands include agricultural fields, farm houses and other stands including the Durham Regional Forest.

**4.2.2 Management of the East Duffins Headwaters**

The properties in the EDH were assembled by James and Olwen Walker in the 1950s to be rehabilitated and reforested. Prior to this, the lands had been operated privately for logging and agriculture. They were acquired in 1991 by the TRCA and since then, no other major land use changes have occurred there.

The Walker Woods Tract covers 744 acres and the Glen Major Resource Management Tract spans 400ha (Figure 3). The total length of trail managed by the TRCA in the EDH is 74km. The permitted activities on these trails include hiking (including leashed-dog walking, snow-shoeing, bird watching and wildlife viewing), cycling, horseback riding, and cross-country skiing. These trails are open for public use and there are no limits on frequency of use for general visitors on these lands; however, there are restrictions for special events such as races and groups that exceed 20 people.

The Pleasure Valley Forest was another property used in this study and it covers 48ha. This property is not open to the public and is leased to an equestrian club. This club has exclusive use of these trails which are ridden daily by horseback riders. The estimated length of trails on this property is 8km.

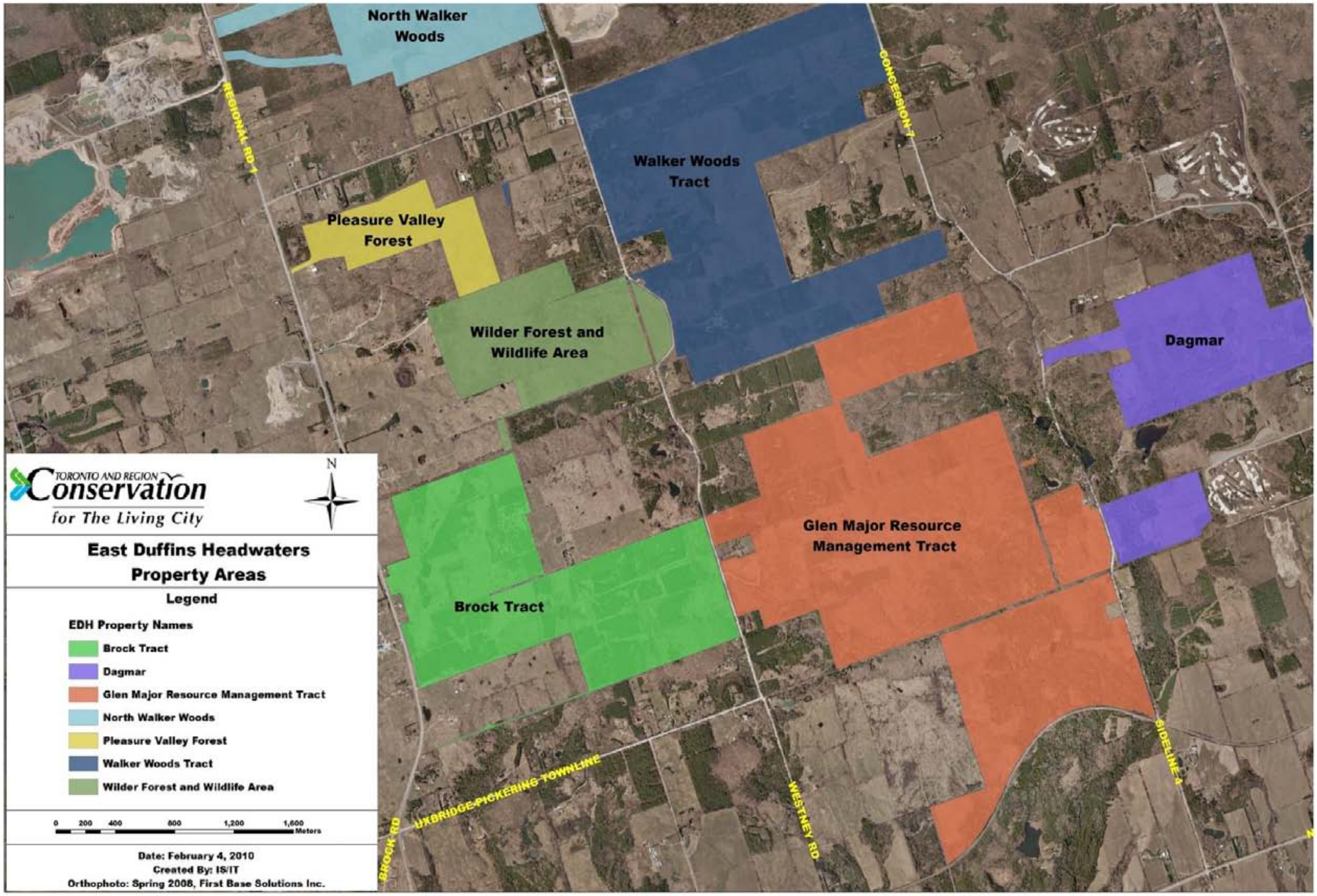


**Figure 3. East Duffins Headwater Properties.** This study focused on the Pleasure Valley Forest (yellow), and the Walker Woods Tract (dark blue).

### 4.2.3 Use Frequency in the East Duffins Headwaters

A user survey which examined the level and types of use in the EDH was undertaken by TRCA staff from November 2008 – December 2009. The results of these surveys should only be considered an approximation as the study is still in draft. Infrared trail counters were used at all of the entrances to the properties and the estimated number of times a person entered the EDH properties was approximately 53 000 a year. Based on extrapolation and weekly averages, the number of users per week was broken down by season and is as follows: winter – 633, spring – 463, summer – 1681, fall – 1070.

The frequency of use types was recorded by TRCA staff and volunteers at the property entrances. The monitoring took place at various times throughout the year and was evenly distributed through the week. Based on the results from the surveys, the trail use groups were broken down as Mountain Biking 52 ± 10 %, Hiking 44 ± 10 %, and Equestrian 4 ± 0 %. These values represent the average percentage of users in the spring, summer and fall months.

The frequency of use of the Pleasure Valley Forest lands was not included in the EDH user study. However, the manager of the equestrian club estimated the number of times the trails were used per week by horseback riders to be approximately 125. When comparing the frequency of use of the EDH to Pleasure Valley the frequency of use of the two properties was similar, based on the weekly averages and the length of trail. The frequency of use on Pleasure Valley lands was estimated being 125/8km = 15.5/km/week and the EDH was based on an average of spring, summer and fall data being 1071/74km = 14.5/km/week. Thus, the frequency of use between the two properties should be considered similar.

The study assumes that the user groups use each of the trails available to their respective groups equally and that the level of use is constant along all of the trail length. Also, this research had no feasible way of determining the level of use for each particular trail, other than equestrian trails provided above. To avoid a bias, the trails chosen for the study were those commonly used by the different trail user types and judged to be representative of the maximum user frequency of those groups. This was determined in consultation with TRCA management.

**4.3 Research Design**

The research design was similar to other studies that investigated ground flora impacts including Cole and Bayfield (1993), Buckley *et al.,* (1997), Gallet and Rozé (2002), and Roovers *et al.,* (2005). These studies sampled vegetation using transects which extended from the trail into the adjacent forest. For the purposes of this study, ten transects were sampled for each trail use type which included hiking, multi-use, biking and equestrian activities as well as the baseline area. The transects were placed randomly by determining the length of trail available for sampling and using a random number generator (Microsoft Office Excel, 2007) to determine where each transect would be sampled. The transects were also placed on either side of the trail, five on one side, five on the other. The geographic coordinates were recorded using a Trimble GeoXH unit for future reference which had an accuracy of ± 30cm. The transects were placed directly adjacent to the trail edge and ran 25m into the forest interior. Six quadrants ($1m^2$) were then sampled within each transect at 0m, 2.5m, 5m, 10m, 17.5m and 25m. This provided a continuous view of the vegetation from the trail and allowed for a gradient of change to be determined which is common in vegetation research (Goldsmith *et al*., 1986). Ten transects were sampled for each trail use type that contained six quadrants each, making 60 samples.

Baseline areas were established to develop a control and were sampled similar to the method for trail user groups using transects and quadrants (0-25m). Transects were distributed in the baseline areas by breaking them down using a grid pattern. Then, using an online random number generator, ten areas were chosen to be sampled. These areas were located using a Trimble GeoXH unit and orthophotos. The cardinal direction that each transect faced was also randomly chosen to avoid bias. The total number of quadrants sampled was 300 when taking into account baseline quadrants. The sampling apparatus was a fixed $1m^2$ (2m x 0.5m) unit used to keep a consistent sample size. The longer edge of the sampling apparatus was placed parallel with the trail and represented a quadrant. The whole of the quadrant was broken down into 4 % subsections to finely assess the species composition and coverage (Figure 4). The sample size was consistent with the vegetation sampling literature (Liddle, 1997).

**Figure 4. Sampling Apparatus.** The sampling apparatus was a fixed $1m^2$ (2m x 0.5m) unit.

### 4.4 Field Measurements

The vegetation data were collected between September 20th and 30th, 2009. Although sampling ground flora in this region is normally conducted in July and August, based on the expertise of senior ecologists at the TRCA, sampling at this time of year would not have changed the results. This is because micro-climates created in the interior of a mature forest such as the ones sampled would sustain most species well into the fall. The only advantage of sampling earlier in the summer would have been the presence of flowers, which makes for easier identification.

*Vegetation (%): Vegetation was sampled within the placed quadrants by indentifying each* individual species and its percent of area coverage. Species were identified with help of Newcomb's Wildflower Guide (1977) and were confirmed by senior ecologists at TRCA using photographs of the species. Percent coverage was used instead of counting the number of individuals of each species because of the difficulty in distinguishing individuals without damaging them. Only species shorter than 1m in height were recorded which was consistent with other vegetation sampling studies (Kershaw and Looney, 1985). Great care was taken to limit the damage caused to flora and invertebrates present during sampling.

*Tread width (metres):* The distance considered to be the beaten path or visible trail surface where underlying soil has been exposed. This distance was taken using a tape measure positioned perpendicular to the direction of the trail.

*Trampled width (metres):* The minimum distance between vegetation on either side of the trail. This distance was taken using a tape measure positioned perpendicular to the trail direction.

*Trail depth (inches):* The maximum depth of the trail from the highest to the lowest point across the trampled width. This was taken by laying a retractable pole across the trampled width and measuring the distance between the leveled pole laid horizontally and the ground.

*Trail slope (degrees):* The typical grade or steepness of the trail was measured by two researchers using Clinometers. First, the instruments were calibrated to 0 degrees on level ground. Then the individuals stood 3m apart, faced each other in the centre of the trail and used Clinometers to determine the degree of slope.

*Trail elevation (metres above sea level):* A Trimble GeoXH unit was used to determine the coordinates and the elevation. These raw data provided a measure of elevation in metres above sea level. Waypoints were taken at the centre of the trail in line with the transect.

*Transect slope (degrees):* The grade or steepness of the transect was also measured using Clinometers. This process also took two operators to complete the task. First, the instruments were calibrated to 0 degrees on level ground. Then, individuals stood facing each other, one at the trail edge and the other at the 25m quadrant. The degree of slope was determined by looking through the Clinometers.

*Transect aspect*: The aspect of each transect was determined using a compass. Aspect refers to the cardinal direction that a hillside or slope faces which affects the amount of sunlight striking the land's surface. In the northern hemisphere, it is generally assumed that south-facing slopes receive more sunlight than north-facing slopes. This method was used to roughly distinguish between areas receiving varying amounts of light exposure. The aspect was recorded as being either north, south or as having no aspect which would indicate level ground.

### 4.5 Sampling Area Criteria

To enable an unbiased comparison between the user groups, it was essential to choose areas that were homogeneous. Comparability was based on the types of vegetative communities, their quality and the presence/absence of other influential factors. The same vegetative communities were sampled for each trail use type using the Ontario Ministry of Natural Resources Ecological Land Classification (ELC) protocol. ELC is a tool used for consistently identifying and describing vegetative communities (Lee, 1998). ELC data previously collected by the TRCA were used to identify comparable study sites. The communities used to compare the trail use types and baseline areas included FOD5-1 (Dry-Fresh Sugar Maple Deciduous Forest) and FOD5-2 (Dry-Fresh Sugar Maple-Beech Deciduous Forest) (Figure 6). Although the canopy in FOD5-2 may consist of a higher proportion of mature Beech trees, the ground vegetation is considered to be heterogeneous. Thus, both have the same understory composition and species distributions. A deciduous forest type was chosen for the study because the vegetation in this environment is considered to be vulnerable to recreational impacts (Kuss, 1986) and is more sensitive than meadow communities (Cole, 1978). It is also the preferred environment for most trail use types and thus likely to incur impact in the future.

The quality of the vegetative communities was also assessed for similarity. This was determined using the TRCA Landscape Analysis Model which is congruent with other models in the landscape ecology literature (McGarigal and McComb, 1995). The model evaluates the quality of a parcel of land based on its size, shape and matrix influence giving it a score. The size of a parcel of land adds to its quality because of the amount of interior habitat it has. Forest interior is the area of the forest that is considered to be free of negative external influences and capable of supporting sensitive species (MTRCA, 1982). The shape of a patch will also influence the amount of interior habitat. For example, a patch that is relatively more linear will have less interior area than a patch that is relatively more round in shape. Patch shape is assessed on the ratio between its size and the amount of edge it has. The matrix (surrounding land use) is assessed by the ratio of urban, agricultural and natural cover surrounding the patch. Using existing TRCA data mapping, it was determined that all of the communities sampled in this study received the highest score as shown in Figure 5 below.

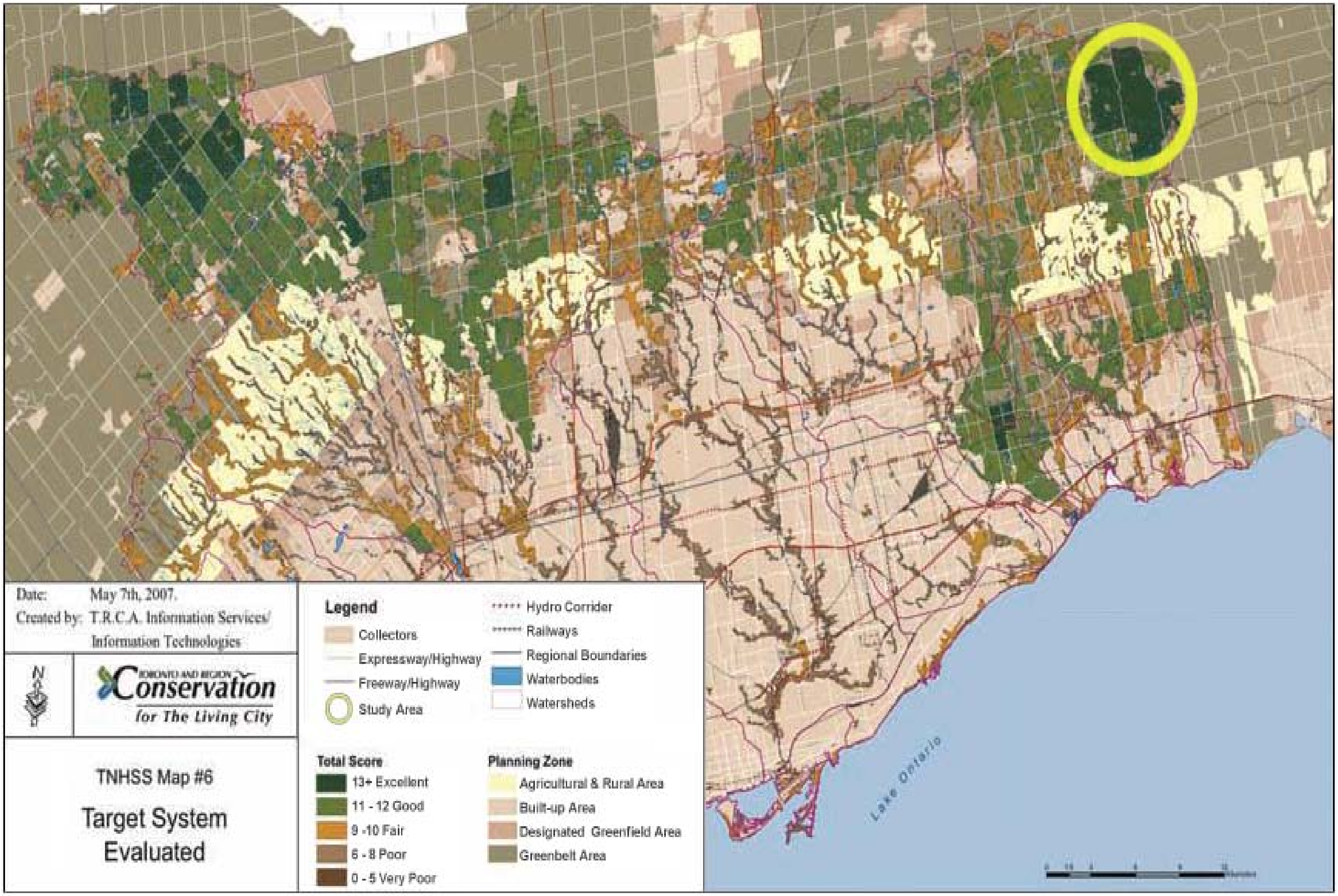


**Figure 5. Map of Evaluated Patch Communities in TRCA Jurisdiction**. The yellow circle indicates the location of the study area.

### 4.6 Sampling Area Controls

The absence of other disturbances was also a requirement in the trail selection process. This allowed for trail user impacts to be assessed in isolation. The non-trail user-related impacts that were avoided were as follows:

*Roads*: The mobility of some species may be impeded by roads. They can also increase habitat loss, and create wildlife disturbances which influence the normal distribution of flora. A minimum distance of 100m from the edge of the road was required for siting of the transects and quadrants. At this distance roads are assumed to have no effect on species composition.

*New Trails*: The creation or rerouting of trails can influence surrounding species distribution and composition because of increased erosion and trampling. Only well established trails of 5 years old or more were included in the study. The minimum ages of the trails studied were confirmed by TRCA trail managers.

*Unauthorized Uses*: Areas where unauthorized uses such as the presence of motorized vehicles as well as mountain bike jumps and stunts were excluded from the study area because of their non-typical impacts.

*Other Trails*: The proximity to other trails can influence species distribution and the degree of impact. If trails assessed are too close together, their impacts to floral diversity may be combined and will not be representative of either trail. A minimum distance of 50m from other trails was required in order to isolate the impacts of a single trail user type. This also included trail junctions where two or more trails met. At this distance other trails are assumed to have no effect on species composition.

*Other Vegetation Types*: The proximity to other vegetation types can influence species distribution and the degree of impact. A 50m buffer from other ELCs for quadrants was required to ensure that samples were not biased by other vegetation types.

*Watercourses*: Trails that traversed watercourses were avoided because they act as a barrier to some species distribution and could potentially influence results.

## 4.7 Trails Studied

The specific trails were chosen because of their proximity to each other; since biodiversity is linked with area and space, it was expected that this would reduce the opportunity for bias to arise (MacArthur and Wilson, 1967; Williamson, 1981). Trails chosen for study were no greater than 1.5km from each other.

*Hiking*: The trail chosen to represent hiking impacts is located on the Walker Woods property and has coordinates of Northing 4877489 and Easting 651909 at the middle of the trail (Figure 6). Although not restricted to other uses, this trail is considered a hiking-only trail for several reasons. Trail managers noted that only hikers had been seen using this trail and suggested that this was because other trails close to the 'hiking-only' trail were more attractive to the other users.

*Multi-Use*: The trails chosen to represent multi-use impacts were also located on the Walker Woods property and had coordinates of Northing 4877013 and Easting 652129 at the middle of the trail (Figure 6). This portion of trail is marked as the Trans Canada Trail and can be enjoyed by all users permitted on the property. The wide trail path accommodated all user groups. The Trail runs primarily through the property and is used by most visitors that enter Walker Woods.

*Mountain Biking*: The trail chosen to represent mountain biking impacts is located on the Walker Woods property and has coordinates of Northing 4877029 and Easting 652218 at the middle of the trail (Figure 6). This portion of trail is marked as 'mountain biking only' at the entrances to the trail. Typically, hikers and equestrians pursue a certain route with a destination in mind, whereas mountain bikers take certain trails because of their challenging topographical and technical features. Mountain biking trails are also notorious for having many switchbacks and do not follow a direct route which is undesirable for most other user types. Therefore, it is highly unlikely that any other user group would choose to travel this trail because of its topography. Moreover, equestrian riders would not be physically able to traverse this trail because of obstacles such as closely spaced trees and low branches that exist on mountain biking trails.

*Equestrian*: The trail chosen to represent equestrian impacts is located on the Pleasure Valley property and has coordinates of Northing 4876328 and Easting 650459 at the middle of the trail (Figure 6). The Pleasure Valley equestrian club has exclusive use of these trails and no other user groups have used these trails previously.

*Baseline/Control Area:* The baseline areas were chosen using the same criteria as the user group study areas, the only difference being the absence of any trails, making each area a control. The absence of trails was established using preexisting maps of all formal and informal trails on the properties. The reserve areas of the properties, where environmental features are thought be to greatest protected, were chosen because of their large undisturbed area with no existing trails present. The reserve areas allow no public access and in some cases are fenced off. A buffer zone of 100m from other disturbances or changes in ELC was used to isolate the area from external impacts. The baseline areas chosen include reserves on the Walker Woods Tract and Pleasure Valley Forest properties (Figure 6).

