The Effect of Vertical Stratification and Forest Management on Beetle (Insecta: Coleoptera) and Spider (Arachnida: Araneae) Communities in a Temperate Forest

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Abstract

Forest canopies may harbour up to 50% of the world's diversity and since the canopy and forest floor have different architectures, the structure of arthropod assemblages may vary by stratum. This has implications for forest management since long-term logging practices can change the composition and structure of northeastern North American forests, and arthropods are sensitive to these differences. While canopy sampling is becoming more common, the interaction between forest management and vertical stratification in temperate forests in North America has remained relatively unexplored. The objective of this thesis is to explore and quantify the structure of arthropod communities in two strata within the context of forest management (time-since harvest and silvicultural technique).

I used beetles (Coleoptera) and spiders (Araneae), important components of forest ecosystems, to test effects of vertical stratification and forest management effects on biodiversity and community structure. I collected 586 species of beetles and spiders, representing 11 137 individuals, over two field seasons in a deciduous hardwood forest in Southern Québec. Beetle diversity was influenced by height, with the understorey harbouring a higher abundance and species richness than the upper canopy. Spider diversity was not affected by height or forest management. Both beetle and spider community compositions were strongly affected by height, with each stratum supporting a different community. Beetle assemblages differed between forest management sites (both time-since harvest and silvicultural technique treatments), but this was largely influenced by the Latridiidae, the most commonly collected family. Spider assemblages did not change with forest management treatment. Responses were taxon dependent, at both the Order and Family level.

I suggest that ongoing biodiversity assessments and future ecological work in temperate forests should include canopy sampling, as omitting it might lead to an inaccurate representation of forest systems. Forest management should also be limited in frequency (time-since harvest) and intensity (silvicultural techniques), since canopy closure seems to be an important factor in shaping arthropod communities.

Résumé

Les canopées forestières peuvent abriter jusqu'à 50 % de la diversité mondiale et comme la canopée et le sol forestier ont des architectures différentes, la structure des assemblages d'arthropodes peut varier selon la strate. Cela a des répercussions sur l'aménagement forestier puisque les pratiques d'exploitation à long terme peuvent modifier la composition et la structure des forêts du nord-est de l'Amérique du Nord, et les arthropodes sont sensibles à ces différences. Bien que l'échantillonnage du couvert forestier soit de plus en plus courant, l'interaction entre l'aménagement forestier et la stratification verticale dans les forêts tempérées en Amérique du Nord est demeurée relativement inexplorée. L'objectif de cette thèse est d'explorer et de quantifier la structure des communautés d'arthropodes en deux strates dans le contexte de la gestion forestière (temps depuis la récolte et technique sylvicole).

J'ai utilisé des coléoptères (Coleoptera) et des araignées (Araneae), composantes importantes des écosystèmes forestiers, pour tester les effets de la stratification verticale et les effets de la gestion forestière sur la biodiversité et la structure des communautés. J'ai récolté 586 espèces de coléoptères et d'araignées, représentant 11 137 individus, sur deux saisons de récolte dans une forêt de feuillus à feuilles caduques du sud du Québec. La diversité des coléoptères a été influencée par la hauteur, avec le sous-étage abritant une abondance et une richesse en espèces plus élevées que le couvert supérieur. La diversité des araignées n'a pas été affectée par la hauteur ou la gestion forestière. La composition des communautés de coléoptères et d'araignées a été fortement influencée par la hauteur, chaque strate supportant une communauté différente. Les assemblages de coléoptères différaient d'un site d'aménagement forestier à l'autre (à la fois depuis la récolte et les traitements techniques sylvicoles), mais ils étaient largement influencés par les Latridiidae, la famille la plus fréquemment récoltée. Les assemblages d'araignées n'ont pas changé avec le traitement de gestion forestière. Les réponses dépendaient du taxon, tant au niveau de l'Ordre qu'au niveau de la Famille.

Je suggère que les évaluations de la biodiversité et les travaux écologiques futurs dans les forêts tempérées devraient inclure l'échantillonnage du couvert forestier, car son omission pourrait conduire à une représentation imprécise des

systèmes forestiers. L'aménagement forestier devrait également être limité en fréquence (temps écoulé depuis la récolte) et en intensité (techniques sylvicoles), car la fermeture du couvert forestier semble être un facteur important dans la formation des communautés d'arthropodes.

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Figure 2.2. Map of the Kenauk Nature property with 2017 sampling sites. Shaded areas correspond to logging activity, with blue areas representing strip cutting, green areas representing selection cutting, and grey areas representing no logging activities. Sites sampled during 2017 are represented by hexagons and coloured based on which type of logging they are located in. There are six sites representing each category. A close up of the twelve northern sites is given inset.

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Thesis format

This thesis is organized as two chapters written in manuscript style. Chapter 1 is a literature review which introduces the key components of the project, in addition to the research questions and objectives. Chapter 2 contains the body of the work, and will be submitted to the journal *Insect Conservation and Diversity.*

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Contribution of authors

I wrote the literature review and original manuscript in this thesis. My supervisor, Christopher Buddle, is a co-author on Chapter 2 and provided the experimental design and conceptual building of this project. He also provided me with edits on the manuscript. All other, non-writing contributions are outlined in the acknowledgments at the end of Chapter 2.

Chapter 1: Introduction and Literature Review

1.1 Introduction

1.1.1 Thesis rationale

Past research has illustrated that arthropods are sensitive to differences between the strata (vertical stratification) in temperate forests (e.g. Maguire et al. 2014; Larrivée and Buddle 2009; Vance et al. 2007; Ulyshen et al. 2010; Oguri et al. 2014). Many reasons have been put forth, including differences in physical (habitat complexity, microhabitat structure) and environmental (sun, wind, and temperature) characteristics (Shaw 2004; Lowman and Rinker 2004; Docherty and Leather 1997; Halaj et al. 2000; Maguire et al. 2014; Oguri et al. 2014). This vertical stratification has implications for forest management, as long-term logging practices have led to changes in the composition and physical structure of northeastern North American forests (Boucher et al. 2009; Graignic et al. 2016), something that is predicted to affect arthropod communities (Maeto et al. 2002).

However, we still do not fully know or understand the long-term effects of logging on the flora and fauna of Southern Québec; only a handful of studies have been conducted on arthropods in and around the Southern Québec region (Moore et al. 2004; Zeran et al. 2006; Zeran et al. 2007). This project will shed light on the relationship between canopy sampling and forest management. I use two different silviculture techniques (selection and strip cutting) and sample in stands cut at two different times (1989-1994 and 2001-2005) to answer my questions.

By doing so, I provide an overall, well rounded understanding of forest management effects, since I am looking at multiple forest management treatments at once. Where previous work has focussed primarily on forest management or vertical stratification (rarely both), I have combined the two, effectively increasing our understanding of the arthropod responses to anthropogenic change on the Kenauk Institute property. The Kenauk Institute aims to find a balance between wildlife protection and preservation and forest management and as such, this work provides much needed information on how arthropod communities are reacting to different sylvicultural techniques. This can then be used as a representative of temperate forests, and will ultimately be useful in predicting how other temperate areas may react to forest management practices similar to those at the Kenauk Institute.

1.2 The use of beetles and spiders in ecological studies

Arthropods are an ideal model taxon and bio-indicator of ecosystem integrity, health, and change, since they reproduce quickly, are sensitive to abiotic changes, are highly abundant, provide many ecosystem services and can be sampled and preserved efficiently and consistently (e.g., Malecque et al. 2006; Malecque et al. 2009)

Previous ecological work has highlighted the importance of using arthropods as bio-indicators, specifically beetles and spiders (Malecque et al. 2009). Beetles are an important component of global diversity estimates since they are widespread and abundant (Leksono et al. 2005a). Even though precise estimates of total species numbers are unavailable, beetles are still considered the most species-rich group of insects (Leksono et al. 2005a). Since Coleoptera are so diverse, they fulfill a wide range of roles in the ecosystem, ranging from phytophagous to predaceous eating habits (Maguire et al. 2014; Dajoz 2007). Spiders are one of the most prevalent groups of predatory arthropods in terms of both biomass and species diversity (Oguri et al. 2014) and they play a role in structuring tree canopy food webs by acting as both predator and prey (Larrivée and Buddle 2009). Spiders seem to be sensitive to habitat change (Robinson 1981), occupy a wide range of habitats, are easily sampled and identified, and have been used effectively in the context of anthropogenic and natural disturbances (e.g., Buddle et al. 2000).

Multiple beetle groups, such as carabid and cerambycid beetles, and spiders have been found to respond to small-scale, disturbance-induced habitat changes caused by forest management and can therefore be used to infer the ecological suitability of forest management treatments (Moore et al. 2004; Maeto et al. 2002; Pearce and Venier 2006; Malecque et al. 2009). For example, Magura et al. (2003) found that forest fragmentation due to forest management can decrease the abundance of large-bodied, poorly dispersing carabid beetles, while also increasing the number of open habitat specialists. Further, it has been shown that cerambycid beetle larvae are dependent on wood and that the adults are associated with flowering plants, coarse woody materials, and old trees that are often seen in old growth forests (Maeto et al. 2002; Ohsawa 2004; Malecque et al. 2009). Due to these ecological responses, carabid and cerambycid beetles are commonly used families as bio-indicators of ecosystem change (Malecque et al. 2009).

Spiders are also responsive to natural and anthropogenic disturbances (Pearce and Venier 2006) and have been found to be bio-indicators of ecosystem changes caused by clear-cutting, forest fires, and complexity of forest stands (Pearce and Venier 2006; Malecque et al. 2009; Buddle et al. 2000). For example, Paradis and Work (2011) found that spider assemblages were drastically affected by forest management, and they attribute these community differences to the age of the stand prior to logging (old forests had a stronger community shift than younger forests), degree of paludification (rise in water table), and to remnant basal area. Put together, arthropods are an important component of ecosystem health and have been found to consistently respond to ecosystem disturbances. As such, they are often used in ecological studies, especially in ones where the goal is to understand the effects of forest management on ecosystem health and integrity.

1.3 Vertical stratification

1.3.1 Vertical stratification in tropical forests

Vertical stratification is characterized as organisms distributing themselves vertically (e.g., plankton in a lake, arthropods in a tree) (Basset et al. 2003a). Vertical stratification of arthropods has been a strong focus of tropical ecology research for decades (Bates 1944); this early work was conducted by felling trees and mainly focused on economically important arthropod species (Basset et al. 2003a). It was only in the 1970s that canopy sampling became more common for ecological work, as canopy access improved drastically with the introduction of small towers and, most notably, canopy fogging with insecticide (Basset et al. 2003b; Erwin 1982).

With this sudden increase in canopy sampling, researchers found distinct differences between the understorey and the upper canopy. For example, Haddow (1961) conducted a series of research projects in Uganda, and found that mosquitoes were sensitive to vertical gradients at the species level, and even between male and female mosquitoes of the same species. Abiotic factors, such as wind and light exposure, and biotic factors, such as tree architecture and resource availability, are most likely the main drivers in causing vertical stratification in arthropods in tropical forests; these factors experience a sharp gradient in tropical forests that is much more apparent than in temperate forests (Basset et al. 2003c; Basset et al. 2003b). Ultimately, these differences laid the foundation for many ecological questions, especially with a focus on estimating arthropod diversity in forest systems, as well as global diversity estimates (Vance et al. 2007).

Erwin (1982) predicted that as many as two-thirds of beetle species are found in the canopies of tropical forests and by estimating their host specificity guessed that roughly 30 million species of arthropods exist in tropical forests. While this is likely a gross overestimation of the true number (see Odegaard et al. 2000 and Novotny et al. 2002), the abundance and species richness of arthropods in tropical canopies still seem to be considerably higher than those of the forest floor (Hirao et al. 2009; Ulyshen et al. 2010). While canopy work has been done in tropical forests for many decades, canopy sampling in the temperate forest has only been recently explored.

1.3.2 Vertical stratification in temperate forests

The same processes that affect arthropods in tropical forests have been shown to influence arthropod vertical distribution in temperate forests. The canopy and forest floor have different architectures, which could be a factor in structuring arthropod assemblages in temperate forests (Oguri et al. 2014). Foliage complexity of the canopy, such as foliage density and number of leaves and branchlets, and structural complexity of forest floor litter and understorey vegetation, such as litter depth and shape, and ground cover by plants, can influence arthropod distributions (Oguri et al. 2014). These physical differences seen between the strata provide different microhabitats for arthropods, which have been found to influence arthropod community structure (Oguri et al. 2014). For example, Halaj et al. (2000) found that manipulating habitat complexity strongly affected arthropod distribution in the canopy, while Uetz (1975) found that the physical structure of leaf litter is an important factor in shaping spider biodiversity and community structure on the forest floor.

In addition, the distribution of species in forests is expected to vary because of different abiotic and environmental factors associated with each vertical stratum (Maguire et al. 2014). The forest canopy is more variable than the understorey due to abiotic factors such as radiation, moisture, temperature, and wind (Maguire et al. 2014). Leksono et al. (2005b), for example, found that phytophagous beetles showed a stratified distribution and attribute this stratification to the herbivorous beetles preferring various vertical gradients of light, temperature, and moisture than other feeding guilds. Vodka and Cizek (2013) investigated saproxylic beetle diversity in multiple strata and found that the most important factors in beetle distribution are sunlight availability and, in turn, evaporation.

This vertical stratification can also be attributed to biotic factors, such as tree architecture, differences in foliage quality, and type and amount of foliage, fungi, dead wood, and carrion (Maguire et al. 2014). For example, Leksono et al. (2005a) found that, while abundance of phytophagous beetles did not change among the strata, scavengers were more abundant on the forest floor than the canopy. They attribute this to the wealth of resources available to scavenging beetles on the forest floor (fallen fruit, animal waste, leaf litter) (Leksono et al. 2005a).

As improved methods for accessing the canopy are made available and more studies investigate stratification in temperate forests, it is becoming clear that arthropods are sensitive to these strata differences (Vance et al 2007). While arthropods are generally equally or more diverse in the canopy than near the ground in tropical rainforests, the opposite pattern is often seen in temperate forests (Ulyshen et al. 2010).

In Southern Québec, quite a few studies have been conducted regarding vertical stratification of arthropods. Maguire et al. (2014) found no differences in beetle diversity between the strata, but did find distinct communities. In comparison, Aikens and Buddle (2012) found that both beetle and spider densities decreased with increasing height in sugar maples (*Acer saccharum* Marshall) and concluded that arthropod assemblages are heterogeneous even at a scale as small as a few metres. Finally, Schroeder et al. (2009) studied the effect of forest gaps and height on beetles and found that the upper canopy harboured a higher diversity of beetles and a different community assemblage than the forest floor. Even though there is some disagreement about patterns of distribution among the strata, these studies, along with many others, demonstrate that diversity is not limited to the understorey in temperate forests.

1.4 Forest management in temperate forests

Forest managers often classify forests into two categories: old growth forest and second-growth forest. Old growth forests are uncut forests that have not been disturbed by human activities or by natural disasters. Second-growth forests are a result of secondary succession; these forests develop after a large-scale disturbance occurred, such as logging or forest fire. These second-growth forests are of interest to foresters and biologists alike: regeneration time is important for foresters, as they need to know when they can return to a specific stand and harvest again, and for biologists, as they want to know how quickly secondary succession (return of previous communities) occurs (Miller Jr and Spoolman 2012). Together, biologists and forest managers have called for more sustainable forest management practices (Miller Jr and Spoolman 2012); this has come in the form of increased logging intervals (longer time between cuts) (Angers et al. 2005) and in new, low-impact logging practices (selection and strip cutting) (Doyon et al. 2005).

Canada is the world’s largest exporter of softwood lumber and the largest producer and exporter of newsprint, making the timber industry an important part of Canada’s economy (Wallace 2000). The majority of forest management practices in

Southern Québec are aesthetically and physically less damaging than clear cutting, with selection cutting being the most commonly used harvesting method (Malo and Messier 2011). However, we still don't fully know or understand the long-term effects of selection cutting on the flora and fauna of Southern Québec; only a handful of studies have been conducted on arthropods in and around the Southern Québec region (Moore et al. 2004; Zeran et al. 2006; Zeran et al. 2007).

1.4.1 Forest management practices and their effects on diversity

There are multiple silviculture techniques but for the sake of this literature review I will focus on selection cutting and strip cutting, as these two techniques are practiced at the Kenauk Institute site and will be the focus of our study outlined in Chapter 2.

Selection cutting is characterized by the removal of single trees and affects up to 30% of the initial basal area (Angers et al. 2005). This technique has been the most widely used and accepted forest management technique in Québec since the 1990s (Bédard and Majcen 2003). It is thought to be the least harmful silviculture technique in terms of biodiversity preservation since it attempts to mimic the natural disturbance regime in northeastern North America (Jones and Thomas 2004; Angers et al. 2005). However, there are no consistent results concerning the effect of selection cutting on the remaining, uncut trees, and on various taxa of organisms found in those forests.

In comparison, strip cutting is characterized as the complete removal of all trees in a single, narrow strip, with the aim that the strip is narrow enough for regeneration after a few years (Miller Jr and Spoolman 2012). After this regeneration, loggers cut another strip next to the original strip (Miller Jr and Spoolman 2012). This silviculture system, along with selection cutting, is considered low-impact and commonly used as an alternative to large-scale clear cuts (Doyon et al. 2005).

Even though both selection cutting and strip cutting are considered more sustainable forms of logging, previous research on the effects of both silviculture systems on biodiversity have been contradictory. Angers et al. (2005) found that there were remarkable structural differences between old growth and selectively cut forests. They go on to explain that while selection cutting may seem close to the natural disturbance regime, re-harvesting the same stand might lead to homogenization of forest structure, leading to a reduction in biodiversity. In contrast, Bédard and Majcen (2003) found that the basal area of sugar maples in selectively cut forests could return to the pre-cut amount in only 20 years.

While this is promising, Graignic et al. (2016) found a bottleneck effect occurring in stands that were selectively cut. They warn of the cumulative effect of reoccurring applications of selection cutting on these stands and how this could lead to deleterious alleles. Malo and Messier (2011) found that fine root growth of the remaining sugar maples in selectively cut forests was reduced fivefold by machinery tracks. This reduction may have a significant effect on the future growth and survival of these remaining trees.

As for arthropods, the effect of forest management on diversity in temperate forests has been a topic of interest for some time. Moore et al. (2004) investigated carabid beetle responses to forest management while Morin et al. (2015) were interested in cerambycid and scotylid (bark) beetles. These projects found contrasting results: forest management had no diversity effect on carabid beetles (Moore et al. 2004) but Morin et al. (2015) did find a higher abundance and diversity of bark beetles in managed stands. Similarly, Maeto et al. (2002) investigated cerambycid diversity and found a higher abundance and diversity in unmanaged stands. These can be explained by taxon-specific characteristics: carabid beetles consist of many open-habitat specialists and therefore logging may not drastically change species richness or abundance (Magura et al. 2003), bark beetles feed on injured and damaged trees, which are bountiful in managed stands (Morin et al. 2015), and cerambycid beetles prefer larger trees due to their ecology and natural history, which are commonly found in unmanaged stands (Maeto et al. 2002).

In Southern Québec and its surrounding areas, relatively little work has been done on arthropods. A few studies by Zeran et al. (2006, 2007), conducted in Southern Ontario, found contrasting results for fungivorous beetle communities. Sap beetles (Coleoptera: Nitidulidae) were found to differ in diversity and species composition between managed and old growth forests (Zeran et al. 2006), whereas no differences were seen for fungivorous Coleoptera (Zeran et al. 2007).

These projects focused primarily on the type of forest management type at one intensity level. There are several projects that compare different intensity levels of forest management applications relative to biodiversity. One well known project in Alberta, the Ecosystem Management by Emulating Natural Disturbance (EMEND) project, aims to compare clear cuts and partial retention cuts (similar to selection cutting), at different retention levels, and with different taxa. Harrison et al. (2005) found that songbirds declined or disappeared entirely from stands with low retention levels (10-20% of pre-cut amount of trees) and suggest that retention levels of 50-75% may be adequate in conserving songbird communities. Lazaruk et al. (2005) found that ectomycorrhizae diversity and community structure changed drastically with increasing amount of tree removal. These articles, along with many others, illustrate that if tree removal is light enough, biodiversity effects can be minimal.

There is no shortage of literature on forest management and through this, it seems as though the effect of selection and strip cutting is taxon-specific (carabid beetles in Moore et al. 2004; bark beetles in Morin et al. 2015; cerambycid beetles in Maeto et al. 2002). Regardless, these studies begin to illustrate a pattern in arthropods: diversity and community structure may shift due to anthropogenic change to habitat structures. As such, the use of arthropods may be beneficial in understanding the ecological suitability of forest management techniques.

1.4.2 Succession associated with forest management

Succession is the change in species composition over time caused by a disturbance (Drury and Nisbet 1973). This succession continues until the species structure best suits the regional climate and site (Drury and Nisbet 1973). Succession can be caused by natural disturbances, such as forest fires and wind storms, or through anthropogenic means, as in forest management. Forest managers must aim to cut stands with consideration to succession if they want to be able to re-harvest the same stand in later years (Angers et al. 2005). Forest managers and biologists have called for more sustainable silvicultural practices because regeneration of both market trees and of floral and faunal communities are of high concern (Miller Jr and Spoolman 2012).

Similarly to management techniques, it seems as though different taxa react differently to anthropogenic succession. Niemelä et al. (1996) found a difference in the community structures of ants, carabid beetles and spiders between stands of different successional stages. Jacobs et al. (2007) found that saproxylic beetles underwent community changes between successional stages in a logged stand and attribute this to the changes in dead wood. Finally, Koivula et al. (2002) found that successional age of stands associated with forest management affected the abundance and the species richness of carabid beetles; community changes started to occur once the forest canopy began closing (open habitat specialists became less abundant after closure).

For spiders, and to a lesser extent beetles (Koivula et al. 2002), canopy closure is an important driver in structuring community assemblages in the understorey stratum (Huhta 1971; McIver et al. 1992; Buddle et al. 2000). Canopy cover influences the microenvironmental conditions found on the forest floor; closed canopies prevent large fluctuations of light, moisture, temperature, and wind, allowing for the colonization and success of more intolerant species (McIver et al. 1992). Previous work has shown that spider communities diverge away from their original state for the first 7 years after a cut (Huhta 1971) but then start to return to pre-cut assemblages once the canopy begins closing, after about 15 years of growth (Buddle et al. 2000).

A few studies have investigated canopy cover effects on arthropods due to forest management. For example, Buddle et al. (2000) found that spider assemblages differed between successional stages of managed stands, and stress that these differences were caused by the amount of canopy cover present. Similarly, Klimaszewski et al. (2008) investigated whether gap size, and in turn, canopy closure, affected staphylinid beetle diversity and composition. They found that beetle composition in stands with many, smaller gaps were more similar to uncut stands than stands with fewer, larger gaps were (Klimaszewski et al. 2008).

Arthropods may be sensitive to the successional gradient created by forest management practices, as some taxa may be dependent on canopy gaps (Koivula et al. 2002). Again, responses to anthropogenic change seem to be taxon-specific (carabid beetles in Niemelä et al. 1996; carabid beetles in Koivula et al. 2002; saproxylic beetles in Jacobs et al. 2007; spiders in Buddle et al. 2000). For ecological studies especially, recognizing natural history and the functional traits of the study taxon is vital, as these are important drivers in the distribution, range, and behaviour of taxa. As such, using a variety of taxa, representing multiple functional groups, may be useful in understanding patterns of community changes caused by succession associated with forest management.

1.5 Vertical stratification and forest management

This literature review has so far focused on a summary of past research on the biodiversity of canopy arthropods and effects of forests management on arthropods in temperate forests. There has been very little previous work done in temperate forests around the world, let alone North America, that focuses on both vertical stratification and forest management. One of the first studies to synthesise both topics was conducted in present day Czech Republic by Simandl in 1993, who found that stand age affected most canopy arthropod taxa (Order and Family level) in terms of community composition.

In North America, Schowalter et al. (1981) found that family-level community changes of canopy arthropods occurred between old growth and clear-cut sites. They also noted that this change in community structure alters the flow of nutrients through the arthropod community (Schowalter et al. 1981). Schowalter (1995) compared herbivory rates and community structures of canopy arthropods in variously managed sites and found that old growth canopies supported the greatest diversity of arthropods and was the least variable in arthropod diversity and abundance.

Canopy arthropod responses to environmental changes may have drastic effects on nutrient cycling and forest productivity (Schowalter 1986) but responses to changing forest conditions due to forest management practices remain relatively unknown (Schowalter 1995). Understanding the relationship between vertical stratification and forest management is important if foresters and biologists alike are interested and invested in conserving the fauna found in the different strata, due to their effects on ecosystem services, for example. Invertebrates as a whole affect most, if not all, ecosystem services (Prather et al. 2013). They play an important role in providing supporting services (accelerating primary production, through pollination and seed dispersal, decomposition, and nutrient cycling), in provisioning services (acting as food and used in pharmaceuticals) and in regulating services (food web stability) (Prather et al. 2013).

This research project aims to understand the link between vertical stratification and forest management, using the community structure of key arthropods as a 'response variable'. Since forest management seems to alter arthropod assemblages at the understorey level due to canopy removal (see Buddle et al. 2000; Klimaszewski et al. 2008; Koivula et al. 2002), and the removal of this canopy cover also eliminates the unique arthropod communities found in this stratum, it is important to consider canopy sampling when working in managed stands. Sampling in multiple sites and strata can help with new conservation management in the area, since stratum specialist insects will generally have higher conservation priorities (Leksono et al. 2005a).

1.6 Research objectives

The overall goal of this thesis is to understand arthropod community dynamics in a northern temperate forest. This work was done at the Kenauk Institute (of which roughly one third is owned by Nature Conservancy Canada) and the broader value is to provide data to the Institute and Nature Conservancy Canada, as both want to find scientifically sound ways to prioritize the conservation of rare and endangered species. Chapter 1 was a review of the literature on vertical stratification in both tropical and temperate forests, current forest management practices in Southern Québec, and the importance of using arthropods in ecological studies. Chapter 2 is a field-based biodiversity study which tests the effect of vertical stratification and forest management on beetle and spider assemblages; essentially, how the communities change with height and with forest management treatment.

The specific research questions for the second chapter are the following:

- How are beetle and spider diversity and community structure changing along a vertical gradient in the forest?
- How is beetle and spider community structure changing with forest management treatment?
- What is the taxonomic diversity of beetles and spiders in the forested areas at the Kenauk Institute?

With this information, I hope to

- Investigate the community variation between forest management sites and height within a tree
- Better understand arthropod diversity and community structure changes caused by forest management
- Provide the Kenauk Institute with a detailed list of present arthropod species

1.7 Summary

Arthropods are an ideal model taxon and bio-indicator of ecosystem integrity, health, and change, since they are sensitive to abiotic changes, provide many ecosystem services, and can be easily sampled and preserved (Malecque et al. 2006; Malecque et al. 2009). Studies on distribution patterns of arthropods in temperate forests show highly stratified communities between the understorey and upper canopy (eg. Larrivée and Buddle 2009, Oguri et al. 2014, Ulyshen and Hanula 2007, Vance et al. 2007). This stratification is caused by multiple factors, most likely working simultaneously, such as forest structure (foliage complexity, microhabitat structure), climatic gradients (light and wind exposure, temperature), and various biotic factors (resource availability, inter-specific interactions, plant community composition) (Ulyshen 2011).

This stratification has implications for forest management practices since long-term logging practices have led to changes in the composition and physical structure of northeastern North American forests (Boucher et al. 2009; Graignic et al. 2016) and it seems that arthropods are sensitive to these differences (Maeto et al. 2002). Most notably, the creation of canopy gaps (seen when selectively cutting) and the removal of the upper canopy entirely (seen when strip cutting) create disturbances large enough to affect arthropod community structures (Koivula et al. 2002). Therefore, due to their sensitivity to anthropogenic change, arthropods are an ideal model taxon in estimating the ecological suitability of forest management techniques.

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1.9 Connecting Statement

This chapter provided the context and background information for the research presented in the following chapter, outlined the objectives of the thesis, and introduced the specific questions which will be in Chapter 2. Chapter 2 is about the effects of vertical stratification and forest management on beetle and spider communities, analyzed at the Family level.

Chapter 2: The Effect of Vertical Stratification and Forest Management on Beetle and Spider Communities in a Temperate Forest

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2.1 Abstract

It is predicted that arthropods will have highly stratified distributions when comparing the forest canopy to the forest floor. This has implications for forest management practices, since logging creates canopy gaps or removes the canopy entirely. This study quantified the diversity and community structure of beetles and spiders in response to vertical stratification (understorey, canopy) and forest management. Sampling was conducted near Montebello, Québec in uncut stands, stands that were logged at 2 different times (1989-1994, and 2001-2005), and stands logged using 2 different methods (selection and strip cutting). I collected 7933 beetles representing 495 species and 3196 spiders representing 91 species over two sampling seasons (summers of 2016 and 2017). Measures of diversity revealed a height effect on beetles but not spiders. Multivariate analysis (nonmetric multidimensional scaling ordination and permutational multivariate analysis of variance) showed that beetle and spider communities differed significantly between height but there were little (beetles) or no (spiders) community assemblage changes between forest management treatments. Height was found to be an important factor in shaping arthropod diversity and community structure and demonstrates that beetles and spiders assemble themselves vertically in a temperate forest in Québec. This research calls for the limitation of frequency (time-since harvest) and intensity (sylvicultural techniques) of forest management, since logging creates canopy gaps or removes the upper canopy entirely; if conservation of biodiversity is an important objective, the conservation of the canopy fauna must be a priority.

2.2 Introduction

Forest canopies are essential to forest health, since they support key ecosystem functions, such as photosynthesis, biogeochemical cycles and energy flow (Ozanne et al. 2003; Larrivée and Buddle 2009). Forest canopies may harbour up to 50% of the world's diversity (Ozanne et al. 2003), and a high diversity of arthropods has already been noted in northern, temperate forests (Larrivée and Buddle 2009; Maguire et al. 2014). The canopy and forest floor have different architectures derived from the abiotic and biotic components of each stratum, which could lead to the vertical stratification of arthropod biodiversity and community structures (Shaw 2004; Oguri et al. 2014).

The forest canopy is more variable than the understorey due to abiotic factors, such as radiation, moisture, temperature, and wind (Shaw 2004; Maguire et al. 2014), and to biotic factors, such as tree architecture, differences in foliage quality, and type and amount of foliage, fungi, dead wood, and carrion (Bassow and Bazzaz 1997; Ulyshen and Hanula 2007; Maguire et al. 2014). More specifically, foliage complexity of the canopy, such as foliage density and number of leaves and branchlets (Halaj et al. 2000; Corcuera et al. 2008; Oguri et al. 2014) and structural complexity of the understorey, such as litter depth and ground cover, have been found to shape spider assemblages (Docherty and Leather 1997; Bultman and DeWitt 2008; Oguri et al. 2014). The canopy and forest floor strata provide different microhabitats for arthropods, which may lead to different assemblages among the strata (Oguri et al. 2014).

Due to the increase in canopy sampling in temperate forests, it is becoming clear that arthropods are sensitive to these stratum differences (Basset et al. 2003b; Vance et al 2007). Most studies show dissimilarities in insect assemblages found in the forest understorey compared to the upper canopy (Vance et al. 2007) but not all report consistent patterns in richness and composition (see Leksono et al. 2005, Coleoptera in Maguire et al. 2014). While arthropods are generally equally or more diverse in the canopy in tropical rainforest, the opposite pattern is often seen in temperate forests (Ulyshen et al. 2010).

This vertical stratification of arthropods has implications for conservation of biodiversity and for forest management. Long-term logging practices have led to communal and physical changes to forests in northeastern North America, thus altering prime habitats for various forest flora and fauna (Boucher et al. 2009; Graignic et al. 2016). For example, forest management often creates younger, fragmented forests with a different composition of tree species than unmanaged forests (Boucher et al. 2009).

In Southern Québec, selection and strip cutting are the most commonly used silvicultural techniques, as they are considered more sustainable than other logging practices (Bédard and Majcen 2003). The effects of selection and strip cutting on plant growth, genetics and communities are well known (Angers et al. 2005; Bédard and Majcen 2003; Graignic et al. 2016; Malo and Messier 2011) but relatively little has been done on animal communities in Southern Québec (but see Morin et al. 2015 and Doyon et al. 2005).

Arthropods were chosen as the focal taxon since they are an important component of forest health and as such provide many ecosystem services, such as pollination, decomposition, nutrient cycling, and stabilizing food webs by buffering species' fluctuations and interactions (Teale and Castello 2011; Prather et al. 2013; Edmonds et al. 2000). More specifically, beetles and spiders are good model taxa for biodiversity studies since they are widespread and usually found in large numbers in forests (Leksono et al. 2005; Larrivée and Buddle 2009). Coleoptera is a highly diverse Order and includes species of all feeding guilds, fulfilling a wide range of roles in ecosystems (Dajoz 2007; Maguire et al. 2014). Spiders are one of the most prevalent groups of predatory arthropods in terms of both biomass and species diversity (Oguri et al. 2014), play a key role in ecosystems, acting as both predators to smaller arthropods and as prey to birds, and have distributions that directly relate to the structural forms of habitats (Larrivée and Buddle 2009).

The objectives of this study were to quantify diversity patterns and community structures of beetles and spiders at two heights (understorey, canopy), and in managed stands cut at two different times (1989-1994, 2001-2005) and using two different management techniques (selection and strip cutting). I hypothesized that beetle and spider diversity and community structure will differ between the strata, and between forest management stands, due to the physical, environmental, and ecological differences seen between the strata and stands.

2.3 Methods

2.3.1 Experimental design and sampling

This research was completed at Kenauk Nature (45°42'N'; 74°53'W'), a 260 km^2 mixed-deciduous forest near Montebello, Québec in 2016 and 2017. The property houses the Kenauk Institute and has a wide range of habitats, including swamps, bogs, and deciduous, coniferous, and mixed forests. All sites were located in mixed deciduous forests, composed primarily of sugar and red maple, *Acer saccharum* Marsh. and *Acer rubrum* L. (Aceraceae), American beech, *Fagus grandifolia* Ehrh. (Fagaceae), and Eastern hemlock, *Tsuga canadensis* L. Carrière (Pinaceae) in both the understorey and the canopy. The natural disturbance regime at Kenauk is characterized by natural, single-tree death disturbances rather than large-scale disturbances, such as fire.

Forest management has been occurring at Kenauk Nature since 1987. Managers employ multiple silvicultural techniques, of which selection cutting and strip cutting are the most commonly used; I conducted the study in stands managed using these two methods. The 2016 sites are located at least 1km away from each other and at least 100m away from the nearest road to avoid pseudoreplication (Table 2.1, Figure 2.1). The 2017 sites are located at least 250m away from each other and 100m away from the nearest road due to constraints with the logging locations (Table 2.1, Figure 2.2).