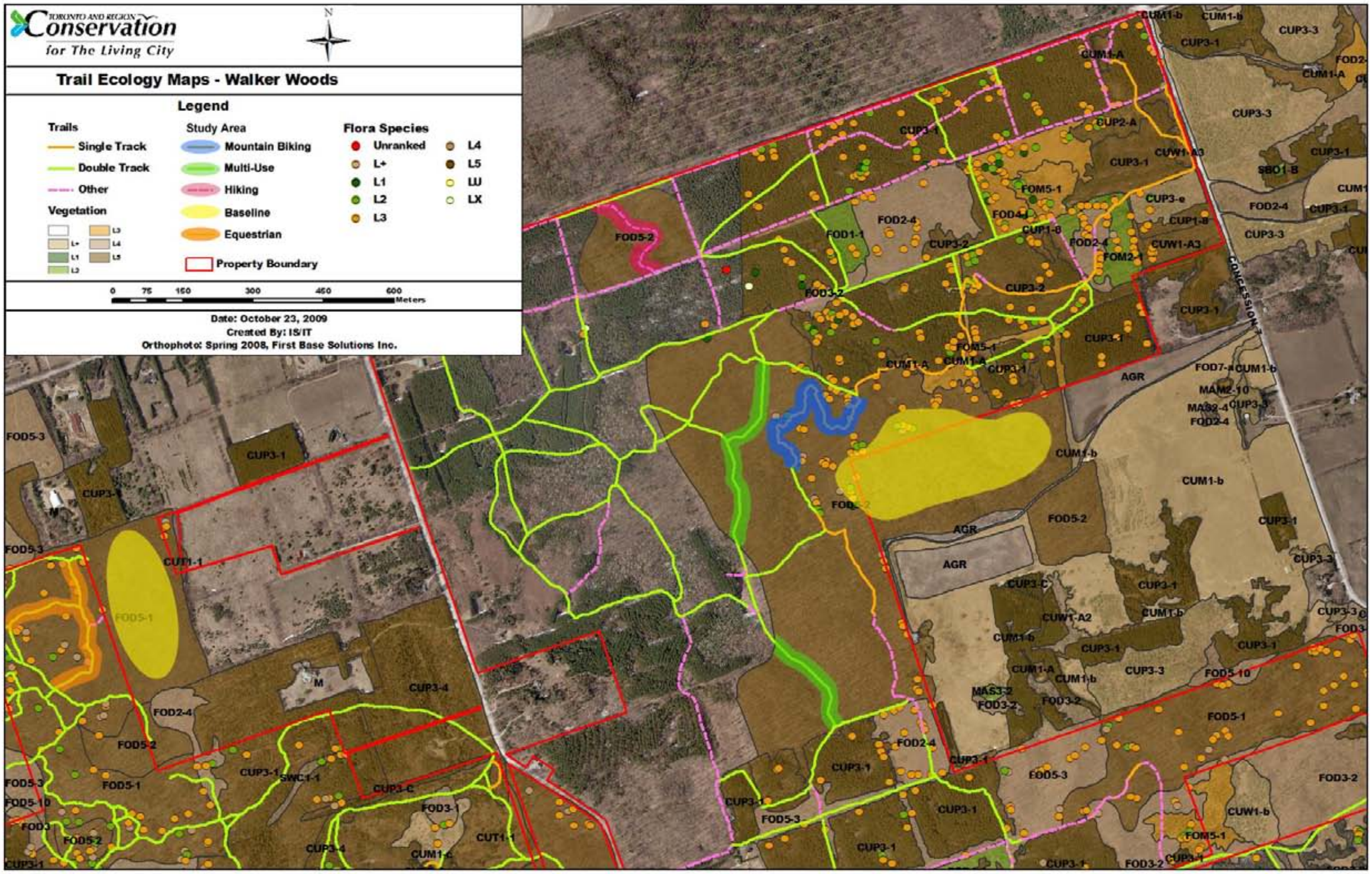


**Figure 6. Map of Trails Sampled, Baseline Areas and ELC Communities in Walker Woods Tract and Pleasure Valley Forest.**

## 4.8 Descriptors of Vegetative Composition

A number of different approaches were employed to describe the vegetative composition of user affected communities and baseline areas. These include species richness, vegetation percent coverage, evenness, diversity indices, species sensitivity scores and invasive species. Diversity measures were used to compare user impacts primarily because the range and severity of impacts of different recreational activities is still unclear. Thus, using measures that account for cumulative impacts may suggest avenues for impact and identify the need for further investigation. The use of sensitivity analysis is expected to enhance the understanding of how impacts occur and what their potential to alter the composition of trail-influenced environments might be.

*Species Richness*: is the total number of unique species found within a certain area (quadrant, transect or trail use type).

*Vegetation Percent Coverage:* is the percentage of area covered with vegetation. This was calculated by summing the percent coverage of each of the species found within a quadrant.

*The Shannon Index:* is a measure of biodiversity that takes into account both species richness and evenness. It provides a score that gives more value to higher species richness than evenness of populations. Thus, this measure is weighted in favor of communities with a high proportion of rare species as opposed to dominant species which is useful for detecting differences between sites.

The Shannon index is calculated from the equation:

$$H = - \Sigma p_i \ln ( p_i )$$

Where, $p_i$ is the proportion of total number of species made up of the $i^{th}$ species and ln is the natural logarithm (Shannon, 1948).

*Evenness:* Species evenness refers to the proportion that each species comprises of the whole. It is a measure of biodiversity and can be derived from the Shannon Index. Evenness is calculated from the equation:

$$E = H / \log( S )$$

Where $H$ is the Shannon Index value, and $S$ is the total number of species.

Evenness values are given between 0 and 1 where 1 represents a situation in which all species are equally abundant.

*TRCA Species Sensitivity Scoring:* The TRCA scores species to provide guidelines for natural heritage protection and management. This is done by assessing the species and giving each one a score on an ordinal scale in relation to other species (Appendix A). The data sources for scoring included a large number of studies in the existing literature, TRCA data and professional experience. For the purpose of this study, sensitivity scores developed by the TRCA were used to compare impacts from different trail users (TRCA, 2010). The sensitivity score criterion identifies eight negative impacts and two types of disturbance that some species may benefit from. Negative impacts include:

*i)* Removal, Collection, Weeding: Some plants are frequently dug or harvested for ornamental, food or medicinal use. Others are removed because they are considered undesirable.

*ii)* Airborne contaminants: Contaminants include acidic effects from car exhaust, particulates, ozone, etc. Effects from car exhaust can affect all of southern Ontario but can be expected to be even higher downwind from major roads.

*iii)* Surface-borne contaminants: Some pollutants can be directly deposited onto the soil or carried to the site by trail users, e.g. road salt, oils and grease, etc.

*iv)* Herbivory: Some species are more prone to being eaten by fauna, e.g. squirrels, deer, raccoons.

*v)* Invasive species: A species may be particularly prone to being out-competed by an invasive species that occupies a similar habitat but is more aggressive.

*vi)* Trampling and trail formation: Some species may be less tolerant to exposure or compaction caused by trail use. This is because some species only produce one set of leaves per growing season. They may also be considered more sensitive if they have delicate leaves, root systems or fibrous stems.

*vii)* Hydrological changes: Trails can cause some changes in hydrological effects such as flooding, drainage and increases in evaporation caused by breaks in the canopy.

*viii)* Dynamic Process, suppression of natural disturbance: Land management may suppress the forces that support a species, such as production-oriented forestry practices which involve the removal or alteration of habitat structure, e.g. snags, logs, unproductive trees.

Disturbances which can benefit some species include:

*ix)* Fertility: The species can benefit from increased nutrient loading as happens with horse or dog excrement, sedimentation or other nearby sources such as agriculture.

*x)* Soil disturbance: The species can benefit from soil disturbances such as ploughing, excavation, and dumping of fill.

The score ranges from 0 to 5 with a score of 2 for uncertainty. Each species starts with a neutral score of 2. For each negative impact (i-viii), one point is added, while for each benefit (ix-x), a point is subtracted. Score interpretation is as follows:

**Table 1. Toronto and Region Conservation Authority Sensitivity to Development Scoring System.**

| SUM OF IMPACTS | SCORE | SENSITIVITY TO DEVELOPMENT |
|---|---|---|
| -2 | 0 | Species benefits significantly from development-related disturbance. |
| -1 | 1 | Species benefits slightly from development-related Disturbance |
| 0 | 2 | Negative impacts of development more-or-less offset by benefits, or unknown. |
| 1 | 3 | Significant negative impact from development. |
| 2 | 4 | Moderately severe negative impact from development. |
| 3 or more | 5 | Severe negative impact from development. |

*Invasive species*: Species designated by the TRCA to be regionally invasive based on their potential to persist in undisturbed areas and influence native species composition were also analyzed. Invasive species were given a score of 0 using the Sensitivity to Development scoring system shown above and are provided in Appendix A.

### 4.9 Data Analysis Method

The data collected through floral inventorying as previously described were evaluated using various descriptive and statistical approaches. Microsoft Office Excel (2007) was used to organize the collected field data which included the presence and abundance of species. From these data, species richness, vegetation percent coverage, species sensitivity, evenness, and Shannon Indices were determined for each quadrant. Total richness values were also determined for each transect and trail use type. The Shannon Index and evenness values were derived from an online calculator (Bioscience, 2010). Each other descriptor has a different method of analysis and will be described individually.

Species richness was evaluated by summing the total number of species for each trail use type. Richness was also evaluated at various distances from the trail (0m, 2.5m, 5m, 10m, 17.5m and 25m) for each trail use type. This was done by summing the number of unique species found at the respective distances, across ten transects.

Species sensitivity data were examined by grouping species with similar sensitivity scores. The coverage of those groups within each transect were summed and calculated as a percentage of the total area sampled. The mean of ten transects was then compared for each sensitivity score between the individual trail use types and baseline areas (i.e. percent coverage of 1 sensitivity species in hiking vs. 1 sensitivity species in equestrian). In addition, the influence of distance on the mean percent coverage of a single sensitivity was examined within individual groups (i.e. percent coverage of 2 sensitivity species at 2.5m vs. 10m, 2.5m vs. 17.5m, etc., within hiking trails).

Vegetation percent coverage, Shannon Indices and evenness were analyzed in two different ways. First, a comparison between the individual trail use types and baseline areas was made. This was done by taking the mean of six quadrants within each transect (n = 10 transects) to which the mean of these ten transects was compared among use types. Secondly, within each trail use type, the mean values (n = 10) of one distance were compared to all other distances (i.e. in hiking trails, 2.5m was compared to 5m, 2.5m to 10m, 5m to 10m, etc.).

### 4.9.1 Statistical Analysis

A one-way analysis of variance (ANOVA) was used to analyze the differences of the means from the obtained values as described above (species richness, vegetation percent coverage, species sensitivity, evenness, and Shannon Indices). If the overall p-value of the one-way ANOVA was statistically significant (< 0.05), the Tukey's HSD (Honestly Significant Difference) multiple comparison post-hoc test was conducted to determine which specific groups differed. This was done with GraphPad InStat (San Diego, USA) and allowed for further interpretation of the results. Similarly, the Tukey (HSD) post-hoc test was statistically significant if the p-value was < 0.05. Two groups were said to be marginally different from each other if the p-value was > 0.05 but < 0.1.

Correlation analysis was used to examine the relationship between a descriptor of floral composition and distance (from 0m to 25m). The Pearson's product-moment correlation coefficient (Pearson r) was determined using GraphPad InStat. This coefficient describes the direction and magnitude of the linear relationship. If the obtained two-tailed p-value was < 0.05, the correlations were considered significantly different from zero, and did not result from chance. It is important to note that prior to the use of any of the aforementioned statistical tests (ANOVA and Tukey's HSD post-hoc test), the raw data was first tested for normality with the Chi-Square Test using GraphPad InStat.

Multiple linear regression analysis was used to discern the complex relationships between the descriptors of vegetation composition and environmental variables (trampled width, tread width, trail depth, trail slope, transect slope, aspect and elevation). The raw values of the environmental variables were normalized by transforming them into z-values. Multiple regression analysis was completed using SYSTAT11 software (Chicago, USA). The environmental variables were removed in a backwards stepwise manner, starting with the least significant value, until all factors remaining in the final regression model had p-values < 0.15, which is a conventional approach. The final regression equation along with the coefficient of determination ($r^2$) and p-values are given in the results. Only variables with p-values $< 0.05$ were considered to significantly influence the dependant variables.

# CHAPTER 5: RESULTS

The overall objective of this study was to assess the spatial influence of trail-related impacts on floral communities which were influenced by different trail use types. As described in detail below, floral communities were examined by their presence/absence, diversity indices, species richness, vegetation coverage, evenness, growth-form, sensitivity to disturbance as well as by their presence of invasive species. Multiple regression analysis was also used to determine if certain characteristics of trails or environmental variables could explain variability in the floral communities.

## 5.1 Assessment of Individual Species within Trail Use Types and Baseline Areas

The assessment of vegetative composition in this study involved the identification of both indigenous and exotic species in Dry-Fresh Sugar Maple Deciduous forests and Dry-Fresh Sugar Maple-Beech Deciduous forests. For the purposes of this study, 'indigenous' refers to species that were found in baseline areas and 'exotic' refers to species which were not found in baseline areas, i.e. found only in trail-influenced environments. As previously mentioned, there exists a third group of species known as 'invasive species' which were designated by the TRCA to be regionally invasive for their potential to invade areas in absence of disturbance and for their threat to native composition. From the 300 quadrants sampled, 46 forbs, 11 trees, 11 shrubs, 4 sedges and 2 grasses were present, making a total of 74 species found. The presence and abundance of these species within trail-influenced environments (hiking, multi-use, biking and equestrian trails) and baseline areas are provided in Appendix A. Table 2 lists the top ten dominant species in these areas in order from highest to lowest percent coverage; these species made up the majority of the coverage in each trail use type. As expected, *Acer saccharum* ssp.

*saccharum* (sugar maple) was the dominant specie in both trail-influenced environments and in baseline areas. Other dominant species which were common to both groups include *Carex pensylvanica* (Pennsylvania sedge), *Fraxinus americana* (white ash), *Dryopteris intermedia* (evergreen wood fern) and *Aralia nudicaulis* (wild sarsaparilla). Furthermore, a large number of species was found exclusively either in baseline areas or only in trail-influenced environments (Table 3). For example, *Alliaria petiolata* (garlic mustard) was found only in areas where trails were present, whereas S*chizachne purpurascens* ssp. *purpurascens* (purple melic grass), *Polystichum acrostichoides* (Christmas fern), and *Corylus cornuta* (beaked hazel) were only found in baseline areas.

**Table 2. Top Ten Dominant Species in Trail Use Types and Baseline Areas as of Percent Coverage.**

| Trail Type | Scientific Name | Common Name | Percent Coverage (%) |
|---|---|---|---|
| **Trail Use Type Areas (Hiking, Biking, Multi-Use, and Equestrian)** | *Acer saccharum* ssp. *saccharum* | sugar maple | 10.8 |
| | *Carex pensylvanica* | Pennsylvania sedge | 4.3 |
| | *Maianthemum canadense* | Canada mayflower | 3.6 |
| | *Dryopteris intermedia* | evergreen wood fern | 3.2 |
| | *Fraxinus americana* | white ash | 3.1 |
| | *Polygonatum pubescens* | downy Solomon's seal | 2.6 |
| | *Aralia nudicaulis* | wild sarsaparilla | 2.3 |
| | *Geranium robertianum* | herb Robert | 1.1 |
| | *Alliaria petiolata* | garlic mustard | 1.0 |
| | *Carex blanda* | common wood sedge | 0.7 |
| **Baseline Areas** | *Acer saccharum ssp. saccharum* | sugar maple | 21.7 |
| | *Fraxinus americana* | white ash | 7.0 |
| | *Viburnum acerifolium* | maple-leaved viburnum | 4.5 |
| | *Carex pensylvanica* | Pennsylvania sedge | 2.2 |
| | *Aralia nudicaulis* | wild sarsaparilla | 2.1 |
| | *Caulophyllum thalictroides* | blue cohosh | 2.0 |
| | *Desmodium glutinosum* | pointed-leaved tick-trefoil | 1.7 |
| | *Dryopteris intermedia* | evergreen wood fern | 1.4 |
| | *Polystichum acrostichoides* | Christmas fern | 1.3 |
| | *Caulophyllum giganteum* | long-styled blue cohosh | 1.1 |

The percent coverage of an individual specie in baseline areas was summed from 60 quadrants (6 quadrants in 10 transects) and calculated as a percentage of the total area sampled, including the amount of bare ground. Similarly, the percent coverage of species from trail use areas was summed from a total of 240 quadrants (pool of hiking, biking, multi-use and equestrian trail quadrants).

**Table 3. Species Exclusive to either Trail-Influenced Environments or Baseline Areas**

| Trail Type | Scientific Name | Common Name | Percent Coverage (%) |
|---|---|---|---|
| **Baseline Areas Only** | *Polystichum acrostichoides* | Christmas fern | 1.26 |
| | *Corylus cornuta* | beaked hazel | 0.80 |
| | *Schizachne purpurascens* ssp. *purpurascens* | purple melic grass | 0.76 |
| | *Tilia americana* | basswood | 0.33 |
| | *Geum canadense* | white avens | 0.20 |
| | *Amphicarpaea bracteata* | hog-peanut | 0.10 |
| | *Equisetum arvense* | field horsetail | 0.06 |
| | *Mitella diphylla* | miterwort | 0.06 |
| | *Diervilla lonicera* | bush honeysuckle | 0.05 |
| **Trail Use Type Areas Only – (Hiking, Biking, Multi-Use, and Equestrian)** | *Alliaria petiolata* | garlic mustard | 1.01 |
| | *Viola pubescens* | stemmed yellow violet | 0.33 |
| | *Sambucus canadensis* | common elderberry | 0.27 |
| | *Impatiens pallida* | yellow touch-me-not | 0.19 |
| | *Osmorhiza claytonii* | woolly sweet cicely | 0.17 |
| | *Trillium erectum* | red trillium | 0.13 |
| | *Lonicera Canadensis* | fly honeysuckle | 0.12 |
| | *Matteuccia struthiopteris* var. *pensylvanica* | ostrich fern | 0.12 |
| | *Mitchella repens* | partridgeberry | 0.11 |
| | *Dryopteris carthusiana* | spinulose wood fern | 0.1 |
| | *Leonurus cardiaca* ssp. *cardiaca* | motherwort | 0.08 |
| | *Prunus serotina* | black cherry | 0.08 |
| | *Pinus strobus* | white pine | 0.06 |
| | *Rhamnus cathartica* | common buckthorn | 0.06 |
| | *Podophyllum peltatum* | May-apple | 0.05 |
| | *Oxalis stricta* | common yellow wood-sorrel | 0.04 |
| | *Anemone acutiloba* | sharp-lobed hepatica | 0.03 |
| | *Aquilegia canadensis* | wild columbine | 0.03 |
| | *Equisetum hyemale* ssp. *affine* | scouring-rush | 0.03 |
| | *Galeopsis tetrahit* | hemp-nettle | 0.02 |
| | *Asclepias exaltata* | poke milkweed | 0.01 |
| | *Fagus grandifolia* | American beech | 0.01 |
| | *Polygonum cilinode* | fringed black bindweed | 0.01 |
| | *Ranunculus abortivus* | kidney-leaved buttercup | 0.01 |
| | *Ribes cynosbati* | prickly gooseberry | 0.01 |
| | *Viola Canadensis* | Canada violet | 0.01 |

Same as in Table 2.

The spatial relationship between individual species and distance from the trail was then assessed using correlation analysis. Within individual trail use types, each species' slope (from 0m to 25m), correlation coefficient (Pearson r) and p-values were determined (Table 4). Few species were found to be correlated with distance from the trail. Interestingly, those that did exhibit this spatial effect were also found to be dominant species (Table 2) suggesting that off-trail trampling has occurred. Furthermore, *Aralia nudicaulis* (wild sarsaparilla) exhibited an opposite correlation with distance from the trail which suggests that not all species experience the same spatial effect. It is important to note that no species in baseline areas were considered to have any correlation among the various distances measured, including those in Table 4 (data not shown).

**Table 4. Relationship between Individual Species Coverage and Distance (from 0m to 25m) within Individual Trail Use Types.**

| Trail Type | Scientific Name | Common Name | Slope | Correlation Coefficient (Pearson r) | p-value |
|---|---|---|---|---|---|
| Hiking | *Carex pensylvanica* | Pennsylvania sedge | 0.921 | 0.856 | 0.029* |
| Multi-use | *Acer saccharum* ssp. *saccharum* | sugar maple | 0.433 | 0.810 | 0.051 |
| Multi-use | *Aralia nudicaulis* | wild sarsaparilla | -0.598 | -0.796 | 0.059 |
| Biking | *Trillium grandiflorum* | white trillium | 0.355 | 0.884 | 0.047* |

Slope, correlation coefficients and p-values were calculated from 0m to 25m for individual species in all trail use types. Only species found to be significant or marginally significant are provided in the table. The * denotes a statistically significant correlation coefficient which is different than zero ($p < 0.05$). Correlation coefficients not denoted with an * indicate marginal significance ($0.05 < p < 0.1$).

Where correlation analysis could not be completed due to species being uncommon, individuals were analyzed for their presence at various distances from the trail. Interestingly, the greatest number of uncommon species identified in trail-influenced environments were found within 2.5m of the trail (n = 18). These species include; *Caulophyllum thalictroides* (blue cohosh), *Viola canadensis* (Canada violet), *Veronica officinalis* (common speedwell), *Oxalis stricta* (common yellow wood-sorrel), *Pteridium aquilinum var. latiusculum* (eastern bracken), *Circaea lutetiana ssp. canadensis* (enchanter's nightshade), *Polygonum cilinode* (fringed black bindweed), *Galeopsis tetrahit* (hemp-nettle), *Leonurus cardiaca* ssp. *cardiac* (motherwort), *Matteuccia struthiopteris* var. *pensylvanica* (ostrich fern), *Desmodium glutinosum* (pointed-leaved tick-trefoil), *Rhus radicans* ssp. *rydbergii* (poison ivy - shrub form), *Ribes cynosbati* (prickly gooseberry), *Quercus rubra* (red oak), *Equisetum hyemale* ssp. *affine* (scouring-rush), *Aquilegia canadensis* (wild columbine), *Osmorhiza claytonii* (woolly sweet cicely), and *Impatiens pallida* (yellow touch-me-not). These species are all considered to be uncommon in this study (percent coverage < 1 %) and were primarily found within only one trail use type. Growth-form analysis revealed that 15 of the 18 species found within 2.5m of the trail were forbs. Also, 12 of the 18 species were identified as exotic or invasive - 4 in hiking, 6 in multi-use, 1 in biking and 2 in equestrian trails.

Several species were found exclusively between 5m and 10m from the trail. These include; *Fagus grandifolia* (American beech), *Podophyllum peltatum* (May-apple), *Asclepias exaltata* (poke milkweed), *Pyrola elliptica* (shinleaf), *Dryopteris carthusiana* (spinulose wood fern) and *Prenanthes altissima* (tall wood lettuce). *Ranunculus abortivus* (kidney-leaved buttercup) and *Anemone acutiloba* (sharp-lobed hepatica) were found exclusively at 17m from the trail. These species were also considered to be uncommon in this study (percent coverage < 1 %) and were primarily found within only one trail use type. Again, most of these species (6 of 8) were identified as exotic. Interestingly, no species were found exclusively at 25m from the trail. These results suggested that a significant portion of uncommon species exist within 2.5m of the trail edge. The next section quantifies the influence of trail impacts using the Shannon Index.