I was interested in two questions regarding forest management: What was the effect of **time-since harvest** on arthropod communities and what was the effect of **forest management type** on those same communities. To answer these questions, I separated the project into two years, one to address each question. For 2016, I tested for time-since harvest, whereas 2017 was dedicated to forest management type.

For both years, I selected 18 sites relating to forest management treatments (Table 2.1). For 2016, these 18 sites were composed of six old-growth (never cut) forest sites, six old cut sites (cut between 1989-1994), and six newly cut sites (cut between 2001-2005). The two forest management treatments (old cut and new cut) were cut using the selection cutting method. For 2017, these 18 sites were composed of six never cut forest sites, six selectively cut sites, and six strip cut sites. The two forest management treatments (selection cut and strip cut) were cut in 1994. In 2017, I sampled in six new trees in the never cut sites.

Each site comprised of one focal tree to sample in and around. I sampled in sugar maples because they are abundant, easy to identify and of economic importance (Farrar 1996). A conscious effort was made to ensure that all trees were of similar height and size by measuring their diameter at breast height (roughly 45 cm). The understorey was considered as the first 2 m above the forest floor and the upper canopy was considered as the top of the tree crown, about 20-25 m above the forest floor. Access to the upper canopy was made possible using a single-rope system, in which one person climbs a rope to the top of the tree.

Each site contained two different types of passive traps: unbaited Lindgren funnels and unbaited Barber pitfall traps. A Lindgren funnel consists of a series of funnels nested together and connected to a collection chamber at the bottom (Lindgren 1983). This trap is efficient at collecting flying arthropods, especially beetles, since arthropods fly into the funnels and are subsequently channelled down into the collection chamber (Lindgren 1983). Barber pitfall traps consist of a collection container, sunk in the ground so that the lip of the container is flush with the soil, and a roof of some sort (the original design for a Barber pitfall trap used a stone as a roof, for example), to prevent rain water from entering the trap (Barber 1931). Ground-dwelling arthropods that happen to travel over the trap fall into the collection container. Pitfall traps come in a wide variety of sizes; ours consisted of a plastic container with a 6cm diameter and a15cmx15cm plastic roof.

Each site consisted of two unbaited Lindgren funnels (Lindgren 1983) and six Barber pitfall traps in 2016 and five Barber pitfall traps in 2017 (Barber 1931). One Lindgren funnel was placed in the upper canopy and the other in the understorey (sugar maple saplings smaller than 2 m in height or branches off the main tree if they were lower than 2 m off the ground), while all pitfall traps were placed in the understorey. All six pitfall traps were analysed for the 2016 sampling season but only four of the five pitfall traps were analysed for the 2017 sampling season, due to time constraints. The discarded pitfall sample was randomly chosen each week using a random number generator. Pitfall traps were placed in a circle 5 metres away from the focal tree and roughly 5 metres away from each other.

Lindgren funnels and Barber pitfall traps were filled with a glycol solution (50:50 water and propylene glycol with a drop of dish soap) to kill and preserve the arthropods until collection. The traps were set up with the glycol solution and left to collect arthropods for the entirety of the 2016 and 2017 summers (June 1-September 20, 2016 and May 16-September 12, 2017), at one week intervals. Twelve sampling periods were selected for each year from the summer collection dates and analysed (Appendix A).

After one continuous week of collection, the two Lindgren funnels and pitfall traps were emptied for that sampling period. Upon emptying the traps, I rinsed samples with water and preserved them with 70% ethanol. At the same time, two foliage shaking beats were performed, one in the forest understorey (sugar maple saplings smaller than 2m in height) and one in the upper canopy. This consisted of beating a sugar maple branch five times and collecting all arthropods that fall from the branch onto a 1mx1m sheet fitted with a funnel and collection container (Wilson 1962). Specimens caught during foliage shaking were killed and preserved in 70% ethanol in the field. The preserved samples were then taken back to the laboratory for identification.

Overall, I used three trap types (Lindgren funnel, Barber pitfall trap, foliage shaking beats), ensuring that I would catch as much of the entire beetle and spider assemblages as possible. In total, each site for the 2016 sampling period had ten samples per week (two Lindgren funnels, six Barber pitfall traps and two foliage shaking beats) and each site for the 2017 sampling season had 8 samples per week (two Lindgren funnels, five Barber pitfall traps-of which four were analyzed, and two foliage shaking beats).

Identification keys and guides were used to identify adult specimens, including the *Guide d'indentification des Araignées (Araneae) du Québec* (Paquin and Dupérré 2003) and *American Beetles* (Arnett and Thomas 2001). Juvenile spiders were only identified to family. Vouchers were made for adult beetle and spider specimens and have been deposited at The Lyman Entomological Museum, McGill University (Ste-Anne-de-Bellevue, Québec) and the Canadian National Collection of Insects, Arachnids, and Nematodes (Ottawa, Ontario).

2.3.2 Data analyses

The data analyses were structured by the research objective. To test the effects of stratum by forest management on biodiversity, abundance and species richness metrics were used as response variables. The mean abundance and mean species richness were analyzed using a multifactorial analysis of variance (ANOVA). Since I was not interested in the change over time nor the effect of trap type, all dates, trap types and replicates were pooled for this analysis. Site number was treated as a random effect, while forest management treatment and height were considered fixed variables. Assumptions of normality and homogeneity of variances were satisfied by the data, but spider abundance and species richness for both years and beetle species richness for 2016 were log transformed before analyses. This analysis was performed for overall beetle and spider abundance and richness, and for individual families. Since pitfall traps were not placed in the forest canopy, these samples were omitted from analyses that directly compared understorey diversity to upper canopy diversity. After the ANOVA, I performed a Tukey honestly significant difference post hoc test (Tukey HSD test) to determine significant differences between means. These analyses were performed using R version 3.3.3 (R Core Team 2017).

I examined species diversity by first creating rarefaction curves to determine if adequate sampling had been achieved (i.e., a plateau had been reached) (Buddle et al. 2005). These were created using the rarefy function in the vegan package (Oksanen et al. 2016) in R 3.3.3 (R Core Team 2017). The rarefaction curves of our sampled communities approached an asymptote (Fig. 2.3), so species richness was used as a measure of diversity, along with Shannon's and Simpson's diversity indices, Pielou's evenness, and Fisher alpha. To infer statistical significance, I performed ANOVAs and Tukey HSD tests. Again, all dates, trap types and replicates were pooled for this analysis.

The differences in community compositions between stratum and forest management treatments were analysed by using a permutational multivariate analysis of variance (PERMANOVA) (Anderson 2001) on family relative abundance. I selected PERMANOVA because it tests the response of multiple variables (family abundances) to multiple factors (stratum and forest management sites) in an ANOVA design by using permutations on distance measures (Anderson 2001). I used the Bray-Curtis dissimilarity index for this analysis. Family-level abundances were used because species-level abundances gave more response variables (in this case, species identities) than degrees of freedom; multivariate tests require the opposite.

To test the effects of stratum and forest management treatments on community assemblages, I used measures of relative abundance and diversity. Family community matrices were plotted in ordination space using the metaMDS function in the vegan package (Oksanen et al. 2016) of R 3.3.3 (R Core Team 2017). This gives a visual representation of community similarity where each point in the ordination space represents an assemblage at a given forest management replicate and height. Stratum and forest management centroids surrounded by 95% confidence intervals (ordiellipse function in vegan) were included on the ordination to obtain statistically testable values and delimit the site boundaries. Each ordination was calculated using two dimensions, since this resulted in an ecologically meaningful stress value (stress <0.20) (Clarke 1993). I chose nonmetric multidimensional scaling ordination (NMDS) as my ordination technique because it makes few assumptions about the composition of ecological species data compared to other ordination techniques (Digby and Kempton 1987). Bray-Curtis distance was selected for this analysis, as it is appropriate for community data (Legendre and Gallagher 2001). Family-level data were used for the ordinations to match the PERMANOVA.

2.4 Results

2.4.1 Abundance and Diversity

Overall, 4623 beetles representing 347 species and 2268 spiders representing 69 species were collected in 2016 (Tables 2.2 and 2.3). I collected 3310 beetles representing 328 species and 928 spiders representing 62 species in 2017 (Tables 2.2 and 2.3). Several families of beetles (e.g., Staphylinidae, Elateridae, Curculionidae) contained species that were difficult to identify to the species level and were therefore treated as morphospecies in the analyses. All adult spiders were identified to species whereas the juveniles were identified to family. In 2016, a total of 812 spiders were juveniles and represented 35.80% of the total abundance of spiders whereas in 2017, juveniles represented 49.13% of the total catch, with a total of 456 individuals. Tables 2.2 and 2.3 represent a subset of the total abundance data, with the most abundant families represented.

Of the beetles, Latridiidae (minute fungus beetles) was the most commonly collected family (37% and 27% of the total catch in 2016 and 2017, respectively) and Elateridae (click beetles) was the most species-rich family (54 and 53 morpho-species in 2016 and 2017, respectively) (Tables 2.2 and 2.3). Family Latridiidae was significantly more abundant in the never cut sites in both 2016 and 2017 (p=0.0002 and p= 0.0001 in 2016 and 2017, respectively) (Tables 2.2 and 2.3). Of the spiders, Lycosidae (wolf spiders) was the most commonly collected family (32% and 19% of the total catch in 2016 and 2017, respectively) and Linyphiidae (micro sheet web spiders) was the most diverse family (27 and 16 species in 2016 and 2017, respectively) (Tables 2.2 and 2.3). A complete list of species and abundance values can be found in Appendix B.

There were no differences in total abundance or species richness of beetles and spiders in either year between forest management treatments (Tables 2.4 and 2.5). Similarly, there was no difference in spider abundance between the two strata for either year (Tables 2.4 and 2.5). However, beetle abundance was found to be significantly higher in the understorey for both 2016 and 2017 (p=2.869e-06 and p=0.0036 for 2016 and 2017, respectively) whereas richness was found to significantly differ between strata in 2016 only (p=2.279e-05) (Tables 2.4 and 2.5). In addition, species richness of spiders was significantly higher in the understorey in 2016, but was insignificant in 2017 (p=0.010 in 2016 and p=0.106 in 2017) (Table 2.4 and 2.5).

Based on rarefaction curves, complete sampling was achieved for both taxa, in both strata and at all forest management treatments (Fig. 2.3), so species richness was used as a measure of diversity, along with Shannon's and Simpson's diversity indices, Pielou's evenness, and Fisher alpha. However, results differed depending on which metric was used: species richness saw differences between the strata in 2016, whereas Shannon's and Simpson's diversity indices did not indicate effects of height on beetle diversity in either year (Table 2.6). Interestingly, both indices found significantly different beetle diversities between forest management treatments, with an interaction effect between stratum and site in 2016 (Table 2.6). Here, never cut sites were found to have a lower beetle diversity than the managed sites (Table 2.7). Pielou's evenness of beetles was found to be significantly different between the forest management treatments in both years, with the never cut sites being more uneven than the managed sites (Tables 2.6 and 2.7). Pielou's evenness also showed a difference between height in 2017, with the understorey showing more unevenness than the upper canopy (Table 2.6).

The tested diversity measures for spiders agreed with species richness and abundance, as no index found any site differences in diversity (Table 2.8). However, Shannon's diversity measure found significant differences between the two strata in both years (Table 2.8). In addition, Pielou's evenness found a significantly more uneven community on the forest floor compared to the upper canopy for both years (Table 2.8).

2.4.2 Community Composition

The NMDS ordination showed a clear separation between the beetle and spider samples collected in the understorey and upper canopy (Figs. 2.4 and 2.5). For both taxa and both years, height treatment clustered together, regardless of forest management treatment. Understorey assemblages of beetles and spiders were similar across the three forest management treatments in both years and therefore clustered together, with overlapping ellipses (Figs. 2.4 and 2.5). The same trend can be seen with the upper canopy communities; with the canopy samples clustering together and overlapping their ellipses regardless of forest management treatment (Figs. 2.4 and 2.5). The smaller ellipses seen around the understorey clusters represent less variability between the samples than the larger ellipses seen around the upper canopy samples (Figs. 2.4 and 2.5).

PERMANOVA indicated a strongly significant stratum and forest management treatment effect on beetle assemblages for both 2016 and 2017 (Table 2.9). The significant differences seen between forest management treatments were driven by family Latridiidae in both years; when removing this family from the analysis, the significance is lost (p=0.33 and p=0.11 in 2016 and 2017, respectively). Spider assemblages were affected by height in both years (Table 2.10). There were no interaction effects for both beetles and spiders between height and forest management treatment (Tables 2.9 and 2.10).

2.5 Discussion

My main objective was to quantify patterns of diversity and distribution and to compare community structure of beetle and spider assemblages in two strata and in three forest management treatments. My overall results showed that beetles and spiders respond to height differences within a forest system, in terms of diversity and community structure. However, there were very little (beetles) or no (spiders) diversity and community assemblage changes between forest management treatments. My results point to the importance of forest canopies in structuring arthropod biodiversity in a temperate forest, and illustrate that at the current level of forest management in my study forest, there are few negative effects on beetles and spiders.

2.5.1 Vertical stratification

Total abundance and species richness of beetles were higher in the understorey than the upper canopy in 2016, but only abundance was significantly different in 2017. These results are similar to those of Aikens and Buddle (2012), who found that the abundance of beetles and spiders decreased significantly with height, along with an overall lower species richness in the upper canopy. In comparison, Schroeder et al. (2009) found the opposite trend in beetles, with higher abundance and higher species richness found in the upper canopy. Maguire et al. (2014) found no difference in beetle abundance or species richness between the strata. Despite contrasting results, these articles, along with many others, illustrate an important pattern in Coleoptera: diversity is not limited to the understorey in temperate forests.

Total abundance of spiders did not differ between heights but species richness was higher in the understorey in 2016 only (2017 was marginally insignificant). Larrivée and Buddle (2009) and Aikens and Buddle (2012) found higher species richness of spiders in the understorey than in the upper canopy, results that support my own. The forest understorey contains a wider variety of microhabitats and hosts a larger complexity than the upper canopy, due to the differences in physical architectures between the strata (Oguri et al. 2014). These physical differences, in combination with habitat stability in the understorey, and environmental variability in the upper canopy, can lead to a higher diversity of arthropods in the understorey (Larrivée and Buddle 2009).

Beetle assemblages showed marked community differences between height in both years, as demonstrated by the PERMANOVA and the ordination, with the understorey being dominated by ground and rove beetles (Carabidae, Staphylinidae) and the canopy harbouring more bark beetles (Cerambycidae). These results are supported by Maguire et al. (2014), Ulyshen and Hanula (2007), and Schroeder et al.

(2009), who found community differences in beetles between the two strata. These communal dissimilarities can be explained by the differences in physical and ecological characteristics seen between the strata, such as forest structural patterns and biological interactions between species (Schroeder et al. 2009).

Similarly, spiders showed significant community differences between height in both years, with wandering spiders (Lycosidae, Amaurobidae, Hahniidae) dominating the forest understorey and jumping spiders (Salticidae) dominating the upper canopy (when removing pitfall traps from the analyses). These results agree with Aikens and Buddle (2012) and Larrivée and Buddle (2009), who found that jumping spiders dominated the upper canopy. Similarly, Oguri et al. (2014) found different spider assemblages between the strata in coniferous forests in Japan.

These family-level differences between the two strata are likely caused by species-specific responses (Maguire et al. 2014) to behavioural (Brown et al. 1997), ecological (Larrivée and Buddle 2009), and abiotic (Basset et al. 2003a) factors. Several species were more abundant in one stratum. For example, *Glischrochilus sanguinolentus* Olivier (Coleoptera: Nitidulidae), while common, was most often found in the understorey, a result that has been seen in past work in this area (Maguire et al. 2014). This species feeds on sap exuding from wounds in older bark, meaning that they are most likely found in the understorey (Craighead 1950). Similarly, *Phyllobius oblongus* Linnaeus (Coleoptera: Curculionidae), an introduced species, was also more abundant in the understorey. This species emerges at the beginning of summer to eat freshly sprouted maple leaves and then moves onto plants with indeterminate leaf growth, such as raspberries (Coyle et al. 2010). Interestingly, *Stenocorus vittiger* Randall (Coleoptera: Cerambycidae) was almost exclusively found in the upper canopy, despite being commonly found on flowering shrubs (Gosling and Gosling 1977), suggesting that this species may be a migrant traveller throughout the forest (Aikens and Buddle 2012).

When removing pitfall traps, jumping spiders (Salticidae) and some comb-footed spiders (Theridiidae) dominated the upper canopy, and this trend can be explained ecologically. The upper canopy needs to be recolonized every year, since canopy species will descend to the forest floor to overwinter (but there are a few that overwinter in canopy bark crevices) (Aikens and Buddle 2012; Larrivée and Buddle 2009). These canopy inhabitants tend to have good dispersal abilities, such as aerial ballooning. While we only found a handful individuals, *Theridion murarium* Emerton (Theridiidae) was only found in the upper canopy, and is a known aerial disperser (Bell et al. 2005). Similarly, jumping spiders are visual hunters and do not build webs to catch their prey. Webs are costly to make and it is difficult to maintain webs in the upper canopy, as the harsher environmental conditions may destroy them, thus selecting for jumping and pursuing spiders (Aikens and Buddle 2012; Larrivée and Buddle 2009).

Based on these results, biodiversity assessments and future ecological work, regardless of location, should consider adding canopy sampling to their protocol. Admittedly, in temperate areas, abundance and species richness seems to be higher in the understorey (Ulyshen et al. 2010). However, if the goal is detailing taxonomic diversity, and to a lesser extent conservation, then canopy sampling may provide additional information about stratum specialist species that would otherwise have been omitted.

Biodiversity monitoring depends on collecting specimens that are representative of regional species pools (Maguire et al. 2014). I have shown that the canopy is a reservoir of diversity, and as such omitting it may lead to an inaccurate understanding and representation of forest systems, which in turn can result in misinformed conservation policies. Su and Woods (2001) pointed out that the vertical range of canopy insects, including beetles, constitutes the basic information essential for conservation plans related to forest management. In addition, species that are restricted to a specific stratum generally have higher conservation priorities than habitat generalists (Leksono et al. 2005). Put together, I call for the inclusion of canopy sampling in arthropod studies, especially when the main goal of the research is taxonomic diversity and conservation.

2.5.2 Forest management

Overall beetle abundance and species richness exhibited no differences between forest management treatments in either year. When analyzing individual families, only family Latridiidae showed significant differences in abundance between forest management treatments, with a higher abundance in the never cut sites in both years. These results agree with Martikainen et al. (2000), who found that forest management did not affect non-saproxylic beetle diversity. However, they did see a difference in saproxylic beetle diversity: they found a higher abundance and diversity in unmanaged stands, something they attribute to the presence of dead and decaying wood. Several other studies have also investigated beetle diversity in relation to forest management, with Maeto et al. (2002) and Zeran et al. (2006) finding diversity differences in Cerambycidae and Nitidulidae, respectively. Only Moore et al. (2004) and Zeran et al. (2007) found no site effect on carabid and fungivorous beetle diversity, respectively. While I did not find any diversity differences for Coleoptera overall, I did see a site effect for family Latridiidae, suggesting that future work in this area should investigate a smaller scope, as it seems that response to anthropogenic change may be taxon-specific.

Total abundance and diversity of spiders did not change between forest management treatments. Buddle et al. (2000) found that while spider communities differed significantly when forests were recently burned or harvested, differences among older stands, despite having different disturbance histories, were minimal. As such, spider communities seemed to differ most significantly as a function of general forest cover, but less so by disturbance history or age of the forest. In contrast, Jennings et al. (1988), found that spider abundance was higher in strip cuts and attribute this result to the new habitats that strip cutting creates, thus leading to a higher abundance of colonizing spiders. Both studies noted that family Lycosidae (wolf spiders) saw a significant increase in abundance in the more recently cut stands and that this increase in abundance is what is most likely driving the significance seen in overall spider abundance in Jennings et al. (1988).

Abundance can increase directly after a cut, since this creates a new habitat for beetles and spiders to exploit. Since spiders are good dispersers and generalist predators (Buddle et al. 2006), they can recolonize a disturbed site rather quickly. Once the canopy cover begins to close, species that cannot survive either die or emigrate. Since I did not see any differences in abundance between managed and unmanaged stands, I concluded that my stands represented an age class that had already started to see canopy closure, and thus the shift from open-habitat, open-canopy stands to closed-forest, closed-canopy stands had already been established. Spiders are especially sensitive to the sharp temperate gradient created by the lack of canopy cover (McIver et al. 1992), and this seems to dramatically affect wolf spiders (Jennings et al. 1988). I believe that if I had chosen stands cut more recently (1-5 years prior the study), I may have seen this increase in abundance and had been able to compare and contrast this spike in abundance to older, managed stands.

While abundance and species richness saw no differences between forest management treatments for beetles, other diversity metrics showed otherwise: Shannon's and Simpson's diversity indices, and Pielou's evenness indicated a lower diversity and higher evenness in the never cut sites, respectively. This could be attributed to the higher abundance of Latridiidae in the never cut sites; family Latridiidae dominated this forest management treatment, thus lowering the overall diversity of the stands. This is confirmed by Pielou's evenness, as never cut sites had a lower evenness than the managed stands.

Forest management, or any other large-scale disturbance, creates new and relatively uniform landscapes compared to pre-disturbance forests. This is especially seen in clear and strip cuts, where large expanses of land are cut and all woody material is removed. Here, all colonizing arthropods have an equal chance at survival (Buddle et al. 2000), which may explain the higher diversity (Shannon's and Simpson's diversity indices) and evenness we saw in our beetle data. As the stands return to their pre-cut plant communities, and most notably, as the canopy cover returns (Koivula et al. 2002), species that cannot survive will either die or emigrate, leading to a more uneven community (Buddle et al. 2000). This may be the case here at the Kenauk Institute property, as the never cut sites were highly uneven compared to the highly even managed stands.

PERMANOVA and ordination showed that there were significantly different beetle communities between forest management treatments in both years. This significance was driven by family Latridiidae, as they were the most abundant family and most commonly found in the never cut sites. Latridiidae (minute fungus beetles) is a fungivorous family that feeds on the fungi that grow on dead or decaying wood, material that is removed in managed stands. Interestingly, Zeran et al. (2007) found that forest management did not affect fungivorous beetle composition, results that contrast our own Latridiidae results. Several other studies have found contrasting results: Maeto et al. (2002) and Zeran et al. (2006) both found compositional differences in Cerambycidae and Nitidulidae whereas Martikainen et al. (2000) found no difference for non-saproxylic beetles.

Spider communities did not differ between forest management treatments, as shown by PERMANOVA and the ordination. Most spiders are generalist predators (Buddle et al. 2006) and as such can easily repopulate a disturbed area. Buddle et al. (2000) argued that community composition may not differ between managed and unmanaged sites since source populations from unmanaged stands are relatively close to strip and selection cuts, and as such spiders re-colonize rather quickly after cutting. This may be the case at the Kenauk Institute property, as the managed and unmanaged stands are intertwined throughout the property.

Previous work has shown that spider communities diverge away from their original state for the first 7 years after a cut (Huhta 1971) but then start to return to pre-cut assemblages once the canopy begins closing, after about 15 years of growth

(Buddle et al. 2000). My managed stands were cut between 11 and 28 years before my study was conducted, and as such the canopy gaps were not substantial; in fact, the stands had relatively few canopy openings. The stands used for my time-since harvest project were all cut using selection cutting, a method that creates small canopy gaps as opposed to complete canopy removal seen in strip cuts, and the stands used for my forest management type project were all cut 28 years before the study was conducted, old enough that the canopy cover was able to return. Considering I did not see any community differences between managed and unmanaged stands, I believe that selection cutting did not compromise canopy cover enough to alter microenvironmental conditions at the forest floor level (McIver et al. 1992), and that strip cut stands can return to pre-cut amount of canopy cover and, in turn, arthropod assemblages in as little as 28 years.

Previous work has also shown that low-impact forest management does not affect biodiversity as much as higher impact techniques, such as clear cutting, do. In Québec, Brais et al. (2004) found that vegetation dynamics differed greatly between clear cut stands and uncut stands, but that difference was smaller between the partially cut stands and uncut stands. In Alberta, Harrison et al. (2005) found that songbirds declined or disappeared entirely from stands with low retention levels (10-20% of pre-cut amount of trees) and suggest that retention levels of 50-75% may be adequate in conserving songbird communities. Lazaruk et al. (2005), also in Alberta, found that ectomycorrhizae diversity and community structure changed drastically with increasing amount of tree removal. These articles, along with many others, including my own, illustrate that if tree removal is light enough, as is the case at the Kenauk Institute site, biodiversity effects are minimal.

These results have shown that selection cutting may be an adequate silviculture technique in ensuring that beetle and spider diversity and community structure do not change drastically. They have also shown that it takes as little as ca 30 years for a strip cut stand to return to its original amount of canopy cover. Because of this, I suggest that, while overall forest management did not have any diversity or community effects on beetles and spiders, forest management be limited in its frequency (time-since harvest) and intensity (sylvicultural techniques), since forest management practices remove forest canopies and create large canopy gaps. I showed that beetle and spider communities between the strata returned to pre-cut heterogeneity in as little as 11 years in selectively cut stands, suggesting that increasing the interval time between cuts and using methods that keep the canopy relatively intact are adequate means in ensuring a robust arthropod community in managed stands.

However, it should be noted that forest management at the Kenauk Institute property is relatively light compared to other areas, as the forest managers tend to selectively cut stands and have not returned to a cut area since they starting logging in 1987. Therefore, these results may not translate well for areas in which logging is more rigorous. Furthermore, it is important to note that relatively minimal effects on two groups of arthropods cannot be generalized to all components of biodiversity.

2.5.3 Conclusion

Forest management and vertical stratification are intimately connected with one another. Forest management alters arthropod diversity and community structure at the understorey level due to canopy removal, and the removal of this canopy cover also eliminates the unique arthropod communities found in this stratum, homogenizing communities between strata (Schroeder et al. 2009). Even though canopy cover is most likely the main driver in structuring arthropod communities in managed stands, relatively few forest management studies actually sample in the canopy (but see Schowalter 1989, 1995). Based on these results, I suggest that forest management studies should include canopy sampling in their protocols, especially in light of this connection between arthropod communities and canopy cover.

Biodiversity monitoring depends on collecting specimens that are representative of regional species pools (Maguire et al. 2014). Therefore, it is important to sample in the upper canopy as well as the understorey, since the upper canopy is unique in its diversity and community composition. I found that the target taxa and height are important to arthropod sampling in temperate forests. I have shown that changes in community structure are taxon specific. I have also shown that height plays a role in structuring these assemblages, since both beetle and spider communities changed with increasing height within a tree. Finally, I found that overall community structure, with height pooled, does not seem to change with forest management treatment. I emphasized that canopy cover is a main driver in community assemblage differences in managed stands, and suggest that the lack in community change seen here is most likely due to the high level of canopy closure seen in my sites.

This study stresses the importance of considering taxon specific responses to vertical stratification and forest management, and the connection between canopies and forest management. Further work on this topic will help expand knowledge on the role canopies play in forest ecosystems, especially within the context of forest management.

This work has provided invaluable information for the Kenauk Institute, as it is the first of its kind to document beetle and spider diversity and community structure on the property. Baseline data is essential for biodiversity monitoring, a goal that the Kenauk Institute wishes to attain through yearly sampling.

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2.8 Tables and Figures

Table 2.1. GPS coordinates for all sampled locations at Kenauk Nature, Montebello, Québec. The time-since harvest project was conducted in 2016, where three different age classes are tested (type of cut is constant) and the forest management type project was conducted in 2017, where three different techniques are tested (year of cut is constant).

2016		2017	
2001-2005, Selective cut, rep 1	45.788, -74.787	Strip cut, 1994, rep 1	45.808, -74.828
2001-2005, Selective cut, rep 2	45.727, -74.796	Strip cut, 1994, rep 2	45.816, -74.822
2001-2005, Selective cut, rep 3	45.779, -74.798	Strip cut, 1994, rep 3	45.821, -74.819
2001-2005, Selective cut, rep 4	45.800, -74.848	Strip cut, 1994, rep 4	45.823, -74.821
2001-2005, Selective cut, rep 5	45.808, -74.845	Strip cut, 1994, rep 5	45.831, -74.818
2001-2005, Selective cut, rep 6	45.829, -74.846	Strip cut, 1994, rep 6	45.844, -74.823
1989-1994, Selective cut, rep 1	45.708, -74.823	Selection cut, 1994, rep 1	45.806, -74.828
1989-1994, Selective cut, rep 2	45.707, -74.813	Selection cut, 1994, rep 2	45.827, -74.818
1989-1994, Selective cut, rep 3	45.728, -74.820	Selection cut, 1994, rep 3	45.828, -74.821
1989-1994, Selective cut, rep 4	45.806, -74.830	Selection cut, 1994, rep 4	45.837, -74.818
1989-1994, Selective cut, rep 5	45.824, -74.819	Selection cut, 1994, rep 5	45.842, -74.821
1989-1994, Selective cut, rep 6	45.832, -74.819	Selection cut, 1994, rep 6	45.845, -74.825
Never cut, Selective cut, rep 1	45.727, -74.828	Never cut, 1994, rep 1	45.780, -74.801
Never cut, Selective cut, rep 2	45.718, -74.843	Never cut, 1994, rep 2	45.773, -74.807
Never cut, Selective cut, rep 3	45.797, -74.791	Never cut, 1994, rep 3	45.743, -74.833
Never cut, Selective cut, rep 4	45.742, -74.833	Never cut, 1994, rep 4	45.727, -74.840
Never cut, Selective cut, rep 5	45.771, -74.810	Never cut, 1994, rep 5	45.719, -74.844
Never cut, Selective cut, rep 6	45.781, -74.802	Never cut, 1994, rep 6	45.727, -74.827

Table 2.2. Total abundance (mean per treatment±SE) of the most abundant Coleoptera and Araneae families across site treatments in summer of 2016. Significantly different values (based on ANOVA) are presented in bold and with different letters.

	Coleoptera Number of individuals (mean ± SE)			
Family	2001-2005	1989-1994	Never cut	No. spp
Latridiidae	**455 (37.92±6.20a)**	**321 (26.75±4.13a)**	**936 (78.00±12.66b)**	5
Carabidae	217 (18.08±6.73a)	376 (31.33±12.30a)	118 (9.83±2.99a)	31
Curculionidae	131 (10.92±2.97a)	137 (11.42±3.33a)	217 (18.08±6.41a)	12
Staphylinidae	136 (11.33±4.48a)	68 (5.67±1.81a)	64 (5.33±2.22a)	43
Elateridae	85 (7.08±1.34a)	98 (8.17±1.53a)	66 (5.50±1.07a)	54
Cerambycidae	99 (8.25±3.73a)	74 (6.17±0.94a)	73 (6.08±1.35a)	26
Nitidulidae	33 (2.75±0.97a)	74 (6.17±2.28a)	133 (11.08±3.54a)	16
Total (all families)	1384	1364	1875	347
	Araneae Number of individuals (mean ± SE)			
Family	2001-2005	1989-1994	Never cut	No. spp
Lycosidae	217 (18.08±8.65a)	172 (14.33±7.62a)	345 (28.75± 9.36a)	3
Linyphiidae	113 (9.42±3.40a)	70 (5.83±1.77a)	117 (9.75±3.27a)	27
Hahniidae	94 (7.83±3.60a)	101 (8.42±3.33a)	101 (8.42±3.03a)	1
Total (all families)	717	685	866	69

Table 2.3. Total abundance (mean per treatment±SE) of the most abundant Coleoptera and Araneae families across site treatments in summer of 2017. Significantly different values (based on ANOVA) are presented in bold and with different letters.

	Coleoptera Number of individuals (mean ± SE)			
Family	Strip cut	Selective cut	Never cut	No. spp
Latridiidae	**166 (13.83±3.33a)**	**223 (18.58±4.16a)**	**493 (41.08±6.71b)**	5
Carabidae	242 (20.17±6.30a)	210 (17.50±5.74a)	96 (8.00±2.60a)	27
Elateridae	150 (12.50±3.78a)	144 (12.00±1.70a)	100 (8.33±1.78a)	53
Curculionidae	147 (12.25±3.52a)	88 (7.33±2.60a)	119 (9.92±3.73a)	17
Nitidulidae	89 (7.42±2.12a)	94 (7.83±2.31a)	31 (2.58±0.58a)	18
Scarabeideae	72 (6.00±1.77a)	40 (3.33±0.85a)	31 (2.58±0.71a)	16
Erotylidae	36 (3.00±1.21a)	33 (2.75± 0.75a)	53 (4.42±2.42a)	6
Staphylinidae	46 (3.83±0.97a)	35 (2.92±0.84a)	39 (3.25±1.17a)	33
Cerambycidae	32 (2.67±1.34a)	30 (2.50±0.88a)	56 (4.67±2.29a)	17
Cantharidae	17 (1.42±0.47a)	35 (2.92±0.61a)	24 (2.00±0.49a)	18
Tenebrionidae	22 (1.83±0.46a)	17 (1.42±0.34a)	13 (1.08±0.31a)	16
Total (all families)	1098	1031	1181	328
	Araneae Number of individuals (mean ± SE)			
Family	Strip cut	Selective cut	Never cut	No. spp
Lycosidae	217 (18.08±8.65a)	345 (28.75± 9.36a)	172 (14.33±7.62a)	3
Linyphiidae	113 (9.42±3.40a)	117 (9.75±3.27a)	70 (5.83±1.77a)	27
Hahniidae	94 (7.83±3.60a)	101 (8.42±3.03a)	101 (8.42±3.33a)	1
Total (all families)	307	280	341	62

Table 2.4. Total abundance and species richness (mean per treatment±SE) of Coleoptera and Araneae across treatments in summer of 2016. Significantly different values (based on ANOVA) are presented in bold and with different letters.

Treatment	Coleoptera (mean±SE)		Araneae (mean±SE)	
	Abundance	Richness	Abundance	Richness
Height				
Understorey	**2102 (116.77±13.09a)**	**195 (30.83±1.84a)**	179 (9.94±1.10a)	**20 (2.17±0.42a)**
Canopy	**1165 (64.72±7.43b)**	**163 (20.44±1.26b)**	143 (7.94±1.09a)	**11 (1.05±0.27b)**
Site				
2001-2005	1384 (114.58±19.09a)	200 (34.66 ±4.14a)	717 (59.75±18.45a)	45 (8.66±2.65a)
1989-1994	1364 (112.83±18.87a)	191 (34.50±3.94a)	685 (57.08±15.10a)	50 (8.33±2.14a)
Never cut	1875 (155.83±26.80a)	196 (34.66±5.73a)	866 (72.16±20.22a)	33 (10.00±2.74a)

Table 2.5. Total abundance and species richness (mean per treatment±SE) of Coleoptera and Araneae across treatments in summer of 2017. Significantly different values (based on ANOVA) are presented in bold and with different letters.