### 5.2 Biodiversity Assessment of Trail Use Types and Baseline Areas

The Shannon Index, referred to as a Shannon score hereafter, is a measure of biodiversity that equates richness and evenness into a single score. Communities which contain high species richness and high evenness values provide a high Shannon score. Trail-influenced environments were expected to be more diverse relative to baseline areas congruent with the IDH. Connell (1978) theorized that competitive species would dominate at low levels of disturbance and would be replaced by disturbance-tolerant species at high levels. At intermediate levels of disturbance however, large numbers of both communities could theoretically coexist resulting in higher diversity compared to control/baseline areas.

Trail use types and baseline areas were first compared and then the influence of distance was examined within and between each group. Shannon scores were determined for each of the six quadrants (0m, 2.5m, 5m, 10m, 17.5m and 25m) within each transect ($n = 10$) for all trail use types and baseline areas (Appendix B). To provide a single value of the Shannon score for a user group, the mean of all six quadrants was determined for each transect. The mean ± SD of the ten transects were then determined and were as follows: $1.28 \pm 0.36$ for baseline areas, $0.98 \pm 0.27$ for hiking trails, $1.01 \pm 0.21$ for multi-use trails, $1.07 \pm 0.20$ for biking trails, and $0.88 \pm 0.19$ for equestrian trails. The mean Shannon scores were then compared among each group using a one-way ANOVA test. Overall, there was a significant difference among the individual trail use types and baseline areas ($p = 0.0175$). According to the Tukey (HSD) post-hoc test, baseline areas had a significantly higher mean Shannon score than equestrian trails ($p < 0.01$). Furthermore, multi-use trails and hiking trails were marginally different than baseline areas ($p > 0.05$ but $< 0.1$). Shannon scores in biking trails did not differ from baseline areas ($p > 0.05$). The mean Shannon scores given above indicated that levels of biodiversity differ between trail-influenced environments and baseline areas. Also, trail-influenced environments were found to be on average 26 % less diverse, which was contrary to expectations. To determine if trail use types affected biodiversity on a spatial scale, the Shannon scores were next compared at various distances from the trail (0m, 2.5m, 5m, 10m, 17.5m and 25m).

The spatial variation of biodiversity between the trail use types and baseline areas was examined. Trail-influenced environments were anticipated to display a gradient of change with the highest diversity observable nearest to the trail. This was hypothesized because trailside vegetation is thought to be both exposed to trail-related disturbance and open to sources of immigration which fits Connell's Intermediate Disturbance Hypothesis. Within individual trail use types, a one-way ANOVA was used to compare whether Shannon scores were significantly different at each distance measured. Table 5 below summarizes the mean Shannon scores for each trail use type at various distances (0m, 2.5m, 5m, 10m, 17.5m and 25m) from the trail. Overall, there was a statistically significant difference within biking trails ($p = 0.0105$). In particular, the Tukey (HSD) post-hoc test revealed that there were significant differences in values between 2.5m and 25m and at 5m and 25m ($p < 0.05$ for both pairs). The data suggest that biodiversity in biking trails were significantly influenced by distance from the trail.

**Table 5. Shannon Scores at Various Distances from Trail Use Types.**

| Trail Type | Shannon Scores at Each Distance from Trail (Mean ± SD) | | | | | |
|---|---|---|---|---|---|---|
| | 0m | 2.5m | 5m | 10m | 17.5m | 25m |
| **Hiking** | 0.96 ± 0.43 | 1.00 ± 0.39 | 0.97 ± 0.48 | 0.92 ± 0.35 | 1.02 ± 0.35 | 1.04 ± 0.40 |
| **Multi-Use** | 1.12 ± 0.52 | 0.98 ± 0.35 | 1.05 ± 0.48 | 1.15 ± 0.27 | 1.02 ± 0.32 | 0.75 ± 0.47 |
| **Biking** | 1.00 ± 0.43 | 1.29 ± 0.29 | 1.29 ± 0.23 | 1.18 ± 0.39 | 0.95 ± 0.42 | 0.72 ± 0.51 |
| **Equestrian** | 1.06 ± 0.39 | 0.88 ± 0.39 | 0.99 ± 0.35 | 0.82 ± 0.38 | 0.61 ± 0.43 | 0.90 ± 0.52 |
| **Baseline** | 1.29 ± 0.32 | 1.19 ± 0.34 | 1.26 ± 0.51 | 1.30 ± 0.31 | 1.36 ± 0.52 | 1.26 ± 0.55 |

Number of transects = 10. Significant p-values are given in the text. In baseline areas, distances were measured from a randomly selected reference point as described in the Methodology (Section 4.7).

The slope (from 0m to 25m), correlation coefficient (Pearson r) and p-values for each relationship were next determined (Table 6). As expected, the correlation coefficient for baseline areas and distance was not significantly different than zero ($p > 0.05$). Hiking, multi-use, biking trails and equestrian trails also were not significantly different than zero indicating that distance did not affect the Shannon scores within these user types.

**Table 6. Relationship between Shannon Scores and Distance (from 0m to 25m).**

| Trail Type | Slope | Correlation Coefficient (Pearson r) |
|---|---|---|
| **Hiking** | 0.0028 | 0.6126 |
| **Multi-Use** | -0.0105 | -0.7133 |
| **Biking** | -0.0178 | -0.7635 |
| **Equestrian** | -0.0087 | -0.5396 |
| **Baseline** | 0.0018 | 0.3159 |

Slope, correlation coefficients and p-values were calculated from 0m to 25m for all trail use types and baseline areas. The correlation coefficient for every group was found not to be significantly different than zero ($p > 0.05$).

Upon comparing the individual trail use types and baseline areas using the Shannon Index, a significant difference was observed between trail-influenced environments and baseline areas. Unexpectedly, trail-influenced environments exhibited higher diversity compared to baseline areas. Furthermore, although Shannon scores were not linearly correlated with distance from the trail, significant differences were found within biking trails as previously mentioned above. Since the Shannon Index takes both richness and evenness into account and differences were found among trail use types and baseline areas as well as at various distances from the trail, further analysis using these descriptive measures was warranted. The following section compares the species richness of baseline areas and trail-influenced environments through several methods of analysis.

**5.3 Species Richness within Trail Use Types and Baseline Areas**

Species richness is a straightforward count of the number of species present within a given area. It was expected that trail-influenced environments would have higher species richness relative to baseline areas congruent with the IDH. The total number of species recorded was summed for each individual trail use type and in baseline areas. Baseline areas (n = 48) were found to contain the highest number of species overall, followed by hiking (n = 39), multi-use (n = 36), equestrian (n = 32) and biking trails (n = 31). On average, trail use types contained 28 % fewer species than baseline areas suggesting that the presence of a trail can eliminate enough individuals to remove several species from an area. This reduction in species richness was further analyzed with regard to the indigeneity of species.

The number of species considered to be indigenous (found in baseline), exotic (found only in trail-influenced environments) and species considered regionally invasive were determined for each trail use type (Table 7). As evident from the table below, each trail use type contained approximately half of the indigenous species found in baseline areas. Furthermore, roughly a third of the species found in areas adjacent to trails were exotic or were considered to be invasive. This suggested that indigenous species were being replaced when under the influence of a trail but not to the extent that equals baseline richness (fewer individuals). Although each trail use type contained half the indigenous species of baseline, few species (n = 9) were not found in any trail-use types. This inferred that of the 240 quadrants sampled in trail-influenced environments (Appendix A), few indigenous species were removed entirely.

**Table 7. Number of Indigenous, Exotic and Invasive Species within Individual Groups and Baseline Areas.**

| | Trail Use Type and Baseline Area (Richness) | | | | |
|---|---|---|---|---|---|
| | **Hiking** | **Multi-Use** | **Biking** | **Equestrian** | **Baseline** |
| **Indigenous** | 23 | 22 | 24 | 21 | 44 |
| **Exotic** | 10 | 10 | 5 | 6 | - |
| **Invasive** | 6 | 4 | 2 | 5 | 4 |
| **Total** | 39 | 36 | 31 | 32 | 48 |

Indigenous refers to species found in baseline areas. Exotic refers to species found only in trail-influenced environments. Invasives are species deemed by the Toronto and Region Conservation Authority to be regionally invasive and are those given a sensitivity score of 0 in Appendix A.

To determine if the richness of trail use types differed, the data were arranged by summing the total number of species within each transect. The mean ± SD of ten transects were then determined for each trail use type and were as follows: 15.0 ± 4.5 for baseline areas, 10.3 ± 2.2 for hiking trails, 10.7 ± 2.3 for multi-use trails, 10.4 ± 2.9 for biking trails, and 9.7 ± 3.2 for equestrian trails. A one-way ANOVA test was then used to compare the groups. Overall, there was a statistically significant difference among individual trail use types and baseline areas ($p = 0.0028$). According to the Tukey (HSD) post-hoc test, the groups which were statistically different from each other were as follows: hiking vs. baseline ($p < 0.05$), multi-use vs. baseline ($p < 0.05$), biking vs. baseline ($p < 0.05$) and equestrian vs. baseline ($p < 0.01$). Interestingly, every trail use type had significantly fewer species compared with baseline areas, contrary to the IDH. Furthermore, there was little difference among trail use types suggesting that the influence trails had over species presence/absence was generally similar among the groups.

To evaluate whether trail use types influence species richness on a spatial scale, richness was also evaluated at various distances (0m, 2.5m, 5m, 10m, 17.5m and 25m) from the trail. It was expected that baseline areas would have consistent species richness throughout transects because of the absence of human disturbance. Alternatively, trail-influenced environments were anticipated to display a gradient of change with the highest diversity observable nearest to the trail. This was hypothesized because trailside vegetation is thought to be both exposed to direct impacts of trail use and open to sources of immigration which fits Connell's IDH. Furthermore, it was expected that the frequency of users venturing off the trail, causing direct disturbances, would decrease with distance from the trail. However, as is evident below, quantifying the spatial influence of species richness depends heavily on the arrangement of the data and the chosen method of analysis. Several methods of analysis were conducted to demonstrate how previous research efforts have failed to identify a spatial influence on richness, which may have deterred researchers from assessing richness beyond a few metres of the trail edge.

The first method of analysis conducted resembles that of previous researchers whereby the number of species found at different distances was compared within a relatively small area. To demonstrate this, a one-way ANOVA test was used to compare richness values among each distance measured within individual groups (Table 8). In this method of analysis, different transects were used as replicates allowing the same species to be accounted for more than once in different transects and at different distances. Richness values were not significantly different for each distance measured (0m, 2.5m, 5m, 10m, 17.5m and 25m) within each trail use type ($p >$

0.05). This method of analysis suggested that the number of species within a single transect does not typically change with increasing increments of distance from the trail and does not reflect how richness of a community is influenced by the trail.

**Table 8. Species Richness at Various Distances from the Trail (not accounting for species overlap).**

| Trail Type | Richness at each Distance from Trail (Mean ± SD) | | | | | |
|---|---|---|---|---|---|---|
| | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Hiking** | 4.4 ± 1.7 | 4.2 ± 1.5 | 4.4 ± 2.0 | 4 ± 1.6 | 4.2 ± 1.5 | 4.4 ± 1.6 |
| **Multi-Use** | 4.7 ± 2.2 | 3.6 ± 1.1 | 3.8 ± 1.6 | 3.9 ± 1.0 | 3.6 ± 1.3 | 2.8 ± 1.5 |
| **Biking** | 3.5 ± 1.4 | 4.6 ± 1.4 | 4.5 ± 0.8 | 4.2 ± 1.8 | 3.6 ± 1.4 | 2.8 ± 1.3 |
| **Equestrian** | 4.1 ± 1.3 | 3.6 ± 0.8 | 3.6 ± 1.2 | 2.9 ± 1.0 | 2.5 ± 1.1 | 3.3 ± 1.5 |
| **Baseline** | 5.4 ± 2.0 | 4.9 ± 1.4 | 5.6 ± 2.2 | 5.2 ± 2.0 | 5.7 ± 2.1 | 5.5 ± 3.0 |

Number of transects = 10. In baseline areas, distances were measured from a randomly selected reference point as described in the methodology (Section 4.7).

In attempt to quantify the spatial influence on a community scale, the number of species found in separate replicates was summed for each distance without accounting for the same species twice (Table 9). These values differ from the ones in the analysis given above because they did not account for species which were present in separate transects but were found at the same distance. Such analysis provided a better representation of the spatial influence of a trail by quantifying species richness over a larger area. As was evident in Table 9, baseline areas had the highest species richness at every distance from the trail relative to the user types. Specifically, multi-use and equestrian trails had similar species richness and their values were higher than those of hiking and biking closest to the trails (0m - 2.5m). In general, species richness was observed to be higher at the trail edge (Table 9).

Since single richness values within a group could not be analyzed due to the lack of replicates, richness values of all trail use types (excluding baseline) were pooled by their corresponding distances (Table 9). Richness values were then compared among distances using a one-way ANOVA test. Overall, there was a significant difference among the distances ($p = 0.0122$). Specifically, richness values directly adjacent to the trail (0m) were significantly higher than at 17.5m ($p < 0.05$) and 25m ($p < 0.01$) from the trail according to the Tukey (HSD) post-hoc test. On average, species richness at 25m in trail-influenced environments were 49 percent lower than average baseline values and 35 percent lower than areas directly adjacent to the trail (0m). Averaged species richness values at 17.5m from the trail were 30.5 percent lower than areas directly adjacent to the trail (0m). These results indicated that significant differences in richness could be observed at greater distances than typically studied if the same species in separate replicates was not accounted for more than once.

The same values were next examined using correlation analysis. The individual group's slope (from 0m to 25m) and correlation coefficients (Pearson r) can be found in Table 10. The relationships between distance and richness were not found to be significant ($p > 0.05$) or to be linearly correlated. However, as mentioned above, significant differences in richness did exist between areas adjacent to the trail and the furthest distances measured (17.5m and 25m).

**Table 9. Total Species Richness at Various Distances from the Trail (accounting for species overlap).**

| Trail Type | Richness at each Distance from Trail | | | | | |
|---|---|---|---|---|---|---|
| | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Hiking** | 18 | 16 | 19 | 13 | 17 | 13 |
| **Multi-Use** | 24 | 14 | 13 | 17 | 12 | 12 |
| **Biking** | 16 | 13 | 18 | 16 | 15 | 13 |
| **Equestrian** | 20 | 15 | 14 | 16 | 11 | 13 |
| **Baseline** | 25 | 25 | 24 | 24 | 29 | 23 |

The number of species found at the respective distances was summed for each trail use type without accounting for the same species in different replicates. In baseline areas, distances were measured from a randomly selected reference point as described in the Methodology (Section 4.7.).

**Table 10. Relationship Between Richness and Distance (from 0m to 25m) (accounting for species overlap).**

| Trail Type | Slope | Correlation Coefficient (Pearson r) |
|---|---|---|
| **Hiking** | -0.157 | -0.60 |
| **Multi-Use** | -0.303 | -0.628 |
| **Biking** | -0.086 | -0.429 |
| **Equestrian** | -0.227 | -0.713 |
| **Baseline** | 0.011 | 0.05 |

Slope, correlation coefficients and p-values were calculated from 0m to 25m for all trail use types and baseline areas. The correlation coefficients of all groups were found not to be significantly different than zero ($p > 0.05$).

In summary, species richness was significantly higher in baseline areas compared to every trail use type, and at all distances, which is contrary to the IDH. Interestingly, trail use types did not differ among each other. Consistently within trail-influenced environments, roughly one third of the species identified were not found in baseline areas and were considered to be exotic. Although species richness was not linearly correlated with distance from the trail, significant differences were found between 0m vs. 17.5m and 0m vs. and 25m when accounting for species only once at each distance. The next section describes the percent coverage of trail use types as well as their spatial relationship with the trail.

### 5.4 Vegetation Percent Coverage of Trail Use Types and Baseline Areas

The percent coverage of vegetation was also assessed to further evaluate the influence of trail use types on adjacent floral composition. The percent coverage was determined by taking the mean of the six quadrants (0m, 2.5m, 5m, 10m, 17.5m and 25m) within each transect ($n = 10$) (Appendix D). The mean ± 1 SD of the ten transects were then determined and were as follows: 56.6 ± 11.3 % for baseline areas, 55.0 ± 15.9 % for hiking trails, 42.8 ± 14.4 % for multi-use trails, 32.3 ± 12.9 % for equestrian trails and 25.7 ± 11.3 % for biking trails. The mean percent coverage's were then compared among each group using a one-way ANOVA test. Overall, there was a statistically significant difference in percent coverage among different groups ($p < 0.0001$). In particular, the groups which were statistically different from each other according to the Tukey (HSD) post-hoc test were: hiking and biking trails ($p < 0.001$), hiking and equestrian trails ($p < 0.01$), multi-use and biking trails ($p < 0.05$), biking trails and baseline areas ($p < 0.001$), and equestrian trails and baseline areas ($p < 0.01$). The data overall indicated that equestrian and biking trails exhibit a significantly lower percent coverage compared to baseline areas and hiking trails. The relationship between percent coverage and distance was next examined.

Similar to species richness, trail-influenced environments were expected to display a gradient of change with respect to their percent coverage. Table 11 summarizes the mean percent coverage for each trail use type at various distances (0m, 2.5m, 5m, 10m, 17.5m and 25m) from the trail. A one-way ANOVA test was used to compare whether the mean percent coverage was significantly different among each distance measured within individual groups. In addition, the slope (from 0m to 25m), correlation coefficient (Pearson r) and p-values for each relationship was determined (Table 12). There was no significant difference between percent coverage at each distance measured within each individual group, including baseline areas (one-way ANOVA test, $p > 0.05$). The correlation coefficient (Pearson r) for multi-use trails was, however, significantly different than zero ($p < 0.05$). Within this trail type however, distances were not found to be significantly different from each other using the ANOVA test. As a result, distance was not considered to influence percent coverage in multi-use trails overall.

**Table 11. Percent Coverage at Various Distances from Trail Use Types.**

| Trail Type | Percent Coverage at Each Distance from Trail (Mean ± SD) | | | | | |
|---|---|---|---|---|---|---|
| | 0m | 2.5m | 5m | 10m | 17.5m | 25m |
| **Hiking** | 43.0 ± 21.8 | 55.0 ± 29.7 | 66.2 ± 18.1 | 59.7 ± 24.8 | 51.5 ± 28.3 | 54.9 ± 30.6 |
| **Multi-Use** | 49.9 ± 19.8 | 52.0 ± 23.3 | 38.8 ± 18.2 | 41.5 ± 19.5 | 40.3 ± 21.4 | 33.7 ± 22.2 |
| **Biking** | 20.6 ± 16.6 | 35.8 ± 25.9 | 31.6 ± 13.0 | 31.2 ± 10.6 | 17.8 ± 7.9 | 17.7 ± 11.5 |
| **Equestrian** | 39.9 ± 18.1 | 38.2 ± 22.8 | 33.1 ± 14.1 | 25.3 ± 19.6 | 23.8 ± 24.1 | 33.2 ± 23.1 |
| **Baseline** | 54.7 ± 20.0 | 50.8 ± 13.7 | 58.7 ± 11.7 | 62.9 ± 15.8 | 65.8 ± 18.9 | 46.8 ± 18.0 |

Number of transects = 10. Significant p-values are given in the text. In baseline areas, distances were measured from a randomly selected reference point as described in the Methodology (Section 4.7).

**Table 12. Relationship Between Percent Coverage and Distance (from 0m to 25m).**

| Trail Type | Slope | Correlation Coefficient (Pearson r) |
|---|---|---|
| **Hiking** | 0.078 | 0.097 |
| **Multi-Use** | -0.595 | -0.823* |
| **Biking** | -0.505 | -0.608 |
| **Equestrian** | -0.377 | -0.553 |
| **Baseline** | -0.056 | -0.074 |

Slope, correlation coefficient and p-values were calculated from 0m to 25m for all trail use types and baseline areas. The * denotes a statistically significant correlation coefficient which is different than zero ($p < 0.05$). All other groups' correlation coefficients were not significantly different than zero ($p > 0.05$).

The percent coverage of indigenous, exotic, and invasive species was also determined for each trail use type and for baseline areas. Data were arranged by summing the coverage of similarly grouped species and are provided as a percentage of the total coverage within a single trail use type (Table 13). Since these values were totals of species coverage's converted into percentages, statistical analysis was not performed. Relative differences could be observed nonetheless. As expected, baseline areas contained the highest proportion of indigenous species. Trail-influenced environments typically contained a higher proportion of invasive species coverage and less indigenous species coverage than baseline areas. Exotic coverage was relatively similar between the groups. Interestingly, the proportion of invasive species was high in equestrian trail but did not appear to differ amongst hiking, multi-use and biking trails. Statistical comparisons of invasive species coverage among groups (sensitivity 0) can be found in Section 5.8.

**Table 13. Relative Percent Coverage of Indigenous, Exotic and Invasive within Individual Groups and Baseline Areas.**

| | Trail Use Type and Baseline Area (Relative Percent Coverage) | | | | |
|---|---|---|---|---|---|
| | **Hiking** | **Multi-Use** | **Biking** | **Equestrian** | **Baseline** |
| **Indigenous** | 92.9 | 88.9 | 93.3 | 77.6 | 98.7 |
| **Exotic** | 3.6 | 5.4 | 4.5 | 4.6 | 0.0 |
| **Invasive** | 3.5 | 5.7 | 2.1 | 17.8 | 1.3 |
| **Total** | 100.0 | 100.0 | 100.0 | 100.0 | 100.0 |

Indigenous refers to species found in baseline areas. Exotic refers to species found only in trail-influenced environments. Invasives are species deemed by the Toronto and Region Conservation Authority to be regionally invasive and are those given a sensitivity score of 0 in Appendix A.

In summary, the percent coverage of vegetation differed among individual trail use types and baseline areas. Baseline areas and hiking trails displayed higher percent coverage values compared to biking and equestrian trails. Contrary to the IDH, distance did not affect percent coverage of vegetation in each trail use type measured in this study. Relative differences in coverage were observed among trail use types when grouping species by their indigenous, exotic and invasive status. In particular, equestrian trails were found to have the highest proportion of invasive species cover. The subsequent section analyzes the evenness of species present in the different trail use types and baseline areas.

### 5.5 Evenness within Trail Use Types and Baseline Areas

Evenness is a measure of biodiversity which quantifies how equal the coverage of a community is by comparing the relative abundance of species present within a certain area. A community containing species with equal abundances is considered to be more diverse (high evenness) than a community dominated by few species. It was anticipated that under the influence of a trail, dominant species such as sugar maple (*Acer saccharum* ssp. *saccharum*) would decrease and the presence of the introduced species would increase. Thus, it was expected that the evenness of a community would increase congruent with the IDH.

In this study, evenness was first assessed by comparing trail use types and baseline areas, and then the influence of distance was examined within and between each group. Evenness values were determined for each of the six quadrants (0m, 2.5m, 5m, 10m, 17.5m and 25m) within each transect (n = 10) for all trail use types and baseline areas (Appendix F). The evenness scores were then averaged across the six quadrants (0m, 2.5m, 5m, 10m, 17.5m and 25m) within each transect (n = 10). To provide a single value of evenness for a user group, the mean ± SD of the ten transects were then determined and were as follows: 0.78 ± 0.09 for baseline areas, 0.71 ± 0.09 for hiking trails, 0.85 ± 0.02 for biking trails, 0.75 ± 0.09 for equestrian trails, and 0.82 ± 0.07 for multi-use trails.

The mean evenness scores were then compared among each group using a one-way ANOVA test. There was a statistically significant difference among individual trail use types and baseline areas in evenness scores ($p = 0.0018$). According to the Tukey (HSD) post-hoc test, the groups which were statistically different from each other were hiking vs. multi-use trails ($p < 0.05$) and hiking vs. biking trails ($p < 0.01$). This indicated that both multi-use and biking trails have significantly higher evenness values than hiking trails. It is important to note that no groups differed from baseline areas.

The relationship between evenness and distance was next examined within each individual trail use type and in baseline areas. It was expected that areas closest to the trail would exhibit higher evenness due to the assumption that areas intermediately disturbed contain the highest diversity. A one-way ANOVA test was used to compare whether the mean evenness values were significantly different among the distances (0m, 2.5m, 5m, 10m, 17.5m, and 25m) (Table 14). In addition, the slope (from 0m to 25m), correlation coefficient (Pearson r) and p-values for each relationship were determined. ANOVA results indicated that evenness values were not significantly different for every distance measured within each group ($p > 0.05$). Also, each group's correlation coefficient was not significantly different than zero ($p > 0.05$). In the latter analysis, the slopes were all close to zero and were not significantly different from each other ($p > 0.05$).

**Table 14. Evenness at Various Distances from Trail Use Types.**

| Trail Type | Evenness Values at each Distance from Trail (Mean ± SD) | | | | | |
|---|---|---|---|---|---|---|
| | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Hiking** | 0.67 ± 0.21 | 0.76 ± 0.10 | 0.67 ± 0.15 | 0.69 ± 0.11 | 0.76 ± 0.14 | 0.71 ± 0.26 |
| **Multi-Use** | 0.76 ± 0.15 | 0.78 ± 0.17 | 0.86 ± 0.11 | 0.87 ± 0.07 | 0.82 ± 0.09 | 0.87 ± 0.11 |
| **Biking** | 0.88 ± 0.10 | 0.87 ± 0.05 | 0.87 ± 0.10 | 0.85 ± 0.14 | 0.77 ± 0.19 | 0.83 ± 0.10 |
| **Equestrian** | 0.70 ± 0.23 | 0.64 ± 0.28 | 0.74 ± 0.25 | 0. 78 ± 0.25 | 0.72 ± 0.26 | 0.70 ± 0.31 |
| **Baseline** | 0.72 ± 0.24 | 0.70 ± 0.23 | 0.65 ± 0.24 | 0.74 ± 0.24 | 0.71 ± 0.24 | 0.70 ± 0.24 |

Number of transects = 10. P-values are given in the text. In baseline areas, distances were measured from a randomly selected reference point as described in the methodology (Section 4.7).

In summary, the results showed that species evenness in trail-influenced environments differs among individual trail use types but not relative to baseline areas. Also, there did not appear to be a relationship between distance and evenness contrary to expectations that disturbance causing higher evenness would be observed adjacent to the trail. The following section compares baseline areas and individual trail use types in terms of their growth-form.

## 5.6 Richness and Percent Coverage of Species Grouped by Growth-Form

Species were pooled by their growth-form within each trail use type to determine if trails influence the richness and coverage of species with similar phenotypes and growth-forms (Tables 15 and 16). The richness of sedges, shrubs and grasses within trail-influenced environments were slightly reduced which is consistent with our general finding that richness is reduced as a result of trail use. The richness of forbs varied among the trail use types and was greatly reduced in biking and multi-use trails. Interestingly, equestrian trails had the highest richness of forbs but also the lowest forb coverage compared with the other groups. Similarly, hiking trails had the least tree richness and the greatest tree coverage.

**Table 15. Number of Species with Similar Growth-Forms within Individual Groups and Baseline Areas.**

| | Trail Use Type and Baseline Area (Richness) | | | | |
|---|---|---|---|---|---|
| **Growth-Form** | **Baseline** | **Hiking** | **Multi-use** | **Biking** | **Equestrian** |
| **Forbs** | 27 | 23 | 25 | 17 | 17 |
| **Grasses** | 2 | 1 | 1 | 0 | 1 |
| **Trees** | 9 | 7 | 4 | 7 | 6 |
| **Shrubs** | 6 | 5 | 4 | 5 | 5 |
| **Sedges** | 4 | 3 | 2 | 2 | 3 |
| **Total** | 48 | 39 | 36 | 31 | 32 |

The numbers of species with similar attributes were summed within a trail use type.

**Table 16. Percent Coverage of Species with Similar Growth-Forms within Individual Groups and Baseline Areas.**

| | Trail Use Type and Baseline Area (Percent Coverage of the Total) | | | | |
|---|---|---|---|---|---|
| **Growth- Form** | **Baseline** | **Hiking** | **Multi-use** | **Biking** | **Equestrian** |
| **Forbs** | 14.62 | 12.32 | 32.78 | 12.27 | 8.23 |
| **Grasses** | 1.30 | 0.03 | 0.30 | 0.00 | 0.03 |
| **Trees** | 31.82 | 28.97 | 7.72 | 6.93 | 14.90 |
| **Shrubs** | 5.18 | 0.68 | 1.10 | 2.62 | 5.50 |
| **Sedges** | 3.70 | 13.05 | 0.80 | 3.97 | 3.58 |
| **Total** | 56.62 | 55.05 | 42.70 | 25.78 | 32.25 |

Raw percent coverage of species with similar attributes were summed within a trail use type.

In summary, the coverage of forbs generally increased in trail-influenced environments whereas tree coverage generally decreased. Forbs made up a significant portion of the total species richness but the same cannot be said for the amount of forb coverage which varied largely. Also, the relative coverage of trees was greater than the relative number of tree species. This seemed to follow the general finding that coverage varied much more widely than species richness. It should be noted that baseline areas contained the highest species richness of each growth-form. The following section compares baseline areas and individual trail use types in terms of species sensitivity.

### 5.7 Species Sensitivity Analysis Within Trail Use Types and Baseline Areas

Species sensitivity was next examined to investigate the effects of trail use types on adjacent floral communities. First, the proportion of species with specific sensitivity scores was established. Then, the percent coverage of species grouped by sensitivity was compared among individual trail use types and baseline areas. Additionally, the influence of distance was examined on the percent coverage of species with similar sensitivities.

The species recorded in this study were assigned sensitivity scores (0-5) as previously described in the Methodology (Table 1, Section 4.8). Species which benefit from disturbance and are considered to be regionally invasive were given a score of 0. Species which were neutral were given a score of 2 and those which were negatively influenced by disturbance were given a score of 5. It was expected that trail-influenced environments would have a higher proportion of tolerant species (i.e. sensitivity 0 and 1) compared to baseline areas. Furthermore, areas closest to the trail were expected to be dominated by tolerant species and devoid of highly sensitive species. The total number of species for each sensitivity score is presented in Table 17. A list of specific species and their sensitivity scores may be found in Appendix A. The number of species with different sensitivities was not even, with the largest number of species having sensitivities of 2 and 5. These data were taken from all groups (hiking, multi-use, biking, equestrian and baseline areas).

**Table 17. Distribution of Species with Different Sensitivity Scores.**

| Sensitivity Score | Number of Species Recorded | Percent of Total Number of Species (%) |
|---|---|---|
| 0 | 8 | 10.8 |
| 1 | 4 | 5.4 |
| 2 | 21 | 28.4 |
| 3 | 12 | 16.2 |
| 4 | 12 | 16.2 |
| 5 | 17 | 23.0 |
| Total | 74 | 100 |

The number of species recorded was summed and grouped by their sensitivity score. The total number of species of each group is presented as a percentage of all species.

The percent coverage of specific sensitivity scores was next compared among individual trail use types and baseline areas. Species with similar sensitivity scores were grouped together within each transect and their percent coverage was summed and calculated as a percentage of the total area sampled. The mean and standard deviation of the ten transects is presented in Table 18. These values were then compared among the groups using a one-way ANOVA test. Overall, there was a significant difference among groups with sensitivity scores of 2, 3 and 5. Groups with sensitivity scores of 0, 1 and 4 were not significantly different from each other ($p > 0.05$) and were not further analyzed. The specific results of the one-way ANOVA and the Tukey (HSD) post-hoc test for the other sensitivities (2, 3 and 5) are provided below, respectively.

**Table 18. Percent Coverage of Species Sensitivities in Trail-Influenced Environments and Baseline Areas.**

| Sensitivity Score | Percent Coverage in Trail Use Types and Baseline Areas (Mean % ± SD) | | | | |
|---|---|---|---|---|---|
| | **Hiking** | **Multi-Use** | **Biking** | **Equestrian** | **Baseline** |
| **0** | 1.88 ± 2.44 | 2.37 ± 2.06 | 0.43 ± 0.85 | 5.47 ± 11.35 | 0.70 ± 0.61 |
| **1** | 0.25 ± 0.26 | 0.17 ± 0.42 | 0 ± 0 | 0.07 ± 0.14 | 0.18 ± 0.36 |
| **2** | 27.55 ± 14.40 | 7.72 ± 9.21 | 9.32 ± 5.63 | 10.23 ± 5.28 | 24.80 ± 15.08 |
| **3** | 4.32 ± 4.07 | 15.52 ± 9.18 | 1.43 ± 1.48 | 9.70 ± 7.01 | 10.63 ± 6.96 |
| **4** | 14.35 ± 13.24 | 5.53 ± 7.37 | 4.82 ± 5.84 | 3.68 ± 8.22 | 6.97 ± 6.08 |
| **5** | 6.70 ± 3.67 | 11.43 ± 4.81 | 9.78 ± 3.40 | 3.10 ± 2.40 | 13.33 ± 8.85 |

Number of transects = 10. P-values are given in the text.

*Sensitivity 2:* The percent coverage of species with sensitivity scores of 2 were significantly different among individual trail use types and baseline areas ($p < 0.0001$). According to the Tukey (HSD) post-hoc test, hiking trails were significantly higher than multi-use ($p < 0.01$), biking ($p < 0.01$) and equestrian trails ($p < 0.01$). Similarly, baseline areas were also significantly higher than multi-use ($p < 0.01$), biking ($p < 0.05$) and equestrian trails ($p < 0.05$). Hiking trails and baseline areas were not different from each other ($p > 0.05$).

*Sensitivity 3:* The percent coverage of species with sensitivity scores of 3 were significantly different among individual groups ($p < 0.0001$). In particular, biking trails had a significantly lower percent coverage than equestrian trails ($p < 0.05$), multi-use trails ($p < 0.001$) and baseline areas ($p < 0.05$). Furthermore, hiking trails were also significantly lower than multi-use trails ($p < 0.01$).

*Sensitivity 5:* The percent coverage of species with sensitivity scores of 5 were significantly different among individual groups ($p = 0.0005$). According to the Tukey (HSD) post-hoc test, equestrian trails were significantly lower than multi-use trails ($p < 0.01$), biking trails ($p < 0.05$) and baseline areas ($p < 0.001$). Differences were also present between hiking trails and baseline areas, with hiking trails having less coverage of sensitivity 5 species (p <

0.05).

The influence of distance on the percent coverage of species with similar sensitivities was next examined within individual groups. A one-way ANOVA test was used to compare whether the mean percent coverage of a single sensitivity was significantly different for each distance measured (0m, 2.5m, 5m, 10m, 17.5m, and 25m). There was no significant difference among percent coverage at each distance measured for each sensitivity score within a group in all cases (one-way ANOVA test, $p > 0.05$). Appendix E summarizes the total percent coverage of species grouped by sensitivity at each distance measured.

In addition, the slope (from 0m to 25m), correlation coefficient (Pearson r) and p-values for each relationship were determined (Appendix E). Within all groups, the correlation coefficient was not significantly different from zero ($p > 0.05$) for each sensitivity, other than 4. Within multi-use trails, the percent coverage of species with sensitivity scores of 4 had a significant correlation coefficient (slope -3.449, $r = -0.854$, $p = 0.0304$). This was the only significant relationship found between distance and species sensitivity. However, since distance observations were not found to be significantly different from each other using the ANOVA test, distance was not considered to influence the coverage of species grouped by sensitivity overall.

In summary, the data indicated there were differences among the groups in terms of the percent coverage of various sensitivities. Interestingly, there were no differences among the groups for sensitivities 0 and 1 which are those which benefit from disturbance and are considered to be regionally invasive. On the contrary, multi-use trails had significantly lower coverage of sensitivity 2 species than baseline areas and hiking trails. Also, equestrian trails exhibited significantly lower sensitivity 5 coverage compared to baseline areas and the other trail use types. Contrary to the expectation that sensitive species would display a distance-decay effect, when species were grouped by their sensitivity scores, their percent coverage's were not influenced by distance. The next section further analyzes the presence of invasive species in trail-influenced environments and baseline areas.

### 5.8 Further Assessment of Invasive Species in Trail Use Types and Baseline Areas

Of the 74 species recorded in this field study, 8 were designated as regionally invasive by the TRCA. Invasive species were present in every trail use type as well as in baseline areas. Specifically, 6 were found in hiking, 4 in multi-use, 2 in biking, 5 in equestrian trails and 4 in baseline areas. In a previous section, invasive species (sensitivity 0) were found not to have a relationship with distance from the trail. This section attempted to analyze invasive species individually for their presence in each trail use type as well as spatially (0m to 25m). Prior to this study, it was anticipated that the percent coverage of invasive species would be greatest at the trail edge and decline with distance into the adjacent interior. It was also expected that baseline areas would have the least density of invasive species due to a lack of disturbance. Equestrian trails were expected to have the greatest invasive coverage due to the fact that horses feed in pastures where invasive species are prevalent and because horses defecate on trails.

The presence of each invasive species identified is described here and their abundances can be found in Appendix A. *Rhamnus cathartica* (common buckthorn) was found only in equestrian quadrants at 17m from the trail. *Veronica officinalis* (common speedwell) was found in baseline areas and hiking quadrants within 2.5m of the trail. *Taraxacum officinale* (dandelion) was found throughout the sampling area (0m to 25m) in baseline areas, hiking, multi-use and equestrian trails. *Alliaria petiolata* (garlic mustard) was found in every trail use type, occupying the most percent coverage in multi-use trails (2.17 %), followed by equestrian (0.93 %), hiking

(0.67 %) and biking trails (0.27 %). Garlic mustard was found at every distance measured from the trail. Interestingly however, this invasive species was not present in baseline areas. *Epipactis helleborine* (helleborine) was found in every trail use type and throughout all transects. Unexpectedly, the percent coverage of this species was actually greater in baseline areas compared to trail-influenced environments. *Galeopsis tetrahit* (hemp-nettle) was present only in multi-use quadrants, directly adjacent to the trail (0m). *Geranium robertianum* (herb Robert) was found in significant amounts in equestrian areas and at every distance measured. This species occupied enough coverage to make it a dominant species overall (Table 2). Herb Robert was also found in baseline areas but occupied a much lower percentage of coverage in these areas. *Leonurus cardiaca ssp. cardiaca* (motherwort) was found only in hiking quadrants, within 2.5m of the trail. Congruent with findings in the previous section, every invasive species correlation analysis was determined to be not significant ($p > 0.05$).

In summary, invasive species did not have an evident spatial relationship with distance from the trail as most were found either throughout transects or directly adjacent to the trail. The next section elucidates if certain characteristics of trails or environmental variables could be attributed to the variation in impact.

### 5.9 Contributions of Trail Characteristics and Environmental Factors

This section examines whether the impacts described above can be explained by certain trail characteristics or environmental variables using multiple regression. First, analysis of the relationships between trail characteristics and descriptors of vegetative composition is given. Following this, comparisons of collected trail characteristics and environmental variables are provided to determine if differences between the groups could be attributed to the influencers of floral composition.

**5.9.1 Multiple Regression**

Multiple regression analysis was used to examine the relationship between several environmental variables and the descriptors of vegetation composition (Shannon Index, species richness, percent coverage, evenness and species sensitivity). Only variables with p-values less than 0.15 were included in the final given models and variables with less than 0.05 were considered significant. The environmental variables which were found to influence vegetation composition included tread width, trail depth, transect slope, aspect and elevation. Trampled width of the trail and the slope of the trail were not found to be significant contributors to the models. The results of each respective descriptor are summarized below.

*Shannon Index:* Shannon scores used in the multiple regression were gathered by taking the median Shannon score of the six quadrants (0m, 2.5m, 5m, 10m, 17.5m and 25m) within each transect (n = 10), of each individual user group (Appendix B). None of the seven variables significantly contributed to the Shannon scores ($p > 0.05$).

*Species Richness:* The total number of species was determined for each transect within each individual user group (Table 8). According to the multiple regression, the model which best described species richness across all user groups was: [Species Richness] = 10.275 + 1.193*[Trail depth] + 0.741*[Transect slope]. The coefficient of determination ($r^2$) of this model was 0.2704. The p-values for each contributing variable were 0.003 for trail depth and 0.057 for transect slope.

*Percent Coverage of Vegetation:* The percent coverage used in the multiple regression was determined by taking the median percent coverage of the six quadrants (0m, 2.5m, 5m, 10m, 17.5m and 25m) within each transect (n = 10) (Appendix D). None of the variables significantly contributed to the percent coverage values ($p > 0.05$).

*Evenness:* Evenness values used in the multiple regression were determined by taking the median evenness of the six quadrants (0m, 2.5m, 5m, 10m, 17.5m and 25m) within each transect (n = 10) of each individual user group (Appendix F). The model which best describes evenness across all user groups was: [Evenness] = 0.796 + 0.033*[Aspect]. The coefficient of determination ($r^2$) of this model was 0.0978 and the p-value was 0.0494.

*Sensitivity Analysis:* Sensitivity scores were assigned to each species recorded as described in the Methodology and can be found in Appendix A. The percent coverage of species with similar sensitivity scores were then grouped within each transect (n = 10), for each individual user group (Appendix E). The percent coverage of species was found with the following equation, using the 0 sensitivity score as an example: (sum of percent coverage of 0 sensitivity species) / 600 * 100. The models which best described coverage across all user groups for sensitivity scores 0, 1 and 3 are summarized in Table 19. None of the variables significantly contributed to the sensitivity scores 2, 4 and 5 ($p > 0.05$).

**Table 19. Multiple Regression of Species Sensitivity Score, Floral Coverage and Environmental Variables.**

| Sensitivity Score | Multiple Regression Equation | Coefficient of determination ($r^2$) | p-value |
|---|---|---|---|
| 0 | [Sensitivity Cover] = 2.538 - 1.687* [Tread width] | 0.075 | 0.088* |
| 1 | [Sensitivity Cover] = 0.121 - 0.082* [Transect slope] | 0.087 | 0.064* |
| 3 | [Sensitivity Cover] = 7.742 + 3.112* [Elevation] | 0.138 | 0.018 |

The * denotes correlations which were marginally significant from zero ($0.05 < p < 0.1$).

### 5.9.2 Description of Various Trail Characteristics

Various environmental variables for each trail use type and for baseline areas were collected at each transect. The variables were selected based on the criteria outlined in the Methodology (Section 4.4) and include trampled width, tread width, trail depth, trail slope, transect slope, aspect and elevation. Table 20 summarizes the mean ± SD of each of these variables collected from ten transects (Appendix G). The aspect of each transect for these specific trail users was also determined and can be found in Appendix H.

**Table 20. Characteristics of Trail Use Types and Baseline Areas.**

| Trail Type | Variables (Mean ± SD) | | | | | |
|---|---|---|---|---|---|---|
| | **Trampled width (m)** | **Tread width (m)** | **Trail depth (in)** | **Trail Slope (°)** | **Transect Slope (°)** | **Elevation (m above sea level)** |
| **Hiking** | 1.55 ± 0.33 | 1.02 ± 0.10 | 2.80 ± 0.85 | 1.35 ± 1.06 | 4.20 ± 2.35 | 350 ± 4.16 |
| **Multi-Use** | 2.35 ± 0.11 | 1.66 ± 0.22 | 4.75 ± 0.92 | 2.00 ± 1.25 | 6.1 ± 3.73 | 350 ± 2.57 |
| **Biking** | 2.33 ± 1.12 | 1.57 ± 0.70 | 3.45 ± 1.82 | 3.80 ± 2.97 | 4.50 ± 3.03 | 355 ± 5.08 |
| **Equestrian** | 1.87 ± 0.74 | 1.47 ± 0.67 | 5.40 ± 3.66 | 3.50 ± 2.87 | 4.40 ± 2.88 | 363 ± 9.49 |
| **Baseline** | n/a | n/a | n/a | n/a | 4.00 ± 2.31 | 342 ± 30.50 |

Number of transects = 10. P-values are given in the text.