Treatment	Coleoptera (mean±SE)		Araneae (mean±SE)	
	Abundance	Richness	Abundance	Richness
Height				
Understorey	**1458 (81.00±6.78a)**	190 (25.11±1.30a)	164 (9.11±0.87a)	18 (2.33±0.32a)
Canopy	**1017 (56.50±6.66b)**	188 (24.89±1.90a)	174 (9.66±1.14a)	17 (2.94±0.28a)
Site				
Strip cut	1102 (90.33±17.88a)	173 (30.83±3.67a)	307 (25.585±5.75a)	37 (7.33±1.64a)
Selection cut	1033 (85.33±10.67a)	179 (32.75±2.12a)	280 (23.33±4.87a)	47 (6.58±1.41a)
Never cut	1183 (98.00±12.62a)	198 (30.33±2.39a)	341 (28.41±6.29a)	28 (5.16±1.24a)

Table 2.6. ANOVA p-values for the effect of height and forest management site on the total abundance, richness, and diversity of beetles at the Kenauk Institute in 2016 and 2017. Significant values are presented in bold. Additional Tukey HSD tests were conducted for "Site" and those p-values can be found in Table 2.9.

	Response	Factor	Chisq	Df	Pr(>Chisq)
Beetles 2016	Simpson	Height	0.675	1	0.411
		Site	19.356	2	**6.266e-05**
		Height*Site	9.188	2	**0.010**
	Shannon	Height	2.116	1	0.145
		Site	11.349	2	**0.0034**
		Height*Site	6.261	2	**0.043**
	Fisher alpha	Height	3.379	1	0.066
		Site	3.710	2	0.156
		Height*Site	2.319	2	0.313
	Pielou's evenness	Height	0.521	1	0.470
		Site	23.811	2	**6.753e-06**
		Height*Site	3.632	2	0.162
	Species richness	Height	17.940	1	**2.279e-05**
		Site	9e-04	2	0.999
		Height*Site	5.675	2	0.058
	Abundance	Height	21.902	1	**2.869e-06**
		Site	2.692	2	0.260
		Height*Site	9.2172	2	**0.0099**
	Response	**Factor**	**Chisq**	**Df**	**Pr(>Chisq)**
Beetles 2017	Simpson	Height	2.845	1	0.091
		Site	25.465	2	**2.953e-06**
		Height*Site	1.826	2	0.401
	Shannon	Height	3.562	1	0.059
		Site	14.142	2	**0.00084**
		Height*Site	1.464	2	0.480

	Fisher alpha	Height	12.364	1	**0.00043**
		Site	5.940	2	0.051
		Height*Site	0.886	2	0.641
	Pielou’s evenness	Height	7.318	1	**0.0068**
		Site	23.567	2	**7.627e-06**
		Height*Site	1.372	2	0.503
	Species richness	Height	0.0007	1	0.978
		Site	0.402	2	0.817
		Height*Site	4.319	2	0.115
	Abundance	Height	8.473	1	**0.0036**
		Site	0.458	2	0.795
		Height*Site	3.056	2	0.216

Table 2.7. Tukey HSD test p-values for the factor “Site” for beetles. Significant values are presented in bold.

	2016			2017		
	1989-1994 2001-2005	Never cut 2001-2005	Never cut 1989-1994	Selection Strip	Never cut Strip	Never cut Selection
Shannon	0.83180	**0.01005**	**0.00764**	0.80605	**0.00353**	**0.00223**
Simpson	0.94815	**0.000366**	**0.000366**	0.61500	**1.27e-05**	**8.36e-05**
Pielou	0.85500	**7.21e-05**	**4.80e-05**	0.43820	**1.7e-05**	**0.000336**

Table 2.8. ANOVA p-values for the effect of height and forest management site on the total abundance, richness, and diversity of spiders at the Kenauk Institute in 2016 and 2017. Significant values are presented in bold; slightly insignificant values are presented in italics.

	Response	**Factor**	**Chisq**	**Df**	**Pr(>Chisq)**
Spiders 2016	Simpson	Height	0.197	1	0.657
		Site	2.269	2	0.321
		Height*Site	0.778	2	0.677
	Shannon	Height	6.791	1	**0.009**
		Site	0.212	2	0.899
		Height*Site	3.155	2	0.206
	Fisher alpha	Height	0.010	1	0.919
		Site	1.587	2	0.452
		Height*Site	0.313	2	0.854
	Pielou’s evenness	Height	12.154	1	**0.0004**
		Site	2.872	2	0.237
		Height*Site	1.117	2	0.571
	Species richness	Height	6.607	1	**0.010**
		Site	0.265	2	0.875
		Height*Site	2.511	2	0.284
	Abundance	Height	2.608	1	0.106
		Site	0.836	2	0.658
		Height*Site	4.056	2	0.131
	Response	**Factor**	**Chisq**	**Df**	**Pr(>Chisq)**
Spiders 2017	Simpson	Height	0.065	1	0.798
		Site	1.745	2	0.417
		Height*Site	0.472	2	0.789

	Shannon	Height	3.325	1	0.068
		Site	2.183	2	0.335
		Height*Site	0.487	2	0.783
	Fisher alpha	Height	1.885	1	0.169
		Site	1.049	2	0.591
		Height*Site	1.760	2	0.414
	Pielou’s even	Height	5.774	1	**0.016**
		Site	2.744	2	0.253
		Height*Site	4.640	2	0.098
	Species richness	Height	2.602	1	0.106
		Site	1.267	2	0.530
		Height*Site	0.817	2	0.664
	Abundance	Height	0.166	1	0.683
		Site	0.439	2	0.802
		Height*Site	1.521	2	0.467

Table 2.9. PERMANOVA on Bray-Curtis distances of Coleoptera assemblages in the summer of 2016 and 2017. Significant p-values (p<0.05) are presented in bold.

2016					
Source	df	SS	MS	F	p
Height	1	2.1390	2.13902	24.7250	**9.999e-05**
Site	2	0.6600	0.33000	3.8145	**0.0007999**
Height x Site	2	0.2909	0.14543	1.6811	0.0981902
2017					
Source	df	SS	MS	F	p
Height	1	1.5256	1.52560	12.103	**9.999e-05**
Site	2	0.7239	0.36197	2.3479	**0.0103**
Height x Site	2	0.3173	0.15867	1.4671	0.1248

Table 2.10. PERMANOVA on Bray-Curtis distances of Araneae assemblages in the summers of 2016 and 2017. Significant p-values ($p<0.05$) are presented in bold.

2016					
Source	df	SS	MS	F	p
Height	1	4.9802	4.9802	30.972	**9.999e-05**
Site	2	0.2628	0.13138	0.42569	0.9471
Height x Site	2	0.2677	0.1338	0.8133	0.566434
2017					
Source	df	SS	MS	F	p
Height	1	3.3829	3.3829	25.069	**9.999e-05**
Site	2	0.3580	0.17898	0.77581	0.6248
Height x Site	2	0.2786	0.1393	1.0576	0.3636

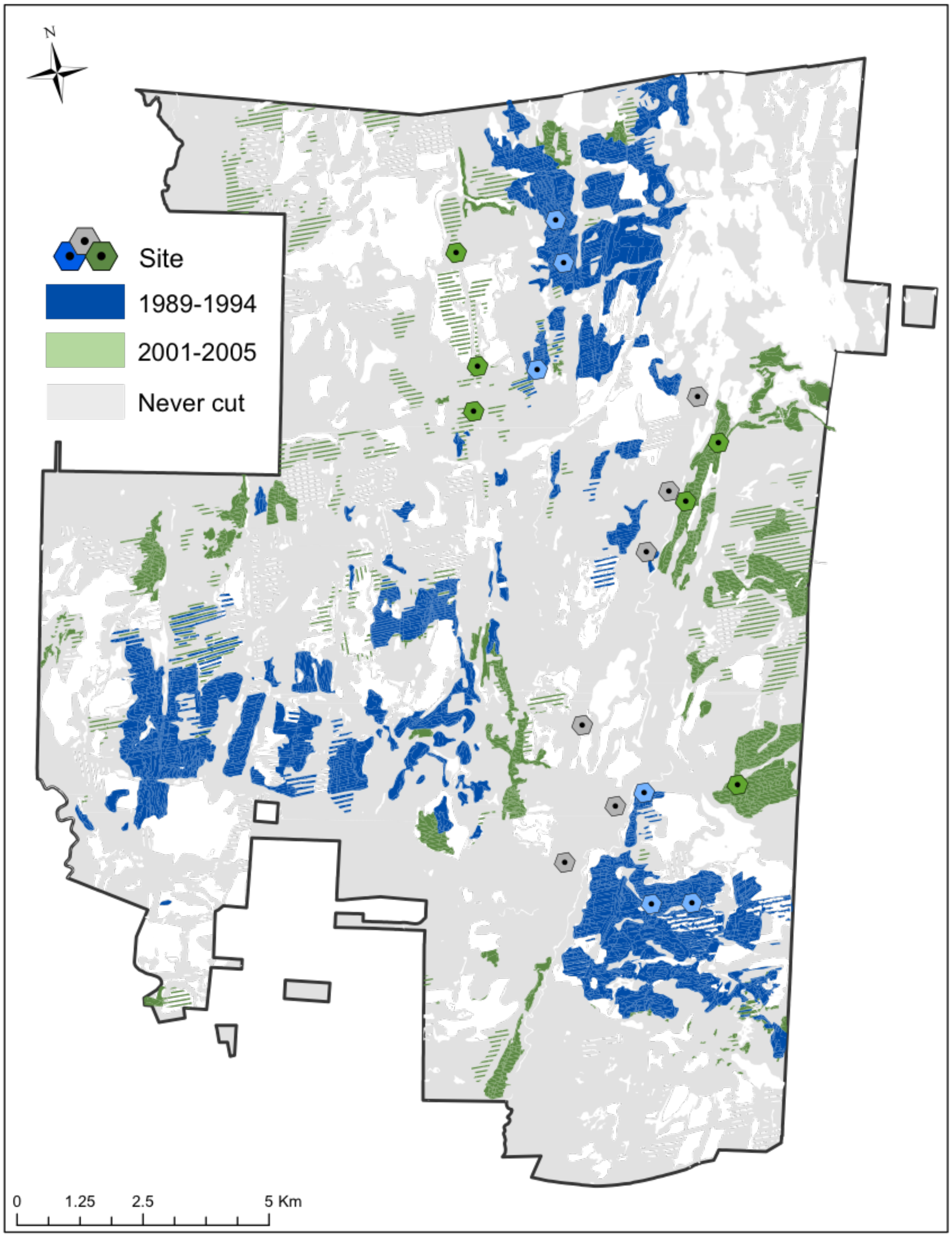

Figure 2.1. Map of the Kenauk Nature property with 2016 sampling sites. Shaded areas correspond to logging activity, with blue areas representing logging activity from 1989-1995, green areas representing logging activities from 2001-2005 and grey areas representing no logging activities. Sites sampled during 2016 are represented by hexagons and coloured based on which age class of logging they are located in. There are six sites representing each category.

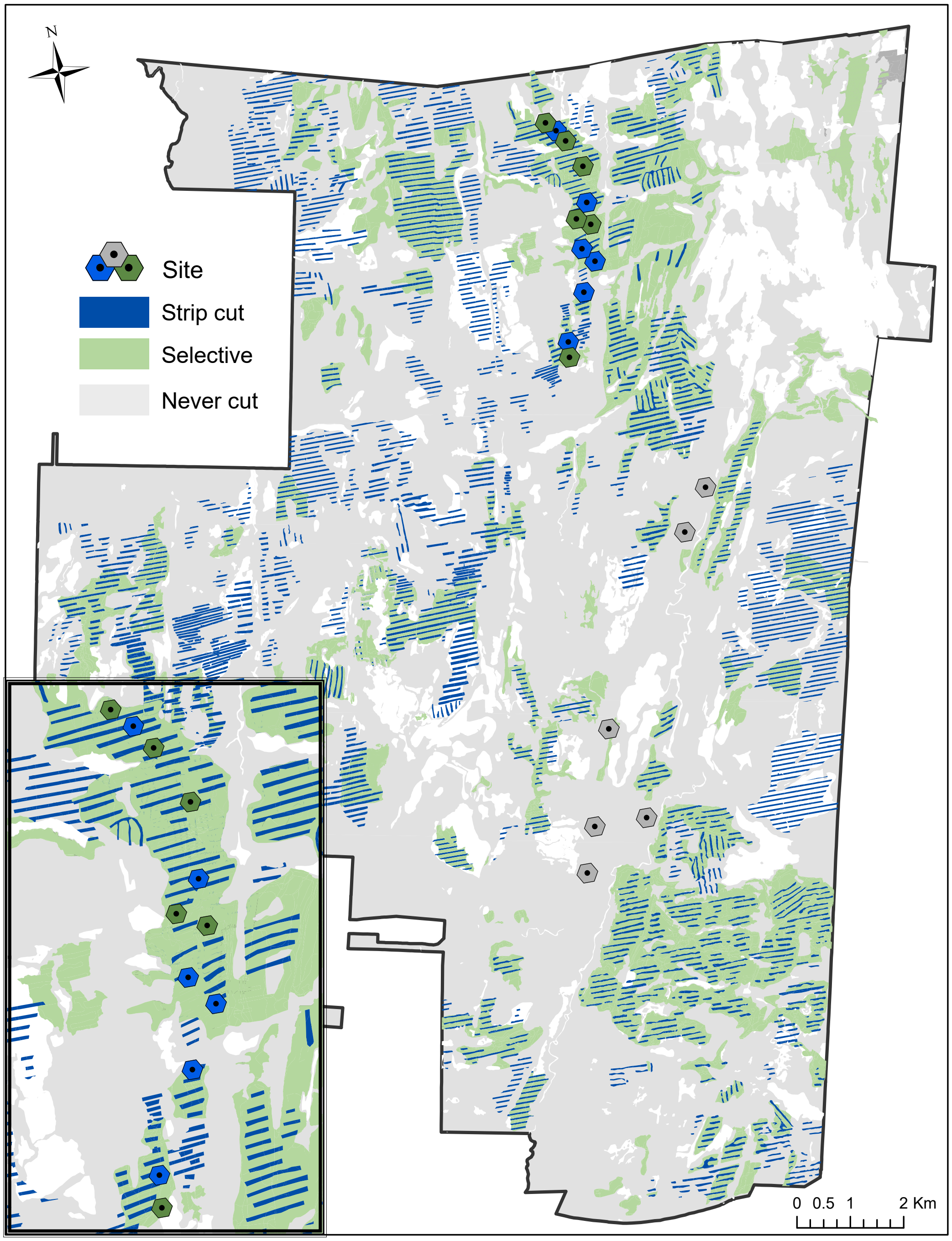

Figure 2.2. Map of the Kenauk Nature property with 2017 sampling sites. Shaded areas correspond to logging activity, with blue areas representing strip cutting, green areas representing selection cutting and grey areas representing no logging activities. Sites sampled during 2017 are represented by hexagons and coloured based on which type of logging they are located in. There are six sites representing each category. A close up of the twelve northern sites is given inset.

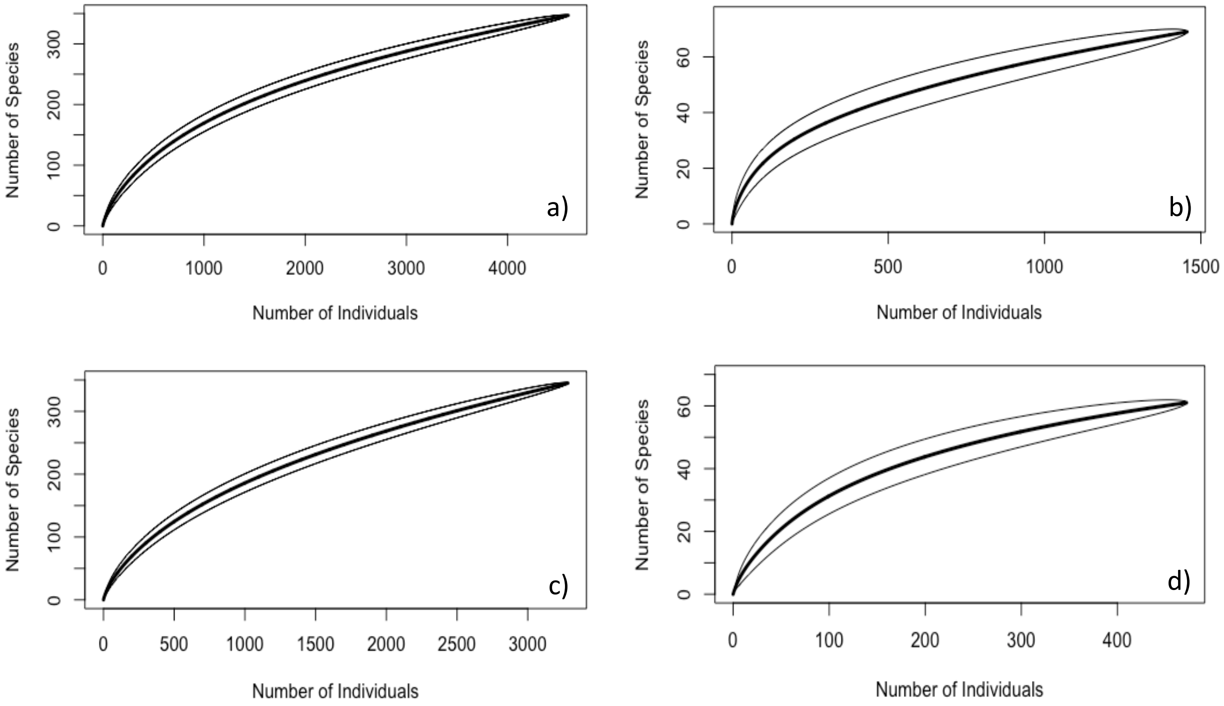

Figure 2.3. Individual-based rarefaction curves with 95% confidence intervals showing species diversity and accumulation of Coleoptera and Araneae in 2016 and 2017, where figure a) represents Coleoptera 2016, figure b) represents Araneae 2016, figure c) represents Coleoptera 2017 and figure d) represents Araneae 2017.

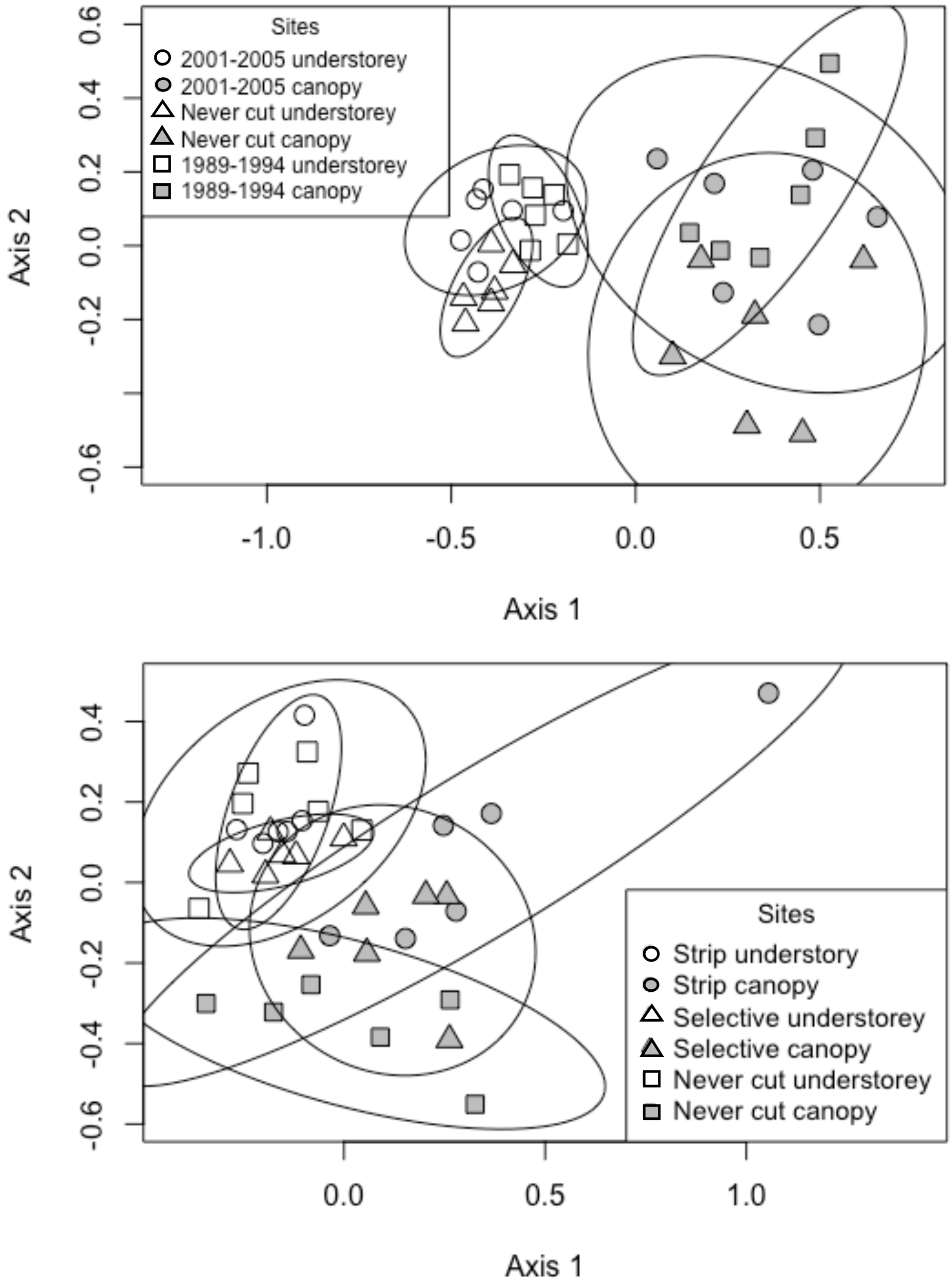

Figure 2.4. Non-metric multidimensional scaling (NMDS) of Coleoptera in a) 2016 and b) 2017 across all replicates using the log-values of family relative abundance. Each point represents a microhabitat, where in a), circles denote 2001-2005, triangles denote never cut, squares denote 1989-1994, white denotes the understorey and grey denotes the upper canopy. Similarly, in b), circles denote strip cut, triangles denote selection cut, squares denote never cut, white denotes the understorey and grey denotes the upper canopy. Points that are situated closer to each other are more similar in community

composition than those that are further apart. Height and forest management centroids surrounded by 95% confidence intervals were included. In both ordinations, there are significant differences between the understorey and upper canopy but no differences between forest management treatments.

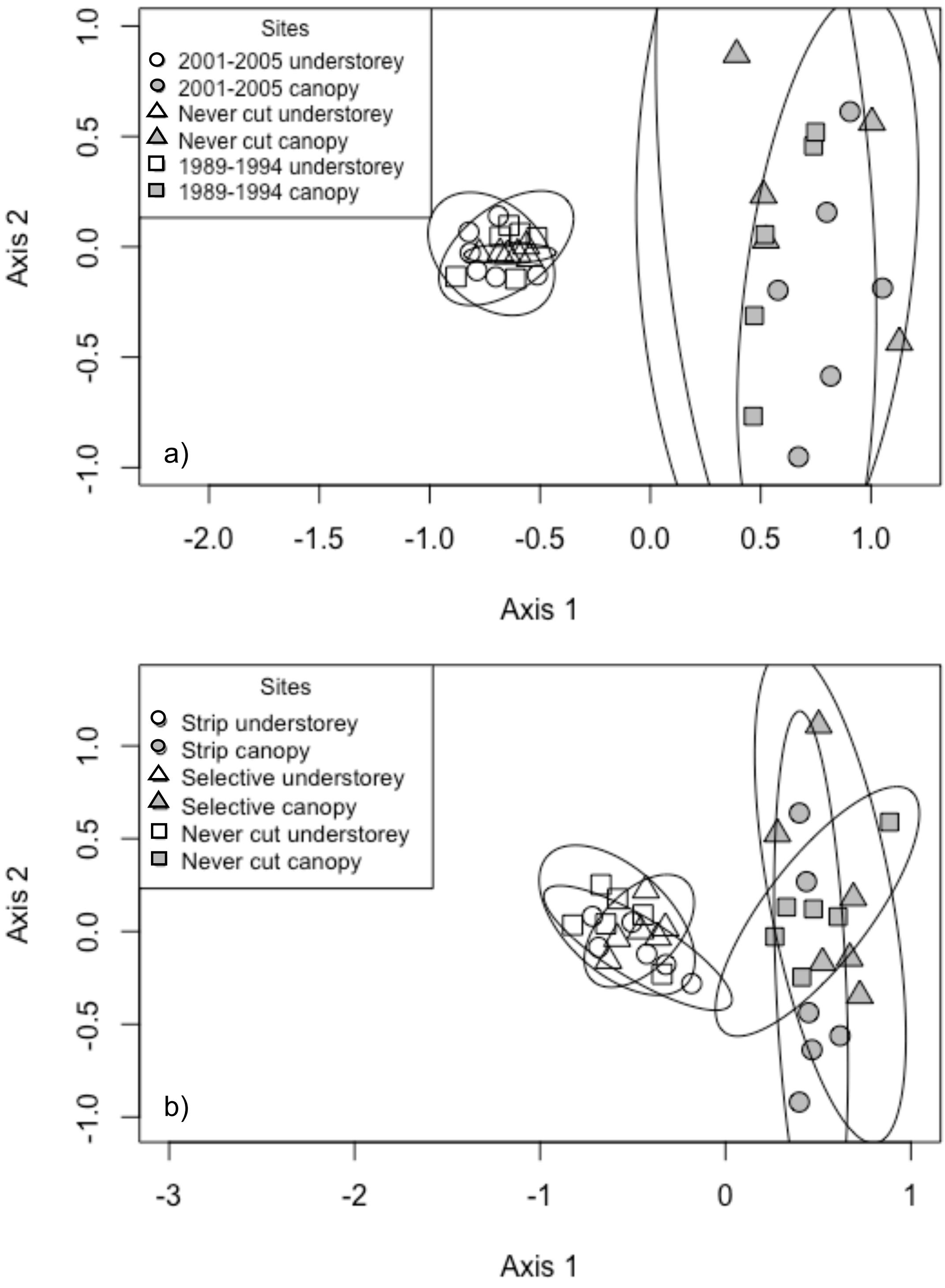

Figure 2.5. Non-metric multidimensional scaling (NMDS) of Araneae in a) 2016 and b) 2017 across all replicates using the log-values of family relative abundance. Each point represents a microhabitat, where in a), circles denote 2001-2005, triangles denote never cut, squares denote 1989-1994, white denotes the understorey and grey denotes the upper canopy. Similarly, circles denote strip cut, triangles denote selection cut,

squares denote never cut, white denotes the understorey and grey denotes the upper canopy in b). Points that are situated closer to each other are more similar in community composition than those that are further apart. Height and forest management centroids surrounded by 95% confidence intervals were included. In both ordinations, there are significant differences between the understorey and upper canopy but no differences between forest management treatments.

2.9 Connecting statement

Chapter 2 provided information on canopy arthropods and forest management impacts on arthropod community structure. It showed that the upper canopy is a reservoir of diversity and that future biodiversity assays must include canopy sampling. It also showed that arthropods assemble themselves vertically, even though there were no differences between forest management sites. Overall, this still calls for more sustainable forest management practices, as logging alters the upper canopy by creating canopy gaps or removing the canopy altogether. The final section of this thesis is a general summary and conclusion.

Thesis summary and conclusions

Chapter 1 outlined previous work done on vertical stratification and forest management. It explained the importance of canopy sampling and defended the use of arthropods in ecological studies. It also outlined the effect of forest management on diversity, broadly speaking, and showed that little work had been done on this in relation to arthropods in Southern Québec, let alone canopy arthropods.

Chapter 2 examined these topics, specifically vertical stratification and forest management, by using beetles and spiders as model taxa. Beetle diversity was higher in the understorey than in the upper canopy, whereas no difference was seen in spiders. Community composition of both beetles and spiders differed between height, indicating unique assemblages in the two strata. Beetle communities differed between forest management sites, but this significance was driven by family Latridiidae; when removed, the significance is lost. Spider communities did not differ between forest management sites. I attributed these results to the presence of intact canopy cover seen at my stands.

Based on these results, I suggest that future biodiversity assessments should include canopy sampling, especially within the context of forest management, as this stratum may contain a high diversity of unique arthropods. This thesis also suggested that forest management practices be limited in its frequency and intensity, as logging leads to the creation of canopy gaps or removes the canopy entirely and may homogenize the arthropod community composition. I call for the use of low-impact forest management techniques, specifically selection cutting, that do not severely compromise the forest canopy. I conclude that the effects of canopies and forest management on arthropod communities are intimately connected, and that any future work investigating forest management should include canopy sampling.

This thesis has provided the Kenauk Institute, and in association NCC, with baseline arthropod data, with the goal of conservation and preservation. The Kenauk Institute and NCC are interested in biodiversity monitoring on the property through the use of multiple focal taxa and through yearly sampling. This work is one of the first projects to provide them with data for use in conservation efforts; one other effort was established on the property and has been highly successful (the preservation of one of the last black maple (*Acer nigrum* F. Michx.) stands in Québec). I have not only provided the institute with data on arthropod assemblages, but have also been able to set up a canopy sampling protocol for all future work on the property. This will encourage canopy work on the property (a project on forest tent caterpillars (*Malacosoma disstria* Hubner) is already underway, for example), and eventually elsewhere as well.

Appendix A: Sampling periods from the 2016 and 2017 field seasons.

	Year	
	2016	**2017**
Sampling period 1	June 1-7	May 16-23
Sampling period 2	June 7-14	May 23-30
Sampling period 3	June 14-21	May 30-June 6
Sampling period 4	June 21-27	June 6-13
Sampling period 5	June 27-July 5	June 13-20
Sampling period 6	July 5-13	June 20-27
Sampling period 7	July 13-20	July 11-18
Sampling period 8	July 20-27	July 18-25
Sampling period 9	July 27-August 3	July 25-August 1
Sampling period 10	August 24-30	August 1-8
Sampling period 11	August 30-September 6	August 29-September 5
Sampling period 12	September 6-13	September 5-12

Appendix B: List of beetle and spider species collected in each of the forest management treatments and at both heights for both 2016 and 2017. Abundance values represent the pooled totals from all sampling periods, replicates and trap types. Bolded values denote family-level or habitat level totals. Note that juveniles are not included in the spider totals.

Coleoptera	2016							2017						
	Forest management/Height							Forest management/Height						
	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
Family/*Species*	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Carabidae							**729**							**558**
Acalanthus advena (LeConte 1846)	0	0	1	0	0	0	1	1	0	0	0	0	0	1
Agonoleptus conjunctus (Say 1823)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Agonum retractum (LeConte 1846)	0	0	0	0	0	0	0	7	0	3	0	1	0	11
Agonum 1 (Bonelli 1810)	6	0	3	0	2	0	11	3	0	1	0	0	0	4
Agonum 2 (Bonelli 1810)	2	0	3	0	0	0	5	5	0	4	0	0	0	9
Agonum 3 (Bonelli 1810)	2	0	0	0	0	0	2	0	0	0	0	0	0	0
Bembidion 1 (Latreille 1802)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Calathus gregarious (Say 1823)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Calosoma frigidum (Kirby 1837)	13	4	2	0	11	3	33	2	0	4	0	1	4	11
Carabus nemoralis (Müller 1764)	0	0	1	0	3	0	4	0	0	0	0	2	0	2
Clivina fossor (Linnaeus 1758)	2	0	0	0	0	0	2	0	0	0	0	0	0	0
Cymindis cribricollis (Dejean 1831)	4	0	0	0	0	0	4	0	0	0	0	0	0	0
Cymindis limbata (Dejean 1831)	3	2	2	2	0	1	10	0	2	1	5	0	0	8
Cymindis platicollis (Say 1823)	2	0	0	2	0	2	6	0	1	0	1	0	4	6
Cymindis unicolor (Kirby 1837)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Dromius piceus (Dejean 1831)	0	0	0	2	0	1	3	0	1	0	2	0	0	3
Myas cyanescens (Dejean 1828)	7	0	9	0	7	0	23	7	0	5	0	4	0	16
Oxypselaphus pusillus (Leconte 1854)	5	0	0	0	0	0	5	0	0	0	0	0	0	0

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Carabidae														
Paranchus albipes (Fabricius 1796)	0	0	0	0	0	0	0	1	0	0	0	1	0	2
Platynus decentis (Say 1823)	2	0	0	0	0	0	2	0	0	0	0	0	0	0
Platynus tenuicollis (Leconte 1846)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Poecilus lucublandus (Say 1823)	1	0	0	0	0	0	1	0	0	0	0	1	0	1
Pseudamara arenaria (LeConte 1847)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Pterostichus adoxus (Say 1823)	0	0	2	0	0	0	2	0	0	7	0	0	0	7
Pterostichus adstrictus (Eschscholtz 1823)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Pterostichus castor (Goulet and Bousquet 1983)	0	0	2	0	0	0	2	0	0	1	0	0	0	1
Pterostichus coracinus (Newman 1838)	20	0	77	0	19	0	116	47	0	50	0	41	0	138
Pterostichus corvinus (Dejean 1828)	1	1	2	0	0	0	4	4	0	1	0	0	0	5
Pterostichus diligendus (Chaudoir 1868)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Pterostichus lacrymosus (Newman 1838)	6	0	36	0	27	0	69	0	0	0	0	1	0	1
Pterostichus melanarius (Illiger 1798)	3	0	18	0	3	0	24	2	0	3	0	2	0	7
Pterostichus pensylvanicus (Leconte 1873)	1	0	3	0	0	0	4	0	0	1	0	0	0	1
Pterostichus tristis (Dejean 1828)	29	0	46	0	5	0	80	58	0	44	0	3	0	105
Sphaeroderus canadensis (Chaudoir 1861)	33	0	23	0	21	0	77	31	0	14	0	10	0	55
Sphaeroderus stenostomus lecontei (Dejean 1826)	3	0	5	0	4	0	12	0	0	0	0	0	0	0
Synuchus impunctatus (Say 1823)	69	1	141	0	11	0	222	67	1	61	1	21	0	151
Trechus rubens (Fabricius 1792)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Carabidae 2	0	0	0	0	0	0	0	6	0	3	0	0	0	9
Dytiscidae							**1**							**1**
Agabus 1 (Leach 1817)	0	0	0	0	0	1	1	1	0	0	0	0	0	1
Leiodidae							**9**							**2**
Agathidium 1 (Panzer 1797)	1	0	0	0	2	0	3	0	0	0	0	0	0	0
Catops 1 (Paykull 1798)	0	0	0	0	1	0	1	0	0	0	0	0	0	0