Each specific environmental variable was next compared between trail use types and baseline areas (where applicable) using a one-way ANOVA test. For trampled width, there was an overall significant difference among trail use types ($p = 0.0384$). Using the Tukey (HSD) post-hoc test, the groups which were marginally significantly different from each other were hiking and multi-use trails as well as hiking and biking trails ($p > 0.05$ but $< 0.1$ for both groups). There was also an overall significant difference among trail use types in terms of mean tread width ($p = 0.0333$). In particular, multi-use trails had a significantly larger mean tread width than hiking trails (Tukey (HSD) post-hoc test, ($p < 0.05$)). For elevation, there was an overall significant difference between trail use types and baseline areas ($p = 0.0332$). Using the Tukey (HSD) post-hoc test, the groups which were significantly different from each other were equestrian trails and baseline areas ($p < 0.05$). Other environmental variables including trail depth, trail slope, aspect and transect slopes were found not to differ among the trail use types and baseline areas ($p > 0.05$). The data suggested that tread width and trampled width were the only trail characteristics to differ among the groups. Furthermore, hiking trails had the least influence on trail characteristics compared to other use types.

In summary, regression modeling revealed that trail depth and transect slope positively influenced species richness. Similarly, a south facing aspect was found to positively influence evenness in trail-influenced environments. As previously mentioned however, trail use types were not found to differ in terms of these collected trail characteristics. Thus, trail depth, transect slope and a south facing aspect were not able to explain variability in trail-influenced environments in terms of species richness and evenness. Regression modeling also revealed that transect slope negatively influenced coverage of tolerant species (sensitivity 1). Again, differences in transect slope were not observed between the groups and were not considered to have explanatory value for comparing differences in coverage of sensitivity 1 species between the user groups. Furthermore, tread width negatively influenced coverage of invasive species (sensitivity 0) and differences in tread width were observed between the trail use types. Both relationships were only marginally significant and were not considered to have explanatory value in this study. Lastly, elevation was positively correlated with increases in coverage of sensitivity 3 species. However, this could only account for roughly 16 percent of all species and was likely not representative of how species in this vegetation type generally correlated with elevation.

### 5.10 Summary of Key Results

Contrary to expectations, trail-influenced environments were found to be approximately 27 % less diverse than baseline areas. Furthermore, roughly one third of the species identified in trail use types were exotic species (not found in baseline) or were considered regionally invasive by the TRCA. Although the average coverage of invasive species did not differ between trail-influenced environments and baseline areas, equestrian trails had higher proportional coverage of invasive species compared to the other use types. Conversely, the richness of indigenous species was strikingly similar among the groups. This was further supported by our finding that species richness and Shannon scores were not different among the groups. Analysis of species by growth-form confirmed our other findings that diversity was reduced in trail-influenced environments and also suggested that trees may be sensitive to trail-related disturbances.

Although the diversity of trail use types did not differ among trail-influenced environments, differences were pronounced between the trail edge and areas furthest from the trail, at 17.5m and 25m. Richness, percent coverage, evenness and Shannon scores were generally not linearly correlated with distance from the trail, however, spatial effects existed nonetheless. Specifically, areas directly adjacent to the trail exhibited higher richness than areas furthest from the trail which consistently experienced reductions in richness upwards of 50 percent compared with baseline areas. The coverage of invasive species was also not overall linearly correlated with distance. Invasive species coverage did however, exhibit spatial effects as most invasive species were found throughout transects or exclusively at the trail edge.

Regression modeling overall revealed that trail depth, transect slope and a south facing aspect positively influenced species diversity or components thereof. However, trail use types were not found to differ in terms of these collected trail characteristics and thus, cannot be used to explain variation in richness and evenness in trail-influenced environments. Despite this, tread and trampled width of multi-use and biking trails were in some cases found to be twice as wide as hiking trails. Although trail width had no influence over vegetation beyond the trail edge, it is a significant finding for trail degradation management.

# Chapter 6: Discussion

## 6.1 Overview

Three principal findings emerged from this study. Firstly, managers can mitigate changes to species richness and prevent shifts in the dominance of communities by choosing routes with high side-slopes perpendicular to the trail and by avoiding areas with south facing aspects. In contrast, trail width likely has no influence on species diversity and trail depth may only act as an indicator of trail impacts occurring to vegetation. Thus, management actions to mitigate these trail impacts likely have little direct influence on composition and the preservation of indigenous species. Secondly, trail-influenced environments experienced significant reductions in species richness at distances beyond the influence of an edge effect. This is thought to be caused by infrequent off-trail trampling which may proliferate throughout forest stands over extended periods of trail use. Thirdly, recreational trail use types do not disproportionately cause greater disturbance influencing indigenous species or cause greater exotic and invasive species coverage despite some groups having greater dispersal potential. This section compares the abovementioned findings with those in the literature and describes their implications with respect to trail management. In addition, this section discusses the limitations of this study and gives recommendations for future research.

## 6.2 Contributors to Changes in Diversity and Composition

Previous research of trail degradation processes has led to significant improvements in sustainable trail design (Marion and Leung, 2004). In particular, land managers are able to use various methods to design and construct trails to limit their degradation. To date, however, little is known about how trail characteristics and topographical features influence vegetative communities and their diversity. Improved understanding regarding the influences of diversity and composition permit the selection of more effective trail management actions, and can be used to justify difficult decisions pertaining to the permittance of certain use types. In addition, it may also provide a means for mitigating trail impacts, where achievable. Multiple regression analysis was used in this thesis to elucidate trail influences on floral composition. Transect slope, trail depth, and a south facing aspect were found to significantly influence vegetation composition and are discussed in greater detail below.

### 6.2.1 The Influence of Transect Slope and Trail Depth

Regression analysis revealed an interesting relationship between species richness and two trail characteristics, trail depth and transect slope (Section 5.9.1). Drawing from the environmental models, it is evident that these covariates were positively correlated with richness and may be an indication of less direct trampling occurring off the trail. Conceivably, trail users are less likely to venture up steep grades compared to evenly graded areas. Areas with steep slopes adjacent to the trail may confine users to its tread and limit the amount of off-trail trampling which consequently prevents diversity loss. Bayfield (1971, 1973) similarly demonstrated that steeper trailside terrain effectively limits the tendency for hikers to stray from the trail. Therefore, trail managers may be able to prevent off-trail trampling from occurring by choosing routes that traverse areas with high side-slopes perpendicular to the trail.

Aside from preventing impacts to species richness, choosing areas with high side-slopes may also prevent damage to the trail. For example, trails on flat terrain are quickly incised due to soil compaction and erosion. Trails with low grades typically also collect water and drain slowly which causes trails to be muddy and promotes trail widening (Calais and Kirkpatrick, 1986; Marion, 1994). Conversely, trails with sloping terrain make it possible to remove water from the tread easily and trail building manuals often suggest avoiding flat terrain to minimize these problems (Birchard and Proudman, 2000; Marion and Wimpey, 2007). It is important to note that transect and trail slopes in different use types were not found to differ from each other (Section 5.9.2). Hence, these environmental variables did not disproportionately cause bias between trail use types when comparing differences in composition and diversity and thus may be attributed to the users' impact potential.

Trail depth may also generally indicate that less off-trail trampling is occurring as frequency of use is well known to be positively correlated with trail depth (Dale and Weaver, 1974). Increased use of a trail may thus suggest that adjacent communities are less frequently trampled. As was seen with transect slope, species richness was positively influenced by increased trail depth (Table 19, Section 5.9.1). Therefore, it may be assumed that areas exposed to infrequent off-trail trampling have deep trail depths and consequently have higher species richness due to a lack of direct trampling. It is important to note that this may not be true for every situation as areas incurring greater off-trail trampling may conceivably experience significant trail degradation from a multitude of environmental factors (Section 2.3.3). Trail managers should also not aim to increase trail depth due to the many problems associated with erosion (Section 2.2). Furthermore, trail managers should not assume that trails experiencing higher frequencies of use will always exhibit greater loss of diversity. It does mean, however, that trail features may be able to physically prevent users from creating impacts to vegetation or be used to indicate a level of impact.

### 6.2.2 The Influence of Light and Tread Width

According to the multiple regression analysis, evenness was found to positively increase as a result of aspect (Section 5.9.1). Specifically, areas with a south facing aspect, which equate with greater light, were less dominant. This may have occurred as result of both the reduction of tree species in trampled areas as well as increased light facilitating the establishment of introduced species. As previously mentioned, areas experiencing off-trail trampling consequently were reduced in tree coverage which were replaced by forbs. Since forbs and other introduced species often require increased light exposure to facilitate establishment during early growth, areas experiencing increased light may be predisposed to higher competition from forbs (Miller, 1990). This would consequently result in lower dominance but overall increased evenness, which was observed (Section 5.5). Dale and Weaver (1974) similarly found that more sunlight favors many introduced species and accounts for higher diversity than areas receiving less light. Thus, choosing areas with north facing aspects may hinder the establishment of introduced species such as forbs and be a valuable measure for preventing changes in dominance.

Since wider trails were anticipated to have greater gaps in canopy and consequently increased light exposure, trail width was expected to be positively correlated with coverage and richness (Dale and Weaver, 1974). However, tread width did not significantly influence either of these descriptors of composition. This may have occurred due to a multitude of factors including, tree height, tree species and stage of succession which may have also influenced the amount of light that vegetation received. Although increased trail width did not result in higher coverage or species diversity, it does not exclude the possibility that areas next to the trail may experience increased light exposure. Others have previously found that trails had greater gaps in canopy and experienced increased light exposure (Dale and Weaver, 1974). The results of this study suggest that management actions to limit trail width less than 2.5m will not change how diversity is influenced. It should be noted that trail width was negatively correlated with sensitivity 0 species coverage (invasives), however, this relationship was only marginally significant and likely was not representative of how vegetation coverage is influenced by trail width.

Overall, these findings may hold significant potential for the planning of trail systems, especially when considered with the application of geographic information system (GIS) technology. Specifically, detailed terrain mapping could be used to locate high side-slopes and north facing aspects. As previously mentioned, choosing routes with these attributes would have the least influence on species diversity and composition. Moreover, areas with deep trails may be indicative of less off-trail trampling occurring but should not be a management objective. Lastly, limiting trail width is likely not an effective method for mitigating changes to composition.

### 6.3 Trail Use Types, Trail Degradation and the Influence on Composition

When considering the management of recreation areas, the question of whether certain activities cause disproportionately greater levels of impact than others is often asked. Experimental studies have previously shown that trail use types differ in their potential to degrade trails, however, only a limited number have compared impact characteristics of different trail use types after prolonged use and on vegetation. Few studies of this type exist likely due to the difficulty of locating areas with comparable impacts. As previously discussed in the methodology, comparisons of trail conditions among the groups were possible in this thesis because of the ability to isolate single use trails with similar frequencies within the same vegetation type.

Another reason why comparative studies have been less common is due to the difficulty in separating impacts that occur as a result of trail wear and those that occur due to other environmental variables. Characteristics of topography have been well documented and several studies have reported a strong positive relationship between trail slopes and soil loss (Leung and

Marion, 1996; Cole and Bayfield, 1993). In particular, increased erosion on a trail is primarily due to the higher velocity of runoff on steep slopes. In this study, trail depth and trail width were used as indicators of the amount of soil loss caused by recreationalists. While it may be presumptuous to say that we isolated trail use impacts from all variables, we were able to determine that trail slope and transect slope did not differ between the groups (Section 5.9.2). Consequently, erosion processes also did not influence trail use types differently in this regard. Thus, it is possible to make comparisons among the groups assuming that specific trail characteristics are a result of the predominant user. The subsequent section discusses the differences between trail use types and how species composition is influenced by two particular trail characteristics, trail depth and trail width, respectively.

### 6.3.1 Trail Depth

In anticipation that equestrian trails would receive greater trail compaction due to the weight of a horse, equestrian trails were expected to have a greater trail depth compared to hiking and biking trails (Cole and Spildie, 1998; Thurston and Reader, 2001; Wilson and Seney, 1994). In particular, Thurston and Reader, (2001) demonstrated that equestrian trails are relatively deeper than hiking trails as horses depress soil greater than that of hikers. Our results confirmed that hiking and mountain biking trails have similar impacts on soils (Section 5.9.2), congruent with previous research (Wilson and Seney, 1994). In addition, equestrian and multi-use trails did not cause greater trail depth than the other groups, which was unexpected due to the weight of horses (Section 5.9.2). A possible explanation for this may be due to the greater length of time that trails were exposed to these particular uses.

In the literature, erosion has been positively correlated with trail depth and has a curvilinear relationship with the frequency of use (Kuss, 1987). At first, increases in trail depth may be pronounced by a limited number of users whereas over the long term, increases in frequency may have negligible influence on its depth. Kuss (1987) demonstrated that trail depth increased by approximately 34mm following 600 hiker passes. As use intensity increased beyond 500/600 passes, less change was observed in trail depth. Similarly, Weaver and Dale (1978) found that trail depth increased with use up to roughly 1000 passes but did not change after this threshold. The trails in our study had frequencies of roughly 1000 passes a year and have experienced trail use for more than 20 years. The similar trail depths between the user groups observed in this study may thus be explained by their extensive use.

Another observation identified in this study was the positive correlation between trail depth and species richness (Section 5.9.1). As previously discussed, areas with increased trail depth may indicate that less off-trail trampling is occurring. However, as shown above, no differences were observed between the trail use types in terms of depth. The variability in trail-influenced environments in terms of richness is thus not likely related to impacts occurring to trail soils and erosion on the trail. This is further supported by our findings that native species richness, which is thought to be sensitive to trampling intensity, was strikingly similar between the groups (Table 7, Section 5.3). Moreover, these results infer that impacts to trail soils did not disproportionately differ among trail use types.

### 6.3.2 Trail Width

Another area of debate among trail managers and recreationalists concerns the potential for specific use types to cause greater trail widening than others. Similar to trail depth, this characteristic has received much attention from both trail managers and the scientific community because greater trail width makes soils more available to erosive forces. In our study, biking and multi-use trails were approximately 1.5x wider than hiking trails suggesting that the former have greater influence over trail width (Section 5.9.2). Consistent with these results, biking trails were shown to be twice as wide as walking paths in level woodland (Dale and Weaver, 1978). Since multi-use trails also incurred influences of biking, we can assume that the increased trail width of both trails was predominantly due to mountain bike use and not by other use types. It is often assumed that biking has the least influence over trail width because of the narrow width of a bike tire. However, bikers often come into contact with vegetation at the trail edge because of the need to take wide turns and maneuver around rough terrain on the path such as rocks. These actions undoubtedly result in further widening of the respective trails and may explain their greater variability in terms of width (Table 20, Section 5.9.2).

An additional misconception that trail managers often have is that equestrian riders cause significantly greater trail widening compared to other use types due to the size of the horse. However, our results demonstrated that equestrian trails were not significantly different from hiking trails, or any other trail use type (Section 5.9.2). This may be attributed to the fact that horse riding groups tend to travel in a single line despite their wide bodies (Torn *et al.*, 2009; Dale and Weaver 1974). Since equestrian users were present on multi-use trails but the latter were not significantly wider than other groups, the notion that mountain bike users are the predominant influencers of trail width is further supported. Overall, our findings reinforce results from previous research that certain trail use types differ in their influence over trail width (Dale and Weaver, 1978). However, after extended periods of trail use, mountain biking caused the greatest amount of trail degradation not equestrian use. This is interesting because it suggests that the user which conceivably require the least amount of space, created the widest trails. Therefore, the causal factor in the potential to cause greater damage to vegetation at the trail edge is not the size of the user so much as a user's behavior and method of travel.

Despite the significance of these findings and their potential for trail degradation and erosion management, multiple regression analysis revealed that trail width does not significantly influence species richness or coverage (Section 5.9.1). Thus, actions to manage trail width are likely limited to preventing erosion and are not directly influential of floral composition. This may explain why differences were observed in trail characteristics among groups but were also not different in terms of species richness or evenness. Conversely, variability in the trail-influenced environments cannot be explained by trail depth, which positively correlated with richness, because differences in trail depth did not exist between the groups. As previously mentioned, this may have been caused by the prolonged use of the trails.

### 6.4 Growth-Form

Analysis of species richness and coverage by growth-form revealed several significant findings related to compositional changes and spatial patterns of disturbance frequency. Specifically, the results revealed that the presence of forbs generally increased in areas directly adjacent to the trail whereas tree coverage decreased under intensified trampling. Due to the concentrated nature of trail use, adjacent areas were assumed to experience the highest frequencies of off-trail trampling which would decrease with distance from its edge. Interestingly, however, areas within 2.5m of the trail were occupied by high numbers of uncommon forb species (Section 5.1). Most of these species were found only within trail-influenced environments and were thus considered exotic (Section 5.1).

#### 6.4.1 Forbs

The increase in forb richness and establishment in disturbed areas has likely occurred for two reasons. Firstly, forbs are able to grow quickly and their fast recovery from physical alteration makes them competitive in occupying areas where there is frequent disturbance (Sun and Liddle, 1993 a,b). Secondly, forbs may be able to withstand increased trampling frequency due to their small size. It is conceivable that small plants such as forbs may be exposed to direct contact less frequently than larger species which may increase their survival likelihood. The results overall suggest that in areas of disturbance, native composition may be replaced by forbs. With respect to trail disturbance in particular, there is a high likelihood that many of these forbs are exotic. Several other studies have also shown that forb richness increases under disturbance which supports this hypothesis (Halpern and Spies, 1995; Jenkins and Parker, 1999; Roberts and Zhu, 2002; Schumann *et al.,* 2003).

**6.4.2 Trees**

The second significant finding related to species richness and coverage by growth-form, specifically, tree coverage decreased by 50% on average in disturbed areas (Table 16, Section 5.6). In particular, sugar maple (*Acer saccharum* ssp. *Saccharum)* dominated trail-influenced environments and was positively correlated with increasing distance from the trail (Tables 2 and 4, Section 5.1). The results strongly suggest that the greatest trampling of vegetation occurs near the trail edge, as expected. They also suggest that trees are sensitive to trampling and less adept to recover from damage compared to other growth-forms. Several other researchers have shown that woody plants are particularly sensitive and are usually eliminated from areas experiencing direct contact from trampling (Wagar, 1964; Dale and Weaver, 1974; Brown *et al.,* 1977). In addition, soil compaction from trampling can impede the development of tree seedling root systems (Hatchell *et al.,* 1970; Froehlich *et al.,* 1985; Reisinger *et al.,* 1988). Dale and Weaver (1974) and Weaver *et al.,* (1979) correspondingly observed that trees disappeared from the trail edge and that other introduced species became more common. They attributed these changes to a multitude of factors including direct effects of trampling, variation in light, an influx in seeds and propagules, changes to soil water and nutrients as well as soil disturbance. These factors may have also similarly contributed to the results of this study.

The reduction of tree sapling coverage is particularly important to species conservation since forests with dense canopies create conditions which assist in the maintenance of native composition. Low light and high moisture are typical of forests and the native species that occupy theses areas may be sensitive to acute changes in these conditions. The presence of a trail may cause breaks in the tree canopy causing increased light at the trail edge which may facilitate the establishment of invasive species (Hammitt and Cole, 1998; Leung and Marion, 2000). Furthermore, the reduction in tree saplings will undoubtedly increase light exposure as succession progresses and will exacerbate changes in composition by supporting light-tolerant species. Thus, large tracts of shaded areas may act as thresholds for some invasive species and aid in the preservation of native composition. This assumption is supported by our findings that many invasive and exotic species were found only within 2.5m of the trail (Sections 5.1 and 6.2). In addition, Dale and Weaver (1974) also observed more sunlight at a woodland trail edge which they attributed higher herbaceous plant diversity to. If tree saplings are removed entirely from an area, the likelihood of maintaining native composition may be poor, as this would eliminate canopies and their associated conditions. Moreover, since soil compaction from trampling can impede the development of tree seedling root systems there is likely a strong justification for prohibiting all trail use in areas where species of conservation concern require shady conditions.

### 6.5 Invasive Species

Recreational trails are widely believed to act as conduits in the spread of invasive species by providing vectors and pathways for species dispersal (Forman and Godron 1981, 1986; Baudry, 1984; Benninger-Truax *et al.*, 1992). Based on these assumptions, it was first anticipated that trails would induce greater disturbances relative to baseline, and adjacent areas would be exposed to higher frequencies of invasive introduction from users. As evident in the results, however, invasive species in hiking, multi-use and biking trails were not more prevalent than in control/baseline areas, overall (Tables 7 and 13, Sections 5.3 and 5.4). This was surprising due to the fact that these trail use types were found to differ with baseline areas in terms of species richness. There are two probable explanations for this observation. First, it is possible that the frequency of introduction by users was no greater than that of natural modes such as wind or transport by fauna. Deer and other mammals may be able to transport invasives in their fur, similar to horses. Second, the invasive species identified in this study may have been equally influenced by anthropogenic and natural disturbances. The presence of invasive species in areas where diversity was highest (baseline areas) may have been observed because invasives respond to greater habitat diversity in the same way that native species do (Pickard, 1984). Thus, the presence of invasive species in baseline areas is not irregular and may suggest that indigenous and invasive species richness are positively correlated without a causal link (Lonsdale, 1999). Unlike the other groups, proportional invasive coverage was found to be relatively higher on equestrian trails. Justification for the observed results of each specific trail use type is discussed throughout this section.

**6.5.1 Spatial Patterns of Invasive Species**

In addition to quantifying the presence of invasive species within different trail-influenced environments, this thesis sought to evaluate whether densities of species were greatest at the trail edge and if a distance-decay relationship exists, similar to the analysis of all species.

Initially, it was expected that the density of invasive species would decline with increasing distance which would exhibit resistance to the spread of these species. Moreover, greater invasive density near the trail edge was expected in anticipation of greater dispersal frequency and disturbance caused by trail users. Contrary to our expectations, invasive species did not have a linear relationship with distance from the trail as half were found throughout sampled transects and the other half directly adjacent to the trail (Sections 5.7 and 5.8). In particular, *Veronica officinalis* (common speedwell), *Galeopsis tetrahit* (hemp-nettle) and *Leonurus cardiaca ssp.cardiaca* (motherwort) were found exclusively at the trail edge (Section 5.8). These results largely confirm the assumption that trails act as conduits in the spread of invasive species and that disturbed areas facilitate their establishment before invading the forest interior. Others have noted that successful invasion requires dispersal, establishment and greater survival rates than extinction (Hobbs, 1989). Since most invading species fail to establish however, it is not surprising that some invasive species were less prevalent than others in our study and that they existed only at the trail edge (Williamson, 1996).