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Leiodidae														
Leiodes 1 (Latreille 1797)	2	0	0	0	3	0	5	0	0	0	0	0	0	0
Sciodrepoides 1 (Hatch 1933)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Leiodidae 13	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Silphidae							**13**							**5**
Necrophila americana (Linnaeus 1758)	4	0	0	0	0	0	4	0	0	0	0	1	0	1
Nicrophorus defodiens (Mannerheim 1846)	0	0	0	0	3	0	3	0	0	0	0	1	0	1
Nicrophorus orbicollis (Say 1825)	0	0	2	0	0	0	2	0	0	0	0	1	0	1
Nicrophorus sayi (Laporte 1840)	0	0	1	0	0	0	1	0	0	0	0	2	0	2
Oiceoptoma noveboracense (Forster 1771)	1	0	0	0	2	0	3	0	0	0	0	0	0	0
Staphylinidae							**272**							**124**
Aleochara castaneipennis (Mannerheim 1843)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Amischa 1 (Thomson 1858)	0	0	0	0	0	0	0	0	2	0	0	0	0	2
Apocellus sphaericollis (Say 1831)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Baeocera 1 (Erichson 1845)	5	0	4	0	4	0	13	0	0	0	0	2	0	2
Bisnius blandus (Gravenhorst 1806)	1	0	0	0	1	0	2	0	0	0	0	0	0	0
Bisnius 1 (Stephens 1829)	1	0	0	0	1	0	2	0	0	0	0	0	0	0
Bisnius 2 (Stephens 1829)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Bisnius 3 (Stephens 1829)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Bisnius 4 (Stephens 1829)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Bisnius 5 (Stephens 1829)	5	0	0	0	0	0	5	0	1	0	0	0	0	1
Bisnius 6 (Stephens 1829)	0	0	0	0	1	0	1	0	0	0	1	0	0	1
Bisnius 7 (Stephens 1829)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Bolitobius 1 (Leach 1819)	0	0	2	0	0	0	2	1	0	0	0	0	0	1
Bryoporus 1 (Kraatz 1857)	0	0	0	0	1	0	1	0	0	0	0	1	0	1
Charhyphus picipennis (LeConte 1863)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Dinothenarus badipes (LeConte 1863)	9	0	2	0	4	0	15	0	0	4	0	5	0	9

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Staphylinidae														
Eusphalerum convexum (Fauvel 1898)	0	0	0	0	0	0	0	0	1	0	0	1	0	2
Eusphalerum pothos (Mannerheim 1843)	0	0	1	0	0	0	1	0	1	0	0	0	0	1
Falagria dissecta (Erichson 1839)	1	0	0	0	1	0	2	0	0	0	0	0	0	0
Gabronthus thermarum (Aubé 1850)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Habrocerus 1 (Erichson 1839)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Lathrobium 1 (Gravenhorst 1802)	0	0	0	0	0	0	0	1	0	0	0	1	0	2
Lathrobium 2 (Gravenhorst 1802)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Lesteva pallipes (LeConte 1863)	3	0	0	0	0	0	3	0	0	0	0	1	0	1
Lordithon 1 (Thomson 1859)	0	0	0	0	0	0	0	1	0	1	0	0	0	2
Meronera venustula (Erichson 1839)	4	0	0	0	0	0	4	0	0	0	0	0	0	0
Neohypnus hamatus (Say 1834)	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Nudobius cephalus (Say 1834)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Olophrum obtectum (Erichson 1840)	0	0	1	0	1	0	2	0	0	0	0	0	0	0
Ontholestes cingulatus (Gravenhorst 1802)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Paederini 1	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Paederini 2	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Philonthus caeruleipennis (Mannerheim 1830)	25	0	20	0	22	0	67	0	0	0	0	0	0	0
Philonthus 1 (Stephens 1829)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Philonthus 2 (Stephens 1829)	2	0	0	0	0	0	2	0	0	0	0	0	0	0
Philonthus 3 (Stephens 1829)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Philonthus 4 (Stephens 1829)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Philonthus 5 (Stephens 1829)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Philonthus 1 (2017) (Stephens 1829)	0	0	0	0	0	0	0	2	0	0	0	1	0	3
Philonthus 2 (2017) (Stephens 1829)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Platydracus viridanus (Horn 1879)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Pselaphus 1 (Herbst 1792)	0	0	0	0	1	0	1	0	0	0	0	0	0	0

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Staphylinidae														
Pseudopsis 1 (Newman 1834)	2	0	0	0	1	0	3	0	0	0	0	0	0	0
Quedius peregrinus (Gravenhorst 1806)	8	0	6	0	1	0	15	17	1	11	0	1	0	30
Quedius 2 (Stephens 1832)	2	0	5	0	0	0	7	0	0	0	0	0	0	0
Quedius 4 (Stephens 1832)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Quedius 5 (Stephens 1832)	0	0	1	1	1	0	3	0	0	0	0	0	0	0
Reichenbachia propinqua (LeConte 1849)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Sepedophilus cinctulus (Erichson 1839)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Stenichnus 1 (Thomson 1859)	0	0	0	0	1	0	1	0	0	1	0	0	0	1
Scydmaenus 1 (LaPorte 1840)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Tachinus fumipennis (Say 1834)	57	1	22	0	18	0	98	10	0	6	1	20	0	37
Tachinus 1 (Gravenhorst 1802)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Tachinus 2 (Gravenhorst 1802)	1	0	0	0	0	0	1	1	0	0	0	0	0	1
Tachinus 3 (Gravenhorst 1802)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Tachinus 4 (Gravenhorst 1802)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Tachinus 5 (Gravenhorst 1802)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Tachinus 6 (Gravenhorst 1802)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Tachinus 7 (Gravenhorst 1802)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Tachinus luridus (?) (Erichson 1840)	0	0	2	0	0	0	2	0	0	0	0	0	0	0
Tasgius melanarius (Heer 1839)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Staphylinidae 3	0	0	0	0	0	0	0	1	0	1	0	1	0	3
Staphylinidae 32	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Staphylinidae 43	0	0	0	0	0	0	0	0	4	0	4	1	0	9
Staphylinidae 50	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Lucanidae							**1**							**1**
Dorcus parallelus (Say 1824)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Platycerus 1 (Geoffroy 1762)	0	0	0	0	0	0	0	1	0	0	0	0	0	1

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Trogidae							**25**							**2**
Trox variolatus (Fabricius 1775)	23	0	2	0	0	0	25	2	0	0	0	0	0	2
Geotrupidae							**9**							**4**
Geotrupes balyi (Jekel 1865)	2	0	2	0	0	0	4	3	0	0	0	0	0	3
Geotrupes hornii (Blanchard 1888)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Geotrupes semiopacus (Jekel 1865)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Geotrupes 1 (Latreille 1797)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Odonteus liebecki (Wallis 1928)	2	0	0	0	1	0	3	0	0	0	0	0	0	0
Scarabaeidae							**64**							**146**
Acrossus rubripennis (Horn 1870)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Aphodius 1 (Illiger 1798)	0	0	0	0	1	0	1	0	0	0	0	1	0	1
Ataenius 1 (Harold 1867)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Caelius humeralis (Brown 1931)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Dialytes 1 (Harold 1869)	1	0	4	0	0	0	5	0	0	0	0	0	0	0
Dialytes 2 (Harold 1869)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Dialytellus dialytoides (Fall 1907)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Dialytellus 2 (Brown 1929)	1	0	2	0	0	0	3	0	0	0	0	0	0	0
Dialytellus 3 (Brown 1929)	6	0	2	0	0	0	8	0	0	0	0	0	0	0
Dichelonyx 1 (2017) (Harris 1827)	0	0	0	0	1	0	1	5	1	3	0	1	0	10
Dichelonyx 2 (Harris 1827)	0	0	0	0	0	0	0	18	1	4	2	2	1	28
Dichelonyx 3 (Harris 1827)	0	0	0	0	0	0	0	3	0	0	0	0	0	3
Dichelonyx 4 (Harris 1827)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Dichelonyx subvittata (LeConte 1856)	0	0	0	0	0	0	0	1	0	0	0	1	2	4
Euphoria inda (Linnaeus 1758)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Macrodactylus subspinosus (Fabricius 1775)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Onthophagus orpheus canadensis (Fabricius 1801)	0	0	0	0	0	0	0	0	0	1	0	0	0	1

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Scarabaeidae														
Osmoderma eremicola (Knoch 1801)	0	1	0	0	0	1	2	0	0	0	0	0	0	0
Osmoderma scabra (Palisot de Beauvois 1805)	2	2	1	2	7	0	14	3	1	0	1	5	0	10
Serica tristis (LeConte 1850)	0	0	0	0	0	0	0	0	0	0	0	2	0	2
Serica 1 (MacLeay 1819)	8	0	10	0	7	1	26	0	0	0	0	1	1	2
Scarabaeidae 7	0	0	0	0	0	0	0	13	11	14	3	5	2	48
Scarabaeidae 30	0	0	0	0	0	0	0	9	6	6	4	5	2	32
Scarabaeidae 67	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Scirtidae							**10**							**14**
Contacyphon 1 (Des Gozis 1886)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Contacyphon variabilis (Thunberg 1785)	1	0	0	4	0	0	5	4	1	1	2	3	1	12
Prionocyphon limbatus (LeConte 1866)	2	2	0	1	0	0	5	1	0	0	0	0	0	1
Buprestidae							**7**							**5**
Agrilus 1 (Curtis 1825)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Dicerca callosa (Casey 1909)	0	0	0	1	1	2	4	0	0	0	0	0	0	0
Dicerca divaricata (Say 1823)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Dicerca 1 (Eschscholtz 1829)	0	1	0	0	0	1	2	0	0	0	2	1	1	4
Artematopodidae							**1**							**0**
Eurypogon harrisi (Westwood 1862)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Eucnemidae							**52**							**15**
Fornax bicolor (Melsheimer 1846)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Hylis frontosus (Say 1836)	0	5	1	0	4	0	10	0	0	1	1	0	1	3
Hylis terminalis (LeConte 1866)	0	4	0	0	6	4	14	0	0	0	1	2	0	3
Isarthrus 1 (LeConte 1852)	0	0	0	0	0	0	0	0	0	0	0	2	0	2
Isorhipis obliqua (Say 1836)	4	0	5	0	8	0	17	1	0	3	0	0	1	5
Microrhagus 1 (Dejean 1833)	1	0	0	0	2	0	3	0	0	0	0	0	0	0
Onichodon canadensis (Brown 1940)	1	0	0	0	1	3	5	0	1	0	0	1	0	2

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Eucnemidae														
Onichodon orchesides (Newman 1838)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Nematodes penetrans (LeConte 1852)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Elateridae							**251**							**394**
Aeolus 1 (Eschscholtz 1829)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Agriotes fucosus (LeConte 1853)	0	0	0	2	0	0	2	4	2	1	0	0	1	8
Agriotes stabilis (LeConte 1853)	0	2	1	0	5	0	8	0	2	0	1	0	3	6
Agriotes 1 (Eschscholtz 1829)	0	0	0	0	0	0	0	2	1	0	2	1	0	6
Ampedus apicatus (Say 1834)	3	1	1	0	0	0	5	0	1	0	0	2	1	4
Ampedus luctuosus (LeConte 1853)	1	0	0	1	0	0	2	0	0	0	1	0	0	1
Ampedus melsheimeri (Leng 1918)	0	2	0	1	0	0	3	0	0	0	0	1	1	2
Ampedus nigricollis (Herbst 1801)	0	0	0	0	0	0	0	0	1	0	1	0	0	2
Ampedus oblessus (Say 1834)	0	3	1	2	1	0	7	1	3	0	6	0	7	17
Ampedus protervus (LeConte 1853)	2	1	1	2	1	2	9	2	5	1	2	2	4	16
Ampedus rubricus (Say 1825)	1	5	0	2	0	3	11	0	4	0	6	0	18	28
Ampedus vitiosus (LeConte 1853)	1	1	0	0	0	0	2	0	0	0	2	0	0	2
Ampedus 1 (2017) (Germar 1844)	0	1	1	0	1	0	3	1	0	2	0	0	0	3
Ampedus 2 (2017) (Germar 1844)	0	0	0	0	0	0	0	0	3	0	2	0	4	9
Ampedus 3 (Germar 1844)	0	0	0	1	0	0	1	0	0	0	1	0	0	1
Ampedus 6 (Germar 1844)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Ampedus 7 (Germar 1844)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Ampedus 10 (2016) (Germar 1844)	0	0	1	0	0	0	1	1	0	0	0	0	0	1
Ampedus 11 (2016) (Germar 1844)	2	1	7	0	4	1	15	0	0	0	0	0	1	1
Ampedus 100 (Germar 1844)	1	0	0	0	0	0	1	0	0	0	1	0	0	1
Ampedus undescribed A	0	0	0	0	0	0	0	0	1	2	4	0	2	9
Ampedus undescribed B	0	1	0	0	0	0	1	0	0	0	0	0	1	1
Ampedus undescribed C	0	1	0	3	0	0	4	0	1	1	1	0	1	4

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Elateridae														
Ascoliocerus sanborni (Horn 1871)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Athous acanthus (Say 1839)	4	0	4	0	6	0	14	1	0	1	0	0	0	2
Athous brightwelli (Kirby 1837)	1	1	3	2	4	1	12	0	0	1	0	1	0	2
Athous cucullatus (Say 1825)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Athous posticus (Melsheimer 1845)	0	0	0	1	1	0	2	0	0	0	0	0	0	0
Athous rufifrons (Randall 1838)	1	0	0	0	1	0	2	0	0	1	0	2	0	3
Athous 1 (2017) (Eschscholtz 1829)	1	1	5	0	0	0	7	0	0	1	0	0	0	1
Athous 2 (2017) (Eschscholtz 1829)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Athous 3 (2017) (Eschscholtz 1829)	0	0	1	0	0	0	1	0	0	0	0	1	0	1
Athous 4 (Eschscholtz 1829)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Athous 5 (Eschscholtz 1829)	0	0	0	0	1	1	2	0	0	0	0	0	0	0
Cardiophorus spp 4 (Eschscholtz 1829)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Dalopius cognatus (Brown 1934)	5	0	2	0	0	0	7	0	0	0	0	0	0	0
Dalopius fuscipes (Brown 1934)	6	1	12	1	6	0	26	10	3	11	4	1	0	29
Dalopius gentilis (Brown 1934)	0	0	0	0	0	0	0	1	0	1	1	1	0	4
Dalopius vagus (Brown 1934)	1	0	2	0	0	0	3	2	0	8	0	0	0	10
Danosoma brevicorne (LeConte 1853)	0	0	0	0	0	0	0	1	1	1	0	0	0	3
Danosoma obtectum (Say 1836)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Denticollis denticornis (Kirby 1837)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Hadromorphus inflatus (Say 1825)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Hemicrepidius 1 (Germar 1839)	2	0	1	0	0	0	3	0	0	0	0	0	0	0
Hemicrepedius brevicollis (Candèze 1863)	0	1	0	0	2	1	4	1	0	0	0	0	0	1
Ligmargus lecontei (Leng 1918)	0	0	0	0	0	0	0	0	0	1	0	0	1	2
Ligmargus 1 (2017) (Stibick 1976)	0	0	0	0	0	0	0	0	0	1	0	0	2	3
Limonius confuses (Leconte 1853)	0	0	0	0	0	0	0	1	0	1	0	0	0	2
Limonius plebejus (Say 1825)	1	0	0	0	0	0	1	0	0	0	0	0	0	0

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Elateridae														
Limonius 1 (Eschscholtz 1829)	3	3	3	0	2	1	12	2	3	5	3	1	5	19
Limonius 3 (Eschscholtz 1829)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Liotrichus spinosus (LeConte 1853)	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Liotrichus 1 (Kiesenwetter 1858)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Margeiostus grandicollis (Leconte 1863)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Megapenthes rogersi (Horn 1871)	0	4	0	0	0	3	7	0	0	0	0	0	1	1
Megapenthes rufilabris (Germar 1844)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Megapenthes solitarius (Fall 1934)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Melanotus castanipes (Paykull 1800)	5	0	2	3	3	0	13	4	0	5	1	1	3	14
Melanotus communis (Gyllenhal 1817)	0	0	0	0	0	0	0	0	0	0	2	0	0	2
Melanotus corticinus (Say 1823)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Melanotus decumanus (Erichson 1842)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Melanotus ignobilis (Melsheimer 1844)	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Melanotus spadix (Erichson 1841)	0	0	0	0	0	0	0	0	1	0	0	0	1	2
Melatonus 1 (Eschscholtz 1829)	1	0	1	0	4	1	7	0	0	0	0	0	0	0
Neohypdonus tumescens (Leconte 1853)	4	0	11	0	0	0	15	8	0	10	0	3	0	21
Paractenicera fulvipes (Bland 1863)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Pseudanostirus hieroglyphicus (Say 1839)	1	2	3	5	1	0	12	2	7	1	5	1	11	27
Pseudanostirus triundulatus (Randall 1838)	0	1	0	2	0	0	3	1	0	1	2	0	1	5
Selatosomus pulcher (LeConte 1853)	1	1	0	2	0	2	6	56	0	21	10	7	2	96
Sericus honestus (Randall 1838)	0	0	1	0	0	0	1	0	0	1	0	0	0	1
Sericus viridanus (Say 1825)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Sylvanelater cylindriformis (Herbst 1806)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Elateridae 31	0	0	0	0	0	0	0	6	2	0	0	2	1	11
Elateridae 47	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Elateridae 84	0	0	0	0	0	0	0	0	0	0	2	0	0	2