Although majority of species were found at the trail edge, we also observed that some invasives were not limited to the disturbance at this location. As described in Section 5.8, *Taraxacum officinale* (dandelion), *Alliaria petiolata* (garlic mustard), *Epipactis helleborine* (helliborine) and *Geranium robertianum* (herb Robert) were all found throughout the sampling areas as well as in baseline areas. Although these species do not require anthropogenic disturbance to invade floral communities, the influx of seed from these species undoubtedly contributes to their potential to spread across landscapes and dominate native species. This is particularly true for *Alliaria petiolata* (garlic mustard), which can be dispersed by humans and can cover large areas, in some cases hundreds of square metres (Nuzzo, 1991; Cavers *et al.*, 1979). Interestingly, garlic mustard was found throughout all transects of trail-influenced environments but not in baseline areas, which may imply that trail users have a greater influence on the dispersal of this specific species more so than natural processes.

Overall, the results suggest that despite the lack of linear correlation, invasive species exhibit spatial effects. Invasive species likely did not exhibit a quantifiable linear relationship with distance from the trail because these species were prevalent throughout transects or were confined to the trail edge. Moreover, establishing a linear relationship would require measurements to be made on a much larger scale than in the current study.

### 6.5.2 Trail Users and the Influence on Exotic and Invasive Species Presence

Knowledge of whether certain trail use types influence the richness and coverage of exotic and invasive species is particularly valuable for the management of these species in recreational areas and is often a subject of debate. Due to the fact that mechanisms responsible for the introduction of species differ among hikers, mountain bikers and equestrians, suspicion has arisen as to whether particular uses have greater potential to introduce invasive and exotic species. Results of this study suggest that the proportional abundance of species may be higher on equestrian trails, however, as explained further below, this did not occur due to greater disturbance.

Although hikers are known to transport seeds and propagules on their shoes and clothing, equestrian users are thought to create greater dispersal opportunities. In particular, horses can disperse seeds in two ways: by attaching to the fur of an animal (epizoochory) as well as through ingestion and excretion (endozoochory), the latter of which is not typical of other use types (Hammitt and Cole, 1998; Couvreur *et al.*, 2005; Newsome *et al.,* 2004; Cosyns and Hoffmann, 2005). Unfortunately, no study has been completed on the dispersal potential of mountain biking for comparison. Multi-use trails were also expected to exhibit a high amount of invasive and exotic species similar to equestrian trails since these trails were exposed to all users with various dispersal mechanisms.

The potential for different use types to act as vectors in the dispersal of invasive and exotic species was assessed by comparing their richness within 2.5m of the trail. Due to the fact that several of these species were not confined to the trail, we did not attempt to attribute dispersal potential with species that were found further than the trail edge (2.5m). Thus, total richness values could not have been used to compare the groups. Interestingly, of the 18 species found exclusively at the trail edge, 12 were identified as exotic or invasive inferring that these species were introduced by trail users (Section 5.1). What was surprising was equestrian trails did not harbour the greatest number of these species but rather, only harboured two species exclusively at the trail edge. In addition, hiking trails and multi-use trails had relatively more exotic and invasive species exclusively at the trail edge (Table 7, Section 5.1 and 5.3). Specifically, 4 species were exclusively found at the edge of hiking trails whereas 6 were in multi-use trails (Section 5.1). These observations collectively suggest that dispersal mechanisms may be greater for hikers and for individuals on trails that accommodate all types of use. It is important to note that although great lengths were taken to ensure hiking trails were exclusively used by hikers, equestrians and mountain bikers may have also used these trails at some point. Despite its unlikelihood, it may explain why hiking trails had relatively higher amounts of introduced species which were similar to multi-use trails. Furthermore, it may suggest that trails which incur more than one use type are prone to having greater numbers of invasive species.

Similar to species richness, invasive and exotic species coverage was also assessed to compare trail use types and their potential to harbour invasive species as a result of disturbance.

The results indicated that relative proportions of invasive species coverage were higher in equestrian trails compared to other use types (Table 13 and Section 5.4). While the average invasive coverage of transects was not found to significantly differ among groups, proportional invasive coverage was higher in equestrian trails (Section 5.4). It is important to note however, that this was primarily due to a single species, *Geranium robertianum* (herb Robert), which only existed within equestrian trails and was particularly dominant (Appendix A).

Increased invasive coverage in equestrian trails is likely not an indication of increased disturbance. For example, if disturbance was greater within equestrian trails it would be expected that dominance of other invasive species would also increase; however, this was not the case (Appendix A). A possible explanation for the relatively higher invasive coverage in equestrian trails may be due to the relatively sparse coverage in this use type overall (Table 13 and Section 5.4). Conceivably, environments with more area for seeds and propagules to establish may facilitate an increase in invasive coverage. The results collectively suggest that invasive and exotic coverage resulting from trail-related disturbances was relatively similar between the groups and that greater dispersal potential may not always be correlated with higher invasive coverage. This may be because species differ in their potential to be transported as well as in their ability to establish in stressed environments and compete with native species.

To my knowledge, the potential for biking to influence the spread of invasive species has not been previously assessed. In comparison with baseline areas, biking-influenced environments had relatively similar invasive species coverage and harboured only one exotic specie at its trail edge (Table 13, Sections 5.1 and 5.4). Thus, biking may potentially act as a dispersal mechanism for invasives, however, it is likely no more effective than that of natural modes. Overall, modes of transport may play a significantly greater role in hiking and multi-use trails leading to increased invasive richness but this does not explain the coverage of invasive species in these environments. Thus, controlling or prohibiting certain use types will not influence how much coverage invasive species occupy in trail-influenced environments. The same can be said for exotic species which had greater richness on hiking and multi-use trails but relatively the same amount of coverage in each trail use type (Section 5.4, Table 13).

Perhaps the greatest threat invasive species pose is to areas devoid of trails which are typically reserved or protected due to their high diversity. As evident from the findings discussed above, invasive species can spread great distances and into areas where disturbance is minimal. Since invasives tend to have relatively high growth rates, they compete vigorously for resources that can inhibit recruitment of native species. Consequently, invasives may decrease the diversity of a community by replacing native species to the extent that enough individuals from an existing population are extirpated (Anderson *et al.,* 1996). Long-term studies on the temperate deciduous forests of eastern North America confirm my general findings that native richness typically declines with the presence of introduced species (Brewer, 1980; Davison and Forman, 1982; Drayton and Primack, 1996; Rooney and Dress, 1997; Rooney *et al*., 2004). Cole (1987) also similarly found that trail users can introduce and transport exotic plant species and that some species can replace native vegetation and migrate away from trails.

In summary, the invasive species in this study were found either throughout sampled transects or were directly adjacent to the trail (Section 5.8). In the literature, several researchers have also demonstrated that invasive species exhibit both of these responses. In particular, Hall and Kuss (1989) determined that a highly aggressive invasive was unable to establish away from the trail edge. Similarly, Gower (2008) determined that in 50m transects, invasives did not exist more than 2.5m from the trail. Furthermore, Tyser and Worley (1992) found significant levels of invasive species 100m from the trail. The significant variations among the results of these studies, and the differences in the spatial patterns of invasive species found in this thesis, are likely due to the specific qualities of the individual species. Nevertheless, the evidence altogether infers that trails act as conduits in the movement of invasive species, that hiking and multi-use trails may have greater dispersal mechanisms and lastly, species which become established in these disturbed environments are able to migrate away from the trail edge (Buckley *et al.*, 2003; Benninger-Truax *et al*,. 1992). The fact that invasive species can achieve the same coverage in areas where disturbance is minimal further suggests that limits to anthropogenically disturbed areas do not act as thresholds to these species.

Invasive species pose a serious ecological threat to natural ecosystems and it is important to implement management actions to minimize their introduction. The invasive and exotic species in this study made up roughly a third of the species in trail-influenced environments and may have been a contributing factor in the overall reduction of species richness. Whether or not invasive and exotic species threatened native species through competition, they occupied approximately 10 percent or more of the total stand area (Table 13) and their threat to future native establishment exists. A significant finding of this study was that the introduction and dispersal of invasive species in recreation areas was certainly assisted by the presence of trail users. In particular, hiking and multi-use had the greatest influence in this regard. We also found that despite their greater potential to introduce species, trail-influenced environments will likely occupy similar coverage of invasive and exotic species. Controlling the spread of invasives to areas of high conservation value will require a significant distance from an area that contains trails or other sources of invasive seeds and propagules. Unless changes are made to the management of recreation areas, these areas will inherently be less diverse and will also be sources for further dispersal of invasive species.

### 6.6 Spatial Patterns of Trail Impacts

The primary objectives of this study were to determine if a relationship between diversity and distance from the trail could be observed and to quantify the spatial extent of trail impacts on vegetation. Our results generally revealed that species richness and composition exhibit spatial effects at distances greater than typically thought. The results also showed that this is likely caused by an influx of species at the trail edge, as well as a reduction of species richness in areas where the influx is less pronounced. Together, these changes caused species richness at the trail edge (0m) to be significantly higher than areas furthest sampled (17.5m and 25m) (Section 5.3). Although species richness did not exhibit a linear correlation with distance from the trail, the variation between the abovementioned distances supports our hypothesis that species richness and composition are influenced beyond the trail edge. The subsequent sections focus on the analytical methods used to quantify the spatial relationship between trail impacts and vegetation as well as the introduction and removal of species within these environments.

#### 6.6.1 Data Arrangement and the Importance of Aggregation in Spatial Analysis

Two methods of analysis were used to quantify the spatial effects of species richness in this thesis. This subsequently revealed an important weakness in the quantification of spatial impacts as applied to recreation resources. Typically, researchers have quantified spatial impacts by comparing the richness values at various distances from the trail, using separate transects as replicates (Dale and Weaver, 1974; Hall and Kuss, 1989; Weaver and Dale, 1978). This method of analysis is, however, misleading because it accounts for species which exist at the same distance more than once. Thus, if attempting to determine if greater amounts of species exist at various distances, the results are likely to be convoluted by common species being accounted for repeatedly. This held true in the present study, as this method of analysis incorrectly observed that species richness did not change with distance from the trail (Table 8, Section 5.3). To better quantify the spatial influence on a community scale, the number of species found in separate replicates should be summed for each distance without accounting for the same species more than once. This method proved to be a better measure of species richness compared to the latter since the aggregate data found richness to differ significantly among several distances from the trail (Table 9, Section 5.3). Thus, if researchers and managers are to accurately determine if species richness changes on a community scale, the latter method should be applied. Such arrangement of richness values may elucidate whether rare and uncommon species are either increasing or reduced in certain areas, assuming that the dominant species have not changed. It also supports the need for analysis of trail impacts at greater distances as the aggregate analysis confirms the observations that individual transects do not clearly reveal.

### 6.6.2 Additions to Species Pool

The variation in species richness between areas adjacent to the trail (0m) and areas furthest sampled can be explained by describing the factors that likely influenced the influx and reduction of species in trail-influenced environments. As discussed in the previous section, trails act as conduits in the spread of species. Seeds and propagules can be spread by recreationalists via a variety of mechanisms and this causes the introduction of new species to areas adjacent to the trail. Our results confirmed this was occurring as roughly half of the exotic and invasive species existed exclusively within 2.5m and higher richness was evident in these areas (Table 9 and Section 5.1). Although some invasive and exotic species were found throughout the sampled areas, far greater amounts of these species existed within 10m of the trail. This suggests that the majority of introduced species exhibited a distance threshold and that seed dispersal beyond 10m from the trail was rarely successful in this system. For the most part, introduced species existed outwards of 10m but were also primarily concentrated within 2.5m of the trail.

There are several reasons why species richness was higher at the trail edge and why there may be a threshold of species influx at roughly 10m from the trail. First, areas closer to the trail are physically closer to the source of introduction hence, it seems reasonable these areas are their point of establishment. Gower (2008) similarly demonstrated that invasives do not exist more than 2m from the trail. Secondly, increased frequencies of off-trail trampling near the trail may have caused voids in vegetation coverage allowing for specie rich forbs to establish. As previously discussed, areas directly adjacent to the trail were exposed to the highest frequencies of off-trail trampling. This was supported by our findings that several dominant species exhibited reduced coverage in these areas (Table 4, Section 5.1). Correspondingly, analysis of species growth-form revealed that the majority of species which existed near the trail were forbs (Section 5.1). Since forbs were the growth-form with the highest number of unique species, they were likely the causal factor contributing to the high richness at the trail edge. Forbs may have been able to occupy these spaces because they are fast growing and can recover quickly from damage. Thus, greater physical disturbance near the trail edge likely facilitated the establishment of forbs, which were high in species richness and contributed to high species richness in these areas overall (Table 15, Sections 5.6 and 5.3).

Since the majority of invasive and exotic species existed within the first 10m of the trail edge, there may be reason to believe that environmental conditions facilitated their establishment. There is also reason to believe that conditions near the trail may not be the same in areas beyond this distance. During construction of a trail, trees are often removed to establish a route of travel. During use, off-trail trampling may also reduce tree coverage in adjacent areas as was confirmed in our results (Table 16, Section 5.6). The removal of trees may cause breaks in the tree canopy and consequently increase light at the trail edge. While it is unknown if light was actually higher near the trail, there is a possibility that this occurred as evenness which is driven by uncommon species was found to be influenced by aspect (Section 5.9.1). Others have also shown that trail edges have more light compared to the interior of forests and have attributed increased invasive establishment to this environmental factor (Hammitt and Cole, 1998; Leung and Marion, 2000). Thus, trailside vegetation may have experienced greater levels of light compared to the forest interior and contributed to the establishment new species. In support of this, Dale and Weaver (1974) and Weaver *et al.,* (1979) similarly observed that introduced species became more common at the trail edge. They ascribed these changes to a multitude of factors associated with breaks in canopy including increased light, direct precipitation, as well as changes to soil water and nutrients. In addition, Hall and Kuss (1989) found that increased light intensity and temperature caused trailside vegetation in eastern hardwood forests to be more diverse than undisturbed vegetation. This strongly supports the concept that increased richness at the trail edge is influenced by changes in environmental conditions in what is known as an ‘edge effect’.

An edge effect is defined as the creation of internal edges in the forest. The distance that an edge effect has is dependent on a multitude of factors including tree height and size of canopy break, among others. Thus, the lack of species influx past 10m from the trail may be reason to believe that this distance is the limit for the influence of edge effect in the present study. Additionally, no introduced species existed exclusively at the greatest distance sampled (25m) further suggesting that 10m remains a threshold for introduced species. In support of this, Gorski

(1975) demonstrated that light is depleted in dense forests and as a result, the germination of shade intolerant species is inhibited. The latter suggests that introduced species which exist exclusively within 10m of the trail in this study were likely shade intolerant and were not well adapt to shady forest conditions (Fenner, 1978). It is important to note that some invasive and exotic species were found throughout the sampled areas and that the 10m threshold applies to the majority of introduced species, not all.

### 6.6.3 Reductions in Species Pool

The second component causing spatial variation in species richness in the present study was the reduction of species furthest from the trail. Compared to the trail edge (0m), quadrants sampled at 17.5m and 25m were reduced in species richness on average of 30.5 and 35 percent, respectively (Section 5.3). Analysis of vegetation dominance further revealed that species eliminated from these areas were typically uncommon and that all dominant species remained in areas furthest sampled from the trail (Table 9, Section 5.3). Interestingly, areas furthest from the trail (25m) exhibited strikingly similar richness values, as every trail use type had either 12 or 13 species.

Although there are likely several factors contributing to the reduction in species richness in areas furthest from the trail, trampling is likely the primary cause. As previously described in the literature review, trampling can cause a multitude of negative influences on vegetation and soil resulting in decreased coverage and richness (Liddle, 1997). Aside from being directly crushed or damaged, species may also be indirectly eliminated from trail-influenced environments due to grazing pressure from horses, soil and root compaction and reduced soil moisture, among other reasons (Bates, 1935; Burden and Randerson, 1972; Dale and Weaver, 1974; Cole, 1981; Liddle and Greig-Smith, 1975). Trampling also requires few occurrences to have severe influence on vegetation. The most impact to vegetation typically occurs with initial contact of the plant (Hammitt and Cole, 1987; Hartley, 1999). Thus, even a single encounter with a recreationalist can potentially destroy an individual plant. Since it is likely that off-trail trampling is an infrequent occurrence, this may explain why species richness, at 25m from the trail, was consistently reduced by 50 percent compared to baseline areas (Section 5.3). This may also explain why few species were removed entirely from trail-influenced environments (Section 5.1). Thus, over extended periods of time, these types of disturbances may proliferate throughout recreational areas.

Another contributing factor to the reduced species richness in areas furthest from the trail was the relatively low coverage of most species. The majority of species not present in areas furthest from the trail were uncommon species that likely occupied little coverage in absence of disturbance. Conceivably, species that are uncommon will be reduced in richness more quickly than species with numerous representatives in a given area subject to the same disturbance. Species with sparse coverage may thus be more likely damaged and consequently result in reduced richness. In addition, species existing in low light conditions of the forest interior often have morphological characteristics which are poorly suited to survive trampling. In comparison with plants which grow in areas with greater light, shade tolerant plants tend to have greater leaf areas and thinner cuticles, stems and cell walls, as well as more supportive and conductive tissue (Daubenmire, 1974; Treshow, 1970). These growth-forms are extremely vulnerable to damage and are quickly eliminated which may allude to the possibility that species at the trail edge are more resistant to damage than interior species. This also may explain why areas experiencing the greatest frequencies of off-trail trampling did not experience significant reductions in vegetation coverage. In support of this, Dale and Weaver (1974) and Cole (1978) suggest that shade-tolerant species are more susceptible to elimination by trampling.

Lastly, species richness at distances furthest from the trail significantly varied from the trail edge possibly due to a lack of introduced species compensating for losses caused by trampling. Despite areas closest to the trail receiving the greatest amount of off-trail trampling, the forest interior did not experience an influx of introduced species. As previously mentioned, this was caused by a multitude of factors related to the 'edge effect' and the inability of most introduced species to penetrate into the forest interior. Thus, where the influx of species is less pronounced species richness will inevitably be severely reduced. These findings are largely confirmed by others who demonstrated that species richness and diversity are generally reduced where recreational impact is present (Young, 1978; Cole, 1993; Hartley, 1979).

### 6.6.4 Spatial Effects of Other Measures of Composition

It is important to note that unlike richness, other measures of species composition including vegetation coverage, evenness, and TRCA sensitivity scores did not exhibit spatial effects. This may have occurred for several reasons. The areas under study largely consisted of forb species which for the most part were uncommon individuals that occupied less than 1 percent of the coverage in their respective areas (Section 5.6). The introduction of these species rich growth-forms may thus explain why these areas were high in richness but did not significantly contribute to the overall coverage. In this thesis, species coverage was generally lower in trail-influenced environments and no spatial effects were observed (Section 5.4). The lack of a visible effect was probably because of off-trail trampling occurring at a frequency sufficient to remove uncommon species but not high enough to cause actual thinning of vegetation. Furthermore, reductions in coverage were filled by forbs in areas closest to the trail and also by shade tolerant dominants in the interior thus compensating for slight reductions in vegetation coverage.

Similar to vegetation coverage, species grouped by sensitivity also did not exhibit a linear relationship with distance from the trail as expected (Section 5.7). Furthermore, not all trail use types were influenced by distanced when assessing use the Shannon scores, similar to richness values. This may have occurred because species differ in their responses to trampling. A lack of an observable effect may also be due to the responses of different species within these groups confounding each other. This was supported by our finding that some species exhibited opposite correlations in their analysis with distance (Table 4). Bratton (1985) similarly demonstrated that the number of shoots of showy orchids (*orchis spectabilis*) was negatively correlated with distance from the trail yet the same effect was not observed with the closely related species ladies slipper orchid (*cypropedium calceolus*) in the same vegetation type. Thus, it is often difficult to make generalizations regarding groups of species and their response to different stressors. Evenness values and typically Shannon scores in most trail use types were not correlated with distance from the trail due to the methods used to quantify these values (Sections 5.5 and 5.2). Shannon scores and evenness values accounted for species that occur more than once in separate transects and as previously discussed with respect to richness, this causes results to be convoluted by common species being accounted for repeatedly. It is important to note however, that it is not possible to determine Shannon scores and evenness values without accounting for species more than once. Therefore, researchers should take this into consideration when selecting a measure of biodiversity in the future. Furthermore, it is recommended that species richness be used alone when assessing spatial effects.

Overall, our findings largely confirm that spatial variation in species richness is caused by both the use of the trail and their associated environmental conditions. These two variables increased richness and changed species composition near the trail edge as well as likely contributed to the reduction in species richness beyond 10m of the trail. While we did observe that species richness was significantly lower at distances of 17.5m and 25m from the trail, our study may only have captured a fraction of the spatial extent of areas reduced in species richness. A much larger study would be required to determine the distance required for diversity to reach baseline conditions. Baseline areas in this study were established approximately 500m from the trail-influenced environments, at the furthest extent. Thus, the spatial influence of species reduction may extend a few hundred metres, however, this needs to be confirmed with additional research. Nevertheless, trail impacts caused by off-trail trampling do extent significant distances from the trail. If managers are to effectively mitigate these impacts and protect native composition greater distances from trail-influenced environments must be established.

**6.7 Limitations**

Although the aims of this research were met, there were unfortunately, some unavoidable limitations with respect to the study design. Firstly, control sites are never perfect replicates of pre-existing conditions. The area under study was within 100km of a large city metropolitan and had areas open to public access. Thus, establishing a control where disturbance was minimal may not have ensured that the chosen areas were unaffected or predisposed to biases. Nevertheless, baseline areas were chosen with well supported criteria as defined in the methodology. Secondly, estimates of the frequency of users in certain areas were established with trail counters but there was no way of determining what the exact frequencies were for the specific trails under study. In addition, it could not be conclusively established whether certain trails were exclusively used by one user type, though this was highly unlikely.

This thesis also aimed to quantify the spatial influence of trail impacts on species composition. Unfortunately, the size of the study area that was chosen did not establish the distance required for diversity to reach baseline conditions. However, it is important to note that this study quantified trail impacts at greater distances than typically studied in the literature. Moreover, it identified that species richness can be significantly reduced in areas previously considered to be unaffected by trail use.

Other limitations in this study were also present with regards to the analysis of the data. Due to the difficulty of estimating the exact percent of vegetation coverage and the small size of some species, it is possible that some individuals may have been obscured in the sampling though great care was taken to prevent this. Estimation of species coverage may have also caused small errors in data collection. Moreover, the lack of replicate trails may have caused some inherent bias in the multiple regression models. This was unavoidable as there are few trails where a single use type could be determined. In addition, limiting sampling to only one growth season could have skewed the results and misrepresented trends that occur due to seasonal fluctuations.

It should be further noted that the results from this study may not be representative of other vegetation types and are likely limited to Sugar Maple-Beech Deciduous Forests. Also, one can speculate on causal factors from correlations and multiple regression analysis, but these relationships can be spurious and actual relationships can be missed due to other confounding or intervening variables not taken into account.

# Chapter 7: Conclusion

Trail managers are often charged with having to provide recreational experiences for the public while attempting to preserve a high level of environmental quality. As one would expect, this task is difficult due to the lack of scientific information to justify management decisions and the complexity of mitigating multiple impacts. An area of debate that this thesis sought to address was the spatial extent of trail impacts on species diversity. Overall, it was demonstrated that vegetation adjacent to a trail does exhibit a spatial effect and at distances greater than previously thought. In particular, areas within 10m of the trail experienced an influx of species caused by both the introduction of recreationalists and conditions which facilitated their establishment, known as an edge effect. However, areas beyond the influence of an edge effect were shown to incur significant reductions in species richness caused by off-trail trampling. These findings hold several implications for trail managers and those seeking to preserve floral composition.

Most importantly, over extended periods of time, areas previously considered to be devoid of trail impacts may experience reductions in species richness. Establishing an appropriate buffer to areas of conservation concern and controlling the density of trails may prove to be important mitigation measures in preventing species loss and is a natural progression for future research. As demonstrated, vegetation coverage and consequently, species richness in this particular vegetation type were sensitive to infrequent trampling. These observations confirmed the longstanding concept that wherever recreational trail use is present, some degree of environmental impact is inevitable and shifts in species composition will occur.

Since resources of trail managers are often limited, the question of whether efforts should be directed towards mitigating trail use impacts or controlling the spread of exotic and invasive species is of significant importance. While introduced species found in the present study occupied roughly a third of the total species richness, they accounted for less than 10 percent of the total coverage in trail-influenced environments. Moreover, most exotic species were confined to the trail edge whereas reductions in species richness upwards of 50 percent were observed at much greater distances. Thus, the threats to native species caused by invasive and exotic competition may be less important compared to disturbances caused by trampling. Considering the greater influence on indigenous species, conservation efforts may be more effective if off-trail trampling is prevented.

Another objective of this thesis was to determine if trail users disproportionately influence species diversity. Although total vegetation coverage and dispersal mechanisms of exotic species differed between the groups, trail use types had similar amounts of indigenous species and did not differ in their amounts of exotic and invasive species coverage. Thus, prohibiting certain use types from recreation areas is likely not an effective management action for floral conservation purposes. Prohibiting all trail use may however prevent severe reductions of indigenous species richness. It may also prevent the spread of exotic species in trail-influenced environments, though the same cannot be said for invasive species.

Lastly, multiple regression analysis revealed that managers can mitigate changes to species composition by choosing routes with high side-slopes and also by avoiding areas with south facing aspects. Such measures may prevent the establishment of exotic species in trail edges and physically prevent users from trampling off the trail tread. In contrast, trail characteristics such as trail depth and trail width were found to not directly influence species diversity. While actions taken to mitigate the latter may be an effective solution for sustaining trail tread and preventing erosion, both trail characteristics likely have little influence on the composition and the preservation of indigenous species.

Overall, species diversity in trail-influenced environments is likely to be significantly reduced as there are typically fewer species tolerant of trail-related impacts compared to original occupants. This may explain why diversity was not higher in trail-influenced environments as expected based on the Intermediate Disturbance Hypothesis. In light of this, creating dual purpose areas which provide recreational opportunities and are charged with conserving species diversity will likely not achieve the latter. If conserving species diversity is of upmost concern, managers should consider either closing trails or concentrating their use since spatial impacts are large and changes to composition are inevitable.

# Appendices

**Appendix A. Percent Coverage and Sensitivity Scores of Individual Species in Trail-Influenced Environments and Baseline Areas.**

| Sensitivity | Scientific Name | Common Name | Baseline | Hiking | Multi-use | Biking | Equestrian |
|---|---|---|---|---|---|---|---|
| 2 | *Cornus alternifolia* | alternate-leaved dogwood | 0.27 | 0.03 | 0.00 | 0.58 | 0.47 |
| 4 | *Fagus grandifolia* | American beech | 0.00 | 0.02 | 0.00 | 0.00 | 0.00 |
| 3 | *Tilia americana* | basswood | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 |
| 4 | *Corylus cornuta* | beaked hazel | 0.80 | 0.00 | 0.00 | 0.00 | 0.00 |
| 2 | *Carya cordiformis* | bitternut hickory | 1.05 | 0.35 | 0.00 | 0.23 | 0.42 |
| 2 | *Prunus serotina* | black cherry | 0.00 | 0.00 | 0.02 | 0.10 | 0.18 |
| 3 | *Sanguinaria canadensis* | bloodroot | 0.17 | 0.00 | 0.97 | 0.03 | 0.40 |
| 5 | *Caulophyllum thalictroides* | blue cohosh | 1.98 | 0.02 | 0.17 | 0.00 | 0.13 |
| 2 | *Solidago caesia* | blue-stemmed goldenrod | 0.27 | 0.07 | 0.00 | 0.07 | 0.13 |
| 4 | *Diervilla lonicera* | bush honeysuckle | 0.05 | 0.00 | 0.00 | 0.00 | 0.00 |
| 5 | *Maianthemum canadense* | Canada mayflower | 1.10 | 5.63 | 3.48 | 4.00 | 1.37 |
| 4 | *Viola canadensis* | Canada violet | 0.00 | 0.00 | 0.00 | 0.03 | 0.00 |
| 2 | *Smilax herbacea* | carrion-flower | 0.07 | 0.05 | 0.10 | 0.35 | 0.00 |
| 1 | *Prunus virginiana* ssp. *virginiana* | choke cherry | 0.10 | 0.23 | 0.03 | 0.00 | 0.00 |
| 5 | *Polystichum acrostichoides* | Christmas fern | 1.27 | 0.00 | 0.00 | 0.00 | 0.00 |
| 0 | *Rhamnus cathartica* | common buckthorn | 0.00 | 0.20 | 0.00 | 0.00 | 0.03 |
| 2 | *Sambucus canadensis* | common elderberry | 0.00 | 0.17 | 0.53 | 0.00 | 0.37 |
| 0 | *Veronica officinalis* | common speedwell | 0.03 | 0.03 | 0.00 | 0.00 | 0.00 |
| 2 | *Carex blanda* | common wood sedge | 0.57 | 0.05 | 0.53 | 2.13 | 0.07 |
| 1 | *Oxalis stricta* | common yellow wood-sorrel | 0.00 | 0.02 | 0.13 | 0.00 | 0.00 |
| 0 | *Taraxacum officinale* | dandelion | 0.05 | 0.52 | 0.03 | 0.00 | 0.03 |
| 5 | *Polygonatum pubescens* | downy Solomon's seal | 0.65 | 0.28 | 5.67 | 3.43 | 1.17 |
| 4 | *Pteridium aquilinum* var. *latiusculum* | eastern bracken | 0.27 | 0.00 | 0.50 | 0.00 | 0.00 |

| Sensitivity | Scientific Name | Common Name | Baseline | Hiking | Multi-use | Biking | Equestrian |
|---|---|---|---|---|---|---|---|
| 1 | *Circaea lutetiana ssp. canadensis* | enchanter's nightshade | 0.02 | 0.00 | 0.00 | 0.00 | 0.07 |
| 3 | *Dryopteris intermedia* | evergreen wood fern | 1.43 | 0.37 | 10.23 | 0.23 | 1.95 |
| 3 | *Maianthemum racemosum* ssp. *racemosum* | false Solomon's seal | 0.60 | 0.07 | 0.53 | 0.03 | 0.47 |
| 3 | *Carex communis* | fibrous-rooted sedge | 0.77 | 0.00 | 0.00 | 0.00 | 0.40 |
| 1 | *Equisetum arvense* | field horsetail | 0.07 | 0.00 | 0.00 | 0.00 | 0.00 |
| 4 | *Lonicera canadensis* | fly honeysuckle | 0.00 | 0.00 | 0.10 | 0.37 | 0.00 |
| 3 | *Polygonum cilinode* | fringed black bindweed | 0.00 | 0.00 | 0.03 | 0.00 | 0.00 |
| 0 | *Alliaria petiolata* | garlic mustard | 0.00 | 0.67 | 2.17 | 0.27 | 0.93 |
| 0 | *Epipactis helleborine* | helleborine | 0.45 | 0.13 | 0.08 | 0.17 | 0.03 |
| 0 | *Galeopsis tetrahit* | hemp-nettle | 0.00 | 0.00 | 0.08 | 0.00 | 0.00 |
| 0 | *Geranium robertianum* | herb Robert | 0.17 | 0.00 | 0.00 | 0.00 | 4.43 |
| 2 | *Amphicarpaea bracteata* | hog-peanut | 0.10 | 0.00 | 0.00 | 0.00 | 0.00 |
| 2 | *Ostrya virginiana* | ironwood | 0.28 | 0.33 | 0.00 | 0.07 | 0.17 |
| 3 | *Arisaema triphyllum* | Jack-in-the-pulpit | 0.13 | 0.03 | 0.00 | 0.00 | 0.33 |
| 2 | *Ranunculus abortivus* | kidney-leaved buttercup | 0.00 | 0.03 | 0.00 | 0.00 | 0.00 |
| 4 | *Caulophyllum giganteum* | long-styled blue cohosh | 1.13 | 0.00 | 0.00 | 0.00 | 0.47 |
| 3 | *Carex laxiflora* | loose-flowered sedge | 0.17 | 0.87 | 0.00 | 0.00 | 0.00 |
| 5 | *Viburnum acerifolium* | maple-leaved viburnum | 4.47 | 0.00 | 0.43 | 1.20 | 0.20 |
| 3 | *Podophyllum peltatum* | May-apple | 0.00 | 0.20 | 0.00 | 0.00 | 0.00 |
| 5 | *Mitella diphylla* | mitrewort | 0.07 | 0.00 | 0.00 | 0.00 | 0.00 |
| 0 | *Leonurus cardiaca* ssp. *cardiaca* | motherwort | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 |
| 2 | *Matteuccia struthiopteris* var. *pensylvanica* | ostrich fern | 0.00 | 0.00 | 0.07 | 0.00 | 0.00 |
| 5 | *Mitchella repens* | partridgeberry | 0.00 | 0.00 | 0.00 | 0.43 | 0.00 |
| 4 | *Carex pensylvanica* | Pennsylvania sedge | 2.20 | 12.13 | 0.27 | 1.83 | 3.12 |
| 5 | *Desmodium glutinosum* | pointed-leaved tick-trefoil | 1.67 | 0.13 | 0.00 | 0.00 | 0.00 |
| 2 | *Rhus radicans* ssp. *rydbergii* | poison ivy (shrub form) | 0.13 | 0.00 | 0.00 | 0.03 | 0.00 |

| Sensitivity | Scientific Name | Common Name | Baseline | Hiking | Multi-use | Biking | Equestrian |
|---|---|---|---|---|---|---|---|
| 5 | *Asclepias exaltata* | poke milkweed | 0.00 | 0.03 | 0.00 | 0.00 | 0.00 |
| 2 | *Ribes cynosbati* | prickly gooseberry | 0.00 | 0.05 | 0.00 | 0.00 | 0.00 |
| 5 | *Schizachne purpurascens* ssp. *purpurascens* | purple melic grass | 0.77 | 0.00 | 0.00 | 0.00 | 0.00 |
| 5 | *Acer rubrum* | red maple | 0.15 | 0.00 | 0.20 | 0.18 | 0.00 |
| 4 | *Quercus rubra* | red oak | 0.40 | 0.00 | 0.00 | 0.03 | 0.00 |
| 5 | *Trillium erectum* | red trillium | 0.00 | 0.00 | 0.53 | 0.00 | 0.00 |
| 2 | *Equisetum hyemale* ssp. *affine* | scouring-rush | 0.00 | 0.00 | 0.13 | 0.00 | 0.00 |
| 5 | *Anemone acutiloba* | sharp-lobed hepatica | 0.00 | 0.00 | 0.00 | 0.00 | 0.10 |
| 4 | *Pyrola elliptica* | shinleaf | 0.03 | 0.00 | 0.00 | 0.03 | 0.00 |
| 2 | *Dryopteris carthusiana* | spinulose wood fern | 0.00 | 0.40 | 0.00 | 0.00 | 0.00 |
| 5 | *Trientalis borealis* ssp. *borealis* | star-flower | 0.40 | 0.05 | 0.03 | 0.00 | 0.00 |
| 2 | *Viola pubescens* | stemmed yellow violet | 0.00 | 0.73 | 0.37 | 0.17 | 0.03 |
| 2 | *Acer saccharum* ssp. *saccharum* | sugar maple | 21.70 | 25.28 | 5.02 | 5.42 | 7.65 |
| 2 | *Galium triflorum* | sweet-scented bedstraw | 0.13 | 0.00 | 0.25 | 0.07 | 0.00 |
| 2 | *Prenanthes altissima* | tall wood lettuce | 0.03 | 0.00 | 0.67 | 0.10 | 0.00 |
| 3 | *Fraxinus americana* | white ash | 6.97 | 2.72 | 2.48 | 0.90 | 6.15 |
| 2 | *Geum canadense* | white avens | 0.20 | 0.00 | 0.00 | 0.00 | 0.00 |
| 3 | *Actaea pachypoda* | white baneberry | 0.07 | 0.07 | 0.60 | 0.23 | 0.00 |
| 4 | *Pinus strobus* | white pine | 0.00 | 0.23 | 0.00 | 0.00 | 0.00 |
| 5 | *Trillium grandiflorum* | white trillium | 0.28 | 0.52 | 0.62 | 0.53 | 0.00 |
| 5 | *Oryzopsis asperifolia* | white-fruited mountain-rice | 0.53 | 0.03 | 0.30 | 0.00 | 0.03 |
| 5 | *Aquilegia canadensis* | wild columbine | 0.00 | 0.00 | 0.00 | 0.00 | 0.10 |
| 4 | *Aralia nudicaulis* | wild sarsaparilla | 2.08 | 1.97 | 4.67 | 2.52 | 0.10 |

| Sensitivity | Scientific Name | Common Name | Baseline | Hiking | Multi-use | Biking | Equestrian |
|---|---|---|---|---|---|---|---|
| 3 | *Osmorhiza claytonii* | woolly sweet cicely | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 |
| 2 | *Impatiens pallida* | yellow touch-me-not | 0.00 | 0.00 | 0.00 | 0.00 | 0.75 |

The percent coverage of an individual specie in baseline areas and individual trail use types was summed from 60 quadrants (6 quadrants in 10 transects) and calculated as a percentage of the total area sampled, including the amount of bare ground. The percent coverage from the individual trail use types (hiking, biking, multi-use and equestrian) were combined (240 quadrants) and calculated as a percentage of the total area sampled. This group is presented as 'All Users'. Data is arranged in alphabetical order of species common names.

**Appendix B. Shannon Scores within Each Quadrant for Trail-Influenced Environments and Baseline Areas.**

| | | Distance from Trail (Shannon Scores) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Hiking** | **1** | 0.92 | 1.21 | 1.31 | 1.15 | 1.14 | 1.35 |
| | **2** | 0.74 | 1.03 | 0.69 | 0.50 | 0.69 | 1.08 |
| | **3** | 1.20 | 1.04 | 0.97 | 1.22 | 1.49 | 0.87 |
| | **4** | 0.30 | 0.86 | 0.91 | 1.16 | 0.56 | 0.07 |
| | **5** | 0.50 | 1.49 | 0.47 | 0.80 | 1.02 | 1.40 |
| | **6** | 0.76 | 0.00 | 0.66 | 0.34 | 0.47 | 0.80 |
| | **7** | 0.96 | 1.25 | 1.72 | 1.00 | 1.17 | 1.44 |
| | **8** | 1.29 | 1.01 | 1.73 | 1.48 | 1.19 | 1.16 |
| | **9** | 1.84 | 1.10 | 0.88 | 0.76 | 1.35 | 1.08 |
| | **10** | 1.06 | 1.04 | 0.38 | 0.80 | 1.15 | 1.10 |

| | | Distance from Trail (Shannon Scores) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Multi-Use** | **1** | 0.61 | 0.92 | 1.40 | 1.33 | 0.87 | 0.00 |
| | **2** | 0.41 | 0.69 | 0.85 | 1.14 | 0.76 | 1.07 |
| | **3** | 1.13 | 1.18 | 1.39 | 1.24 | 0.89 | 0.64 |
| | **4** | 1.38 | 0.38 | 1.26 | 1.20 | 0.94 | 0.69 |
| | **5** | 0.92 | 1.27 | 0.98 | 1.40 | 1.22 | 0.80 |
| | **6** | 0.60 | 0.69 | 1.22 | 1.51 | 1.16 | 1.28 |
| | **7** | 1.88 | 1.24 | 0.93 | 0.98 | 1.70 | 1.06 |
| | **8** | 1.89 | 1.48 | 1.74 | 1.22 | 0.50 | 1.33 |
| | **9** | 0.98 | 0.69 | 0.00 | 0.80 | 1.02 | 0.67 |
| | **10** | 1.42 | 1.22 | 0.68 | 0.66 | 1.10 | 0.00 |

| | | Distance from Trail (Shannon Scores) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Biking** | **1** | 1.02 | 1.66 | 1.05 | 1.06 | 0.69 | 1.13 |
| | **2** | 1.29 | 1.72 | 1.33 | 1.74 | 1.35 | 0.87 |
| | **3** | 0.69 | 0.99 | 1.16 | 0.99 | 0.35 | 1.10 |
| | **4** | 1.03 | 0.91 | 1.49 | 1.68 | 0.50 | 0.95 |
| | **5** | 0.00 | 1.15 | 1.21 | 1.29 | 1.05 | 0.00 |
| | **6** | 0.99 | 1.34 | 1.75 | 1.28 | 1.29 | 0.00 |
| | **7** | 0.96 | 1.04 | 1.12 | 1.05 | 0.57 | 0.00 |
| | **8** | 1.10 | 1.54 | 1.50 | 1.05 | 0.94 | 1.19 |
| | **9** | 1.36 | 1.15 | 1.03 | 1.28 | 1.64 | 1.03 |
| | **10** | 1.59 | 1.44 | 1.30 | 0.36 | 1.07 | 0.97 |

| | | Distance from Trail (Shannon Scores) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Equestrian** | **1** | 0.82 | 0.60 | 0.80 | 1.07 | 0.35 | 0.68 |
| | **2** | 1.63 | 1.31 | 0.88 | 0.25 | 0.59 | 0.94 |
| | **3** | 1.05 | 0.69 | 0.57 | 0.64 | 0.88 | 1.00 |
| | **4** | 1.20 | 0.91 | 0.90 | 0.69 | 1.28 | 1.87 |
| | **5** | 1.04 | 1.46 | 0.90 | 0.45 | 0.00 | 1.33 |
| | **6** | 1.21 | 1.17 | 1.72 | 1.01 | 0.96 | 0.50 |
| | **7** | 0.68 | 0.82 | 1.05 | 1.09 | 0.67 | 0.29 |
| | **8** | 0.98 | 0.35 | 1.35 | 0.45 | 0.00 | 0.14 |
| | **9** | 1.62 | 1.13 | 0.61 | 1.18 | 1.03 | 0.99 |
| | **10** | 0.38 | 0.35 | 1.10 | 1.41 | 0.38 | 1.28 |

| | | Distance from Trail (Shannon Scores) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Baseline** | **1** | 1.64 | 1.32 | 1.49 | 1.72 | 1.36 | 1.95 |
| | **2** | 1.91 | 1.60 | 1.73 | 1.69 | 2.08 | 1.75 |
| | **3** | 1.43 | 1.80 | 2.17 | 1.63 | 1.73 | 1.94 |
| | **4** | 1.22 | 1.27 | 1.00 | 1.33 | 0.41 | 1.23 |
| | **5** | 0.76 | 1.07 | 1.06 | 1.27 | 1.24 | 1.47 |
| | **6** | 1.08 | 0.81 | 0.26 | 0.98 | 0.70 | 0.30 |
| | **7** | 1.26 | 0.64 | 0.94 | 0.82 | 1.08 | 0.55 |
| | **8** | 1.15 | 1.18 | 1.50 | 1.08 | 1.49 | 1.09 |
| | **9** | 1.08 | 1.16 | 1.22 | 1.30 | 1.53 | 1.21 |
| | **10** | 1.41 | 1.09 | 1.30 | 1.14 | 1.94 | 1.09 |