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Lycidae							**4**							**1**
Dictyoptera aurora (Herbst 1784)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Greenarus thoracicus (Randall 1838)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Leptoceletes basalis (LeConte 1847)	1	0	0	0	2	0	3	0	0	0	0	0	0	0
Lampyridae							**12**							**17**
Ellychnia corrusca (Linnaeus 1767)	1	0	0	0	6	2	9	1	0	4	1	2	3	11
Lucidota atra (Olivier 1790)	0	0	1	0	1	0	2	0	0	0	0	0	1	1
Photinus obscurellus (LeConte 1851)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Pyractomena borealis (Randall 1838)	0	0	0	1	0	0	1	0	0	0	0	0	4	4
Cantharidae							**75**							**76**
Cantharis rufa (Linnaeus 1758)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Dichelotarsus cinctipennis (LeConte 1866)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dichelotarsus limbellus (LeConte 1881)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Dichelotarsus pattoni (LeConte 1866)	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Malthodes fragilis (LeConte 1851)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Pacificanthia rotundicollis (Say 1825)	3	1	7	3	3	2	19	1	0	2	1	1	3	8
Podabrus diadema (Fabricius 1798)	0	0	1	0	0	0	1	2	2	7	4	0	0	15
Podabrus modestus (Say 1823)	0	0	0	1	0	0	1	2	0	2	1	2	0	7
Podabrus planulus (Green 1947)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Podabrus tricostatus (Say 1835)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Rhagonycha fraxini (Say 1823)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Rhagonycha hirticula (Green 1941)	0	0	1	0	0	0	1	0	1	0	0	1	0	2
Rhagonycha imbecillis (LeConte 1851)	3	3	1	2	6	2	17	0	2	0	6	2	3	13
Rhagonycha luteicollis (Germar 1824)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Rhagonycha mollis (Fall 1936)	2	1	3	4	1	1	12	0	0	0	1	1	0	2
Rhagonycha oriflava (LeConte 1874)	1	3	0	1	0	0	5	0	0	0	0	0	2	2
Rhagonycha parvicolis (Green 1941)	0	0	1	0	2	0	3	0	0	1	1	0	0	2

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Cantharidae														
Rhagonycha recta (Melsheimer 1846)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Rhagonycha scitula (Say 1825)	0	0	0	3	1	0	4	5	0	0	2	2	3	12
Rhagonycha tantilla (LeConte 1881)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Rhagonycha tenuis (Green 1941)	0	1	1	0	0	0	2	1	0	0	0	1	1	3
Rhagonycha vilis (LeConte 1851)	0	0	2	0	0	1	3	0	0	0	1	0	0	1
Silis percomis (Say 1835)	0	0	0	0	1	0	1	0	1	2	0	0	0	3
Silis spathulata (LeConte 1881)	0	0	1	0	0	0	1	0	0	1	0	0	0	1
Dermestidae							**3**							**2**
Anthrenus castaneae (Melsheimer 1844)	1	0	0	0	0	0	1	0	0	0	0	0	1	1
Anthrenus museorum (Linnaeus 1761)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Anthrenus 1 (Müller 1764)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Dermestes signatus (Leconte 1874)	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Ptinidae							**9**							**6**
Euvrilletta harrisii (Fall 1905)	2	0	0	1	1	0	4	0	0	0	0	0	0	0
Hemicoelus pusillus (Fall 1905)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Oligomerus obtusus (LeConte 1865)	0	2	1	0	1	0	4	0	0	1	0	0	1	2
Ptilinus ruficornis (Say 1823)	0	0	0	0	0	0	0	0	0	1	1	1	0	3
Ptinidae 1	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Trogossitidae							**15**							**21**
Grynocharis quadrilineata (Melsheimer 1844)	0	0	0	0	4	0	4	3	2	4	1	1	1	12
Tenebroides 1 (Piller & Mitterpacher 1783)	0	0	0	0	1	1	2	0	2	1	4	0	1	8
Thymalus marginicollis (Chevrolat 1842)	2	0	1	0	5	1	9	0	0	1	0	0	0	1
Cleridae							**20**							**15**
Chariessa pilosa (Forster 1781)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Cymatodera bicolor (Say 1825)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Cymatodera inornata (Say 1835)	0	0	0	0	0	0	0	0	0	0	1	0	0	1

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Cleridae														
Enoclerus nigripes (Say 1823)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Placopterus thoracicus (Olivier 1795)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Phyllobaenus pallipennis (Say 1825)	0	1	0	0	0	0	1	0	2	0	0	0	1	3
Zenodosus sanguineus (Say 1835)	2	0	5	1	8	0	16	3	0	3	0	3	1	10
Erotylidae							**25**							**122**
Dacne quadrimaculata (Say 1835)	1	1	0	0	4	0	6	25	2	20	6	36	11	100
Ischyrus quadripunctatus (Olivier 1791)	0	1	0	0	2	0	3	0	1	0	0	0	0	1
Triplax dissimulator (Crotch 1873)	1	0	0	0	0	0	1	3	0	1	1	0	0	5
Triplax frosti (Casey 1924)	1	0	1	1	0	0	3	0	0	1	0	0	1	2
Triplax macra (LeConte 1854)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Triplax thoracica (Say 1825)	3	1	1	0	7	0	12	4	0	2	2	3	2	13
Monotomidae							**0**							**1**
Pycnotomina cavicollis (Horn 1879)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Cryptophagidae							**1**							**5**
Atomaria 1 (Stephens 1829)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Cryptophagus 1 (Herbst 1792)	0	1	0	0	0	0	1	0	0	1	0	0	1	2
Caenoscelis 1 (Thomson 1863)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Henotiderus centromaculatus (Reitter 1877)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Silvanidae							**0**							**1**
Dendrophagus cygnaei (Mannerheim 1846)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Cucujidae							**1**							**0**
Cucujus clavipes (Fabricius 1781)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Phalacridae							**8**							**13**
Stilbus apicalis (Melsheimer 1844)	1	1	0	3	3	0	8	0	0	0	0	11	2	13
Laemophloeidae							**1**							**4**
Laemophloeus bigutattus (Say 1827)	0	0	1	0	0	0	1	0	0	0	0	0	0	0

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Laemophloeidae														
Laemophloeus fasciatus (Melsheimer 1846)	0	0	0	0	0	0	0	1	0	0	2	0	0	3
Laemophloeidae 48	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Sphindidae							**0**							**1**
Eurysphindus comatulus (McHugh 1993)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Nitidulidae							**240**							**215**
Carpophilus brachypterus (Say 1825)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Carpophilus marginellus (Motschulsky 1858)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Carpophilus sayi (Parsons 1943)	0	0	2	0	6	1	9	0	2	0	0	0	0	2
Carpophilus 1 (Stephens 1830)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Carpophilus 2 (Stephens 1830)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Carpophilus 3 Stephens 1830)	0	0	0	2	2	0	4	0	0	0	0	0	0	0
Colopterus truncatus (Randall 1838)	1	0	0	1	0	0	2	0	1	0	1	1	1	4
Cryptarcha ampla (Erichson 1843)	0	0	0	0	1	0	1	0	0	0	1	0	0	1
Cryptarcha concinna (Melsheimer 1853)	0	0	0	0	0	0	0	0	2	0	0	0	0	2
Epuraea flavomaculata (Mäklin 1853)	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Epuraea rufida (Melsheimer 1846)	0	0	0	0	0	0	0	0	0	1	1	0	0	2
Epuraea 1 (Erichson 1843)	0	2	1	0	2	0	5	1	0	2	0	2	0	5
Epuraea 2 (Erichson 1843)	1	0	0	0	1	0	2	0	0	0	0	0	0	0
Epuraea 3 (Erichson 1843)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Glischrochilus fasciatus (Olivier 1790)	1	0	1	0	2	0	4	7	0	3	0	1	0	11
Glischrochilus quadrisignatus (Say 1835)	0	0	0	0	1	0	1	0	1	1	1	0	2	5
Glischrochilus sanguinolentus (Olivier 1790)	25	3	56	0	103	2	189	60	7	58	18	13	7	163
Glischrochilus siepmanni (Brown 1932)	0	0	2	0	1	0	3	1	0	1	0	0	0	2
Pallodes pallidus (Beauvois 1805)	0	0	0	0	1	0	1	0	0	0	0	1	0	1
Phenolia grossa (Fabricius 1801)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Stelidota 1 (Erichson 1843)	0	0	6	0	9	0	15	0	0	0	0	0	0	0

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Nitidulidae														
Nitidulidae 1	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Nitidulidae 10	0	0	0	0	0	0	0	2	0	1	1	0	0	4
Nitidulidae 20	0	0	0	0	0	0	0	1	4	3	0	0	0	8
Nitidulidae 45	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Cerylonidae							**0**							**1**
Cerylon 1 (Latreille 1802)	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Endomychidae							**23**							**11**
Danae testacea (Ziegler 1845)	3	0	3	0	5	0	11	3	0	0	0	0	0	3
Endomychus biguttatus (Say 1824)	2	0	0	0	4	0	6	0	0	0	0	0	0	0
Hadromychus chandleri (Bousquet and Leschen 2002)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Lycoperdina ferruginea (LeConte 1824)	0	0	1	0	1	0	2	0	0	0	0	0	0	0
Phymaphora pulchella (Newman 1838)	1	0	1	0	2	0	4	0	0	0	1	6	0	7
Coccinellidae							**11**							**24**
Anatis 1 (Mulsant 1846)	0	0	1	0	0	0	1	0	0	3	1	0	0	4
Chilocorus stigma (Say 1835)	0	0	1	0	0	0	1	0	1	1	0	0	0	2
Harmonia 1 (Mulsant 1850)	0	1	0	0	1	1	3	1	0	0	0	0	0	1
Hyperaspis bigeminata (Randall 1838)	0	0	0	0	0	1	1	1	4	0	3	0	1	9
Hyperaspis binotata (Say 1826)	1	2	1	1	0	0	5	1	3	1	2	0	1	8
Corylophidae							**2**							**2**
Clypastraea lunata (LeConte 1852)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Holopsis marginicollis (LeConte 1852)	0	1	0	0	1	0	2	0	0	0	0	0	0	0
Rypobius marinus (LeConte 1852)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Latridiidae							**1713**							**886**
Corticaria elongate (Gyllenhal 1827)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Corticaria serrata (Paykull 1800)	286	169	166	155	630	303	1709	114	52	108	112	297	195	878

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Latridiidae														
Corticarina cavicollis (Mannerheim 1844)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Cortinicara gibbosa (Herbst 1973)	0	0	0	0	0	0	0	0	0	2	1	0	0	3
Enicmus tenuicornis (LeConte 1878)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Enicmus 1 (Thomson 1859)	0	0	0	0	0	1	1	0	0	0	0	0	1	1
Melanophthalma pumila (LeConte 1855)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Latridiidae 15	0	0	0	0	0	0	0	0	0	2	1	0	0	3
Mycetophagidae							**4**							**1**
Mycetophagus pluripunctatus (LeConte 1856)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Mycetophagus serrulatus (Casey 1900)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Mycetophagus 1 (Hellwig 1792)	1	0	2	0	0	0	3	0	0	0	0	0	0	0
Ciidae							**3**							**6**
Cis fuscipes (Mellié 1848)	0	0	1	0	2	0	3	1	0	0	2	3	0	6
Tetratomidae							**18**							**11**
Eustrophus tomentosus (Say 1827)	3	0	0	0	10	0	13	0	1	2	0	1	0	4
Penthe pimelia (Fabricius 1801)	1	0	0	0	4	0	5	1	0	2	0	1	0	4
Synstrophus repandus (Horn 1888)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Tetratoma tessellata (Melsheimer 1844)	0	0	0	0	0	0	0	0	1	0	0	0	1	2
Melandryidae							**54**							**10**
Dircaea liturata (LeConte 1866)	2	0	3	0	3	2	10	0	0	1	0	2	0	3
Emmesa connectens (Newman 1838)	0	0	2	0	2	0	4	0	0	0	0	0	0	0
Emmesa labiata (Say 1824)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Emmesa 1 (Newman 1838)	0	2	0	0	0	0	2	0	0	0	0	1	0	1
Enchodes sericea (Haldeman 1848)	10	1	2	1	4	0	18	0	0	0	0	0	0	0
Hypulus simulator (Newman 1838)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Melandrya striata (Say 1824)	2	0	4	0	3	0	9	2	0	0	0	1	0	3
Orchesia castanea (Melsheimer 1846)	1	1	0	0	0	0	2	0	0	0	0	0	0	0

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Melandryidae														
Orchesia 1 (Latreille 1807)	1	0	0	0	0	0	1	0	0	0	0	0	1	1
Prothalpia undata (LeConte 1862)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Serropalpus 1 (Hellenius 1786)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Spilotus quadripustulatus (Melsheimer 1846)	0	0	1	3	0	1	5	0	0	0	0	0	0	0
Symphora flavicollis (Haldeman 1848)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Mordellidae							**74**							**5**
Mordella 1 (Linnaeus 1758)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Mordellaria serval (Say 1835)	4	7	9	5	14	10	49	0	0	0	1	0	1	2
Mordellaria 1 (Ermisch 1950)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Mordellina andreae (LeConte 1862)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Mordellina 1 (Schilsky 1908)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Mordellistena limbalis (Melsheimer 1845)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Mordellistena 1 (Costa 1854)	0	0	0	0	0	1	1	0	0	0	1	0	0	1
Paramordellaria triloba (Say 1824)	2	0	1	0	4	0	7	0	0	0	0	1	0	1
Tomoxia lineella (LeConte 1862)	1	0	1	5	4	0	11	0	0	0	1	0	0	1
Yakuhananomia bidentata (Say 1824)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Ripiphoridae							**1**							**0**
Pelecotoma flavipes (Melsheimer 1846)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Zopheridae							**7**							**1**
Endeitoma granulata (Say 1826)	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Synchita fuliginosa (Melsheimer 1846)	2	0	0	0	4	1	7	0	0	0	0	0	0	0
Tenebrionidae							**64**							**53**
Androchirus erythropus (Kirby 1837)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Arthromacra aenea (Say 1824)	0	0	2	2	0	0	4	0	0	0	0	1	0	1
Capnochroa fugilinosa (Melsheimer 1846)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Centronopus calcaratus (Fabricius 1798)	0	0	0	0	0	0	0	0	1	0	0	0	0	1

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Tenebrionidae														
Hymenochara rufipes (LeConte 1824)	0	1	0	3	2	0	6	0	0	0	1	2	1	4
Hymenorus 1 (Mulsant 1852)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Hymenorus (pilosus) (Mulsant 1852)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Hymenorus 1 (2017) (Mulsant 1852)	1	0	1	1	0	0	3	5	0	3	0	0	0	8
Hymenorus 2 (2017) (Mulsant 1852)	0	0	0	0	0	2	2	3	0	2	0	0	0	5
Hymenorus 3 (2017) (Mulsant 1852)	0	0	2	0	0	1	3	0	0	2	1	0	1	4
Hymenorus 4 (2017) (Mulsant 1852)	1	0	2	1	0	0	4	0	1	0	0	0	0	1
Isomira pulla (Melsheimer 1846)	1	0	2	2	0	0	5	0	0	0	0	0	0	0
Isomira quadristriata (Couper 1865)	1	0	0	0	0	0	1	2	1	2	0	2	0	7
Isomira sericea (Say 1824)	1	0	3	1	0	0	5	1	1	0	2	0	0	4
Lobopoda 1 (Solier 1835)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Mycetochara bicolor (Couper 1865)	1	0	0	0	2	0	3	0	0	0	0	0	0	0
Mycetochara binotata (Say 1824)	2	0	0	2	0	1	5	0	1	0	0	0	1	2
Mycetochara foveata (LeConte 1866)	0	0	0	2	1	0	3	0	0	0	0	0	2	2
Mycetochara fraterna (Say 1824)	0	3	0	8	1	0	12	0	3	0	2	0	1	6
Paratenetus punctatus (Spinola 1844)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Paratenetus 1 (Spinola 1844)	0	2	0	0	0	0	2	0	0	0	1	0	0	1
Tenebrionidae 1	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Tenebrionidae 19	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Tenebrionidae 28	0	0	0	0	0	0	0	2	0	1	0	2	0	5
Stenotrachelidae							**1**							**1**
Cephaloon lepturides (Newman 1838)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Cephaloon ungulare (LeConte 1874)	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Oedemeridae							**1**							**4**
Asclera puncticollis (Say 1823)	0	0	0	0	0	1	1	0	2	0	0	0	1	3
Asclera ruficollis (Say 1823)	0	0	0	0	0	0	0	1	0	0	0	0	0	1

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Pyrochoidae							**29**							**4**
Dendroides canadensis (Latreille 1810)	0	1	2	0	1	0	4	0	0	0	0	0	0	0
Dendroides concolor (Newman 1838)	3	0	0	0	2	0	5	0	0	0	0	1	0	1
Neopyrochroa flabellata (Fabricius 1787)	3	1	1	4	10	0	19	0	0	0	0	1	1	2
Pedilus 1 (Fischer von Waldheim 1820)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Schizotus cervicalis (Newman 1838)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Salpingidae							**2**							**0**
Rhinosinus viridiaeneus (Randall 1838)	0	0	0	0	2	0	2	0	0	0	0	0	0	0
Ischaliidae							**3**							**0**
Ischalia costata (LeConte 1862)	1	0	1	0	1	0	3	0	0	0	0	0	0	0
Aderidae							**6**							**2**
Zonantes fasciatus (Melsheimer 1846)	3	0	1	0	2	0	6	0	0	0	1	1	0	2
Scraptiidae							**5**							**6**
Anaspis 1 (Geoffroy 1762)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Canifa 1 (LeConte 1866)	0	0	0	0	0	0	0	0	0	0	0	3	1	4
Canifa 2 (LeConte