**Appendix C. Species Richness of Individual Quadrants in Trail-Influenced Environments and Baseline Areas.**

| | | Distance from Trail (Richness) | | | | | |
|---|---|---|---|---|---|---|---|
| | Transect | 0m | 2.5m | 5m | 10m | 17.5m | 25m |
| Hiking | 1 | 5 | 5 | 6 | 7 | 6 | 7 |
| | 2 | 3 | 4 | 3 | 2 | 2 | 3 |
| | 3 | 4 | 3 | 3 | 5 | 6 | 5 |
| | 4 | 2 | 3 | 4 | 4 | 3 | 2 |
| | 5 | 5 | 6 | 3 | 3 | 4 | 5 |
| | 6 | 4 | 1 | 3 | 2 | 2 | 3 |
| | 7 | 4 | 5 | 8 | 4 | 4 | 5 |
| | 8 | 6 | 5 | 7 | 6 | 4 | 6 |
| | 9 | 8 | 6 | 5 | 3 | 5 | 3 |
| | 10 | 3 | 4 | 2 | 4 | 6 | 5 |

| | | Distance from Trail (Richness) | | | | | |
|---|---|---|---|---|---|---|---|
| | Transect | 0m | 2.5m | 5m | 10m | 17.5m | 25m |
| Multi-Use | 1 | 4 | 5 | 6 | 5 | 3 | 1 |
| | 2 | 2 | 3 | 4 | 4 | 3 | 3 |
| | 3 | 5 | 4 | 5 | 4 | 3 | 2 |
| | 4 | 5 | 2 | 4 | 4 | 3 | 2 |
| | 5 | 3 | 4 | 3 | 5 | 4 | 3 |
| | 6 | 2 | 3 | 4 | 5 | 4 | 5 |
| | 7 | 8 | 4 | 3 | 3 | 7 | 4 |
| | 8 | 8 | 5 | 6 | 4 | 2 | 5 |
| | 9 | 4 | 2 | 1 | 3 | 4 | 2 |
| | 10 | 6 | 4 | 2 | 2 | 3 | 1 |

| | | Distance from Trail (Richness) | | | | | |
|---|---|---|---|---|---|---|---|
| | Transect | 0m | 2.5m | 5m | 10m | 17.5m | 25m |
| Biking | 1 | 4 | 7 | 5 | 3 | 2 | 4 |
| | 2 | 4 | 6 | 5 | 8 | 4 | 3 |
| | 3 | 2 | 3 | 4 | 3 | 2 | 3 |
| | 4 | 3 | 3 | 5 | 6 | 2 | 4 |
| | 5 | 1 | 4 | 4 | 5 | 3 | 1 |
| | 6 | 4 | 5 | 6 | 4 | 5 | 1 |
| | 7 | 3 | 3 | 4 | 3 | 3 | 1 |
| | 8 | 3 | 6 | 5 | 4 | 5 | 4 |
| | 9 | 5 | 4 | 3 | 4 | 6 | 4 |
| | 10 | 6 | 5 | 4 | 2 | 4 | 3 |

| | | Distance from Trail (Richness) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Equestrian** | **1** | 4 | 3 | 3 | 3 | 2 | 3 |
| | **2** | 6 | 4 | 3 | 3 | 3 | 3 |
| | **3** | 4 | 2 | 3 | 2 | 3 | 3 |
| | **4** | 4 | 4 | 3 | 2 | 4 | 7 |
| | **5** | 5 | 5 | 4 | 2 | 1 | 4 |
| | **6** | 4 | 4 | 6 | 3 | 3 | 2 |
| | **7** | 3 | 4 | 4 | 3 | 2 | 2 |
| | **8** | 3 | 3 | 5 | 2 | 1 | 2 |
| | **9** | 6 | 4 | 2 | 4 | 4 | 3 |
| | **10** | 2 | 3 | 3 | 5 | 2 | 4 |

| | | Distance from Trail (Richness) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Baseline** | **1** | 7 | 5 | 6 | 7 | 6 | 11 |
| | **2** | 9 | 7 | 7 | 10 | 9 | 7 |
| | **3** | 5 | 7 | 11 | 6 | 7 | 9 |
| | **4** | 6 | 5 | 5 | 5 | 2 | 4 |
| | **5** | 3 | 4 | 4 | 4 | 5 | 5 |
| | **6** | 3 | 4 | 3 | 4 | 4 | 2 |
| | **7** | 4 | 2 | 4 | 3 | 4 | 2 |
| | **8** | 6 | 5 | 6 | 4 | 5 | 4 |
| | **9** | 4 | 5 | 5 | 5 | 7 | 7 |
| | **10** | 7 | 5 | 5 | 4 | 8 | 4 |

**Appendix D. Percent Coverage of Individual Quadrants in Trail-Influenced Environments and Baseline Areas.**

| | | Distance from Trail (% Coverage) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Hiking** | **1** | 61 | 74 | 51 | 90 | 88 | 69 |
| | **2** | 8 | 19 | 46 | 20 | 4 | 8 |
| | **3** | 33 | 8 | 51 | 70 | 56 | 79 |
| | **4** | 44 | 31 | 65 | 55 | 60 | 81 |
| | **5** | 45 | 85 | 46 | 42 | 24 | 73 |
| | **6** | 47 | 60 | 71 | 56 | 67 | 85 |
| | **7** | 89 | 72 | 96 | 63 | 53 | 15 |
| | **8** | 48 | 84 | 90 | 78 | 30 | 71 |
| | **9** | 29 | 85 | 66 | 28 | 38 | 14 |
| | **10** | 26 | 32 | 80 | 95 | 95 | 54 |

| | | Distance from Trail (% Coverage) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Multi-Use** | **1** | 62 | 61 | 58 | 77 | 6 | 2 |
| | **2** | 14 | 26 | 27 | 30 | 56 | 22 |
| | **3** | 41 | 60 | 40 | 30 | 28 | 53 |
| | **4** | 65 | 63 | 54 | 34 | 43 | 55 |
| | **5** | 22 | 26 | 26 | 48 | 39 | 14 |
| | **6** | 70 | 80 | 58 | 32 | 76 | 69 |
| | **7** | 67 | 69 | 51 | 70 | 62 | 48 |
| | **8** | 65 | 65 | 50 | 50 | 40 | 40 |
| | **9** | 53 | 8 | 10 | 28 | 41 | 20 |
| | **10** | 42 | 62 | 14 | 16 | 12 | 14 |

| | | Distance from Trail (% Coverage) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Biking** | **1** | 57 | 92 | 62 | 46 | 22 | 35 |
| | **2** | 24 | 64 | 38 | 42 | 14 | 12 |
| | **3** | 8 | 28 | 38 | 43 | 18 | 12 |
| | **4** | 28 | 22 | 36 | 38 | 10 | 30 |
| | **5** | 4 | 9 | 16 | 22 | 5 | 6 |
| | **6** | 16 | 53 | 24 | 20 | 22 | 10 |
| | **7** | 7 | 16 | 28 | 20 | 12 | 2 |
| | **8** | 6 | 28 | 26 | 27 | 28 | 22 |
| | **9** | 20 | 18 | 20 | 20 | 30 | 32 |
| | **10** | 36 | 28 | 28 | 34 | 17 | 16 |

| | Transect | Distance from Trail (% Coverage) 0m | 2.5m | 5m | 10m | 17.5m | 25m |
|---|---|---|---|---|---|---|---|
| **Equestrian** | **1** | 32 | 32 | 40 | 36 | 9 | 36 |
| | **2** | 55 | 50 | 52 | 74 | 76 | 61 |
| | **3** | 28 | 36 | 24 | 6 | 28 | 82 |
| | **4** | 26 | 23 | 26 | 8 | 14 | 38 |
| | **5** | 51 | 36 | 28 | 24 | 2 | 12 |
| | **6** | 16 | 11 | 40 | 12 | 7 | 5 |
| | **7** | 34 | 32 | 57 | 28 | 10 | 24 |
| | **8** | 36 | 68 | 32 | 24 | 4 | 32 |
| | **9** | 41 | 12 | 20 | 15 | 40 | 22 |
| | **10** | 80 | 82 | 12 | 26 | 48 | 20 |

| | Transect | Distance from Trail (% Coverage) 0m | 2.5m | 5m | 10m | 17.5m | 25m |
|---|---|---|---|---|---|---|---|
| **Baseline** | **1** | 39 | 43 | 50 | 61 | 57 | 64 |
| | **2** | 84 | 65 | 70 | 61 | 46 | 33 |
| | **3** | 31 | 54 | 48 | 34 | 60 | 31 |
| | **4** | 60 | 43 | 51 | 77 | 70 | 42 |
| | **5** | 30 | 34 | 38 | 42 | 42 | 24 |
| | **6** | 82 | 74 | 71 | 60 | 96 | 44 |
| | **7** | 54 | 60 | 59 | 88 | 88 | 84 |
| | **8** | 69 | 30 | 62 | 70 | 54 | 56 |
| | **9** | 37 | 52 | 74 | 72 | 57 | 54 |
| | **10** | 61 | 53 | 64 | 64 | 88 | 36 |

**Appendix E. Total Percent Coverage of Species Grouped by Sensitivity at Each Distance Measured. Data is Presented for Trail-Influenced Environments and Baseline Areas.**

| | | Distance from Trail (% Coverage) | | | | | |
|---|---|---|---|---|---|---|---|
| **Sensitivity** | **Trail Use Type** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **0** | Hiking | 1.63 | 4.36 | 0.76 | 1.51 | 2.72 | 9.84 |
| | Multi-Use | 8.02 | 4.62 | 6.70 | 2.41 | 0.00 | 12.46 |
| | Biking | 4.85 | 0.00 | 1.27 | 0.00 | 5.62 | 1.13 |
| | Equestrian | 23.56 | 10.47 | 7.85 | 27.67 | 26.47 | 10.54 |
| | Baseline | 1.83 | 0.79 | 1.02 | 1.59 | 0.61 | 1.71 |
| **1** | Hiking | 0.23 | 0.00 | 1.06 | 0.50 | 0.39 | 0.36 |
| | Multi-Use | 1.60 | 0.00 | 0.00 | 0.48 | 0.00 | 0.00 |
| | Biking | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | Equestrian | 0.50 | 0.52 | 0.00 | 0.00 | 0.00 | 0.00 |
| | Baseline | 0.00 | 0.20 | 0.68 | 0.32 | 0.61 | 0.00 |
| **2** | Hiking | 63.26 | 52.73 | 53.93 | 42.55 | 43.30 | 46.81 |
| | Multi-Use | 19.64 | 14.04 | 15.72 | 21.69 | 17.87 | 19.88 |
| | Biking | 49.03 | 27.09 | 29.11 | 37.50 | 44.38 | 41.24 |
| | Equestrian | 26.07 | 32.46 | 20.54 | 24.11 | 38.66 | 49.70 |
| | Baseline | 38.21 | 40.94 | 44.46 | 40.38 | 46.81 | 52.99 |
| **3** | Hiking | 4.19 | 11.82 | 12.69 | 8.71 | 5.24 | 2.37 |
| | Multi-Use | 46.29 | 34.81 | 31.96 | 27.23 | 45.41 | 29.38 |
| | Biking | 4.85 | 7.82 | 12.03 | 2.56 | 1.12 | 0.00 |
| | Equestrian | 21.05 | 25.39 | 57.10 | 33.20 | 18.91 | 25.00 |
| | Baseline | 16.64 | 14.57 | 19.93 | 21.30 | 20.21 | 19.02 |
| **4** | Hiking | 16.51 | 18.36 | 21.60 | 33.00 | 35.92 | 29.87 |
| | Multi-Use | 13.63 | 21.54 | 20.10 | 11.57 | 2.48 | 4.75 |
| | Biking | 8.74 | 29.89 | 18.99 | 20.51 | 10.11 | 12.43 |
| | Equestrian | 19.55 | 24.87 | 4.83 | 5.53 | 1.68 | 4.22 |
| | Baseline | 18.10 | 25.59 | 7.16 | 14.47 | 5.47 | 4.27 |
| **5** | Hiking | 14.19 | 12.73 | 9.97 | 13.74 | 12.43 | 10.75 |
| | Multi-Use | 10.82 | 25.00 | 25.52 | 36.63 | 34.24 | 33.53 |
| | Biking | 32.52 | 35.20 | 38.61 | 39.42 | 38.76 | 45.20 |
| | Equestrian | 9.27 | 6.28 | 9.67 | 9.49 | 14.29 | 10.54 |
| | Baseline | 25.23 | 17.91 | 26.75 | 21.94 | 26.29 | 22.01 |

**Appendix F. Evenness Values within Each Quadrant for Trail-Influenced Environments and Baseline Areas.**

| | | Distance from Trail (Evenness) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Hiking** | **1** | 0.57 | 0.75 | 0.73 | 0.59 | 0.64 | 0.70 |
| | **2** | 0.67 | 0.74 | 0.63 | 0.72 | 1.00 | 0.99 |
| | **3** | 0.86 | 0.95 | 0.88 | 0.76 | 0.83 | 0.54 |
| | **4** | 0.44 | 0.78 | 0.65 | 0.84 | 0.51 | 0.10 |
| | **5** | 0.31 | 0.83 | 0.43 | 0.72 | 0.74 | 0.87 |
| | **6** | 0.55 | -- | 0.60 | 0.49 | 0.68 | 0.73 |
| | **7** | 0.69 | 0.78 | 0.82 | 0.72 | 0.84 | 0.89 |
| | **8** | 0.72 | 0.63 | 0.89 | 0.82 | 0.86 | 0.65 |
| | **9** | 0.88 | 0.61 | 0.55 | 0.69 | 0.84 | 0.98 |
| | **10** | 0.96 | 0.75 | 0.54 | 0.58 | 0.64 | 0.69 |

| | | Distance from Trail (Evenness) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Multi-Use** | **1** | 0.44 | 0.57 | 0.78 | 0.82 | 0.79 | -- |
| | **2** | 0.59 | 0.63 | 0.61 | 0.82 | 0.69 | 0.97 |
| | **3** | 0.70 | 0.85 | 0.87 | 0.89 | 0.81 | 0.92 |
| | **4** | 0.86 | 0.55 | 0.91 | 0.87 | 0.85 | 0.99 |
| | **5** | 0.83 | 0.91 | 0.90 | 0.87 | 0.88 | 0.72 |
| | **6** | 0.86 | 0.63 | 0.88 | 0.94 | 0.84 | 0.79 |
| | **7** | 0.90 | 0.89 | 0.85 | 0.89 | 0.88 | 0.76 |
| | **8** | 0.91 | 0.92 | 0.97 | 0.88 | 0.72 | 0.83 |
| | **9** | 0.71 | 1.00 | -- | 0.72 | 0.74 | 0.97 |
| | **10** | 0.79 | 0.88 | 0.99 | 0.95 | 1.00 | -- |

| | | Distance from Trail (Evenness) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Biking** | **1** | 0.74 | 0.85 | 0.65 | 0.97 | 0.99 | 0.81 |
| | **2** | 0.93 | 0.96 | 0.83 | 0.84 | 0.98 | 0.79 |
| | **3** | 1.00 | 0.90 | 0.84 | 0.90 | 0.50 | 1.00 |
| | **4** | 0.94 | 0.83 | 0.93 | 0.94 | 0.72 | 0.69 |
| | **5** | -- | 0.83 | 0.88 | 0.80 | 0.96 | -- |
| | **6** | 0.71 | 0.83 | 0.98 | 0.92 | 0.80 | -- |
| | **7** | 0.87 | 0.95 | 0.81 | 0.96 | 0.52 | -- |
| | **8** | 1.00 | 0.86 | 0.93 | 0.76 | 0.59 | 0.86 |
| | **9** | 0.84 | 0.83 | 0.94 | 0.92 | 0.92 | 0.75 |
| | **10** | 0.89 | 0.89 | 0.94 | 0.52 | 0.77 | 0.89 |

| | | Distance from Trail (Evenness) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Equestrian** | **1** | 0.59 | 0.55 | 0.73 | 0.97 | 0.50 | 0.62 |
| | **2** | 0.91 | 0.94 | 0.80 | 0.23 | 0.54 | 0.86 |
| | **3** | 0.76 | 0.99 | 0.52 | 0.92 | 0.80 | 0.91 |
| | **4** | 0.87 | 0.65 | 0.82 | 1.00 | 0.92 | 0.96 |
| | **5** | 0.64 | 0.91 | 0.65 | 0.65 | -- | 0.96 |
| | **6** | 0.88 | 0.84 | 0.96 | 0.92 | 0.87 | 0.72 |
| | **7** | 0.62 | 0.59 | 0.76 | 1.00 | 0.97 | 0.41 |
| | **8** | 0.89 | 0.32 | 0.84 | 0.65 | -- | 0.20 |
| | **9** | 0.90 | 0.81 | 0.88 | 0.85 | 0.75 | 0.91 |
| | **10** | 0.54 | 0.32 | 1.00 | 0.88 | 0.54 | 0.92 |

| | | Distance from Trail (Evenness) | | | | | |
|---|---|---|---|---|---|---|---|
| | **Transect** | **0m** | **2.5m** | **5m** | **10m** | **17.5m** | **25m** |
| **Baseline** | **1** | 0.84 | 0.82 | 0.83 | 0.89 | 0.76 | 0.81 |
| | **2** | 0.87 | 0.82 | 0.89 | 0.73 | 0.95 | 0.90 |
| | **3** | 0.89 | 0.92 | 0.90 | 0.91 | 0.89 | 0.88 |
| | **4** | 0.68 | 0.79 | 0.62 | 0.83 | 0.59 | 0.89 |
| | **5** | 0.70 | 0.77 | 0.76 | 0.92 | 0.77 | 0.91 |
| | **6** | 0.98 | 0.59 | 0.23 | 0.71 | 0.50 | 0.44 |
| | **7** | 0.91 | 0.92 | 0.68 | 0.74 | 0.78 | 0.79 |
| | **8** | 0.64 | 0.74 | 0.84 | 0.78 | 0.93 | 0.79 |
| | **9** | 0.78 | 0.72 | 0.76 | 0.81 | 0.79 | 0.62 |
| | **10** | 0.72 | 0.67 | 0.81 | 0.82 | 0.93 | 0.78 |

**Appendix G. Raw Data of Trail Characteristics and Environmental Variables.**

| | | Variables | | | | | |
|---|---|---|---|---|---|---|---|
| **Trail Use Type** | **Transect** | **Trampled width (m)** | **Tread width (m)** | **Trail depth (in)** | **Trail Slope (°)** | **Transect Slope (°)** | **Elevation (m above sea level)** |
| **Hiking** | 1 | 1.8 | 1 | 2 | 1 | 6 | 352 |
| | 2 | 1.6 | 1.3 | 3 | 4 | 1 | 346 |
| | 3 | 2 | 1 | 4 | 0 | 6 | 353 |
| | 4 | 1.3 | 1 | 3.5 | 1 | 3 | 351 |
| | 5 | 1 | 1 | 3 | 1 | 1 | 352 |
| | 6 | 2 | 1 | 1 | 1 | 7 | 348 |
| | 7 | 1.3 | 1 | 2.25 | 2 | 2 | 345 |
| | 8 | 1.5 | 1 | 3.25 | 1.5 | 5 | 349 |
| | 9 | 1.3 | 0.9 | 3 | 1 | 4 | 359 |
| | 10 | 1.7 | 1 | 3 | 1 | 7 | 349 |
| **Multi-Use** | 1 | 2.3 | 1.6 | 5 | 3 | 5 | 351 |
| | 2 | 2.5 | 1.8 | 4 | 5 | 1 | 350 |
| | 3 | 2.2 | 1.8 | 5 | 2 | 8 | 355 |
| | 4 | 2.4 | 1.4 | 4.5 | 2 | 8 | 352 |
| | 5 | 2.3 | 1.8 | 4 | 1 | 12 | 349 |
| | 6 | 2.4 | 1.9 | 5 | 2 | 3 | 353 |
| | 7 | 2.5 | 1.7 | 6 | 1 | 10 | 347 |
| | 8 | 2.2 | 1.9 | 3 | 2 | 5 | 353 |
| | 9 | 2.4 | 1.4 | 5 | 1 | 8 | 352 |
| | 10 | 2.3 | 1.3 | 6 | 1 | 1 | 348 |
| **Biking** | 1 | 2.6 | 1.9 | 5 | 4 | 10 | 351 |
| | 2 | 1.2 | 1.2 | 7 | 8 | 7 | 351 |
| | 3 | 2.3 | 1.4 | 3 | 3 | 2 | 351 |
| | 4 | 3.3 | 2.3 | 2 | 8 | 1 | 350 |
| | 5 | 1.9 | 1.9 | 0.5 | 7 | 3 | 354 |
| | 6 | 1.5 | 0.9 | 3 | 3 | 4 | 360 |
| | 7 | 5 | 3 | 4.5 | 2 | 8 | 357 |
| | 8 | 2.2 | 1.3 | 3.5 | 3 | 1 | 362 |
| | 9 | 1.9 | 0.9 | 2 | 0 | 4 | 365 |
| | 10 | 1.4 | 0.9 | 4 | 0 | 5 | 356 |

| | | Variables | | | | | |
|---|---|---|---|---|---|---|---|
| **Trail Use Type** | **Transect** | **Trampled width (m)** | **Tread width (m)** | **Trail depth (in)** | **Trail Slope (°)** | **Transect Slope (°)** | **Elevation (m above sea level)** |
| **Equestrian** | 1 | 1.4 | 1.1 | 4 | 2 | 2 | 378 |
| | 2 | 0.8 | 0.5 | 4 | 5 | 2 | 373 |
| | 3 | 0.8 | 0.5 | 6 | 2 | 2.5 | 373 |
| | 4 | 2.6 | 1.4 | 6 | 10 | 7 | 363 |
| | 5 | 2.4 | 2 | 0 | 1 | 10 | 354 |
| | 6 | 2.7 | 1.9 | 12 | 2 | 4 | 356 |
| | 7 | 2.7 | 2.7 | 6 | 5 | 6.5 | 352 |
| | 8 | 1.4 | 1.4 | 0 | 1 | 5.5 | 355 |
| | 9 | 2.1 | 1.5 | 8 | 1.5 | 4 | 361 |
| | 10 | 1.8 | 1.7 | 8 | 5.5 | 0.5 | 373 |
| **Baseline** | 1 | n/a | n/a | n/a | n/a | 4 | 327 |
| | 2 | n/a | n/a | n/a | n/a | 9 | 337 |
| | 3 | n/a | n/a | n/a | n/a | 4 | 345 |
| | 4 | n/a | n/a | n/a | n/a | 1 | 374 |
| | 5 | n/a | n/a | n/a | n/a | 1 | 387 |
| | 6 | n/a | n/a | n/a | n/a | 5 | 289 |
| | 7 | n/a | n/a | n/a | n/a | 3 | 303 |
| | 8 | n/a | n/a | n/a | n/a | 3 | 348 |
| | 9 | n/a | n/a | n/a | n/a | 5 | 370 |
| | 10 | n/a | n/a | n/a | n/a | 5 | 344 |

**Appendix H. Aspect of Transects in Trail-Influenced Environments and Baseline Areas.**

| Transect Aspect of Different Users | | | | | |
|---|---|---|---|---|---|
| **Transect** | **Hiking** | **Multi-Use** | **Biking** | **Equestrian** | **Baseline** |
| 1 | S | -- | S | -- | S |
| 2 | S | N | S | S | S |
| 3 | S | N | -- | S | S |
| 4 | N | N | -- | S | -- |
| 5 | N | S | N | N | -- |
| 6 | N | N | N | N | N |
| 7 | N | S | N | N | N |
| 8 | N | -- | -- | N | N |
| 9 | N | N | -- | S | N |
| 10 | S | S | N | N | N |

N denotes a North facing slope, S denotes a South facing slope and -- denotes that there was no observable aspect.

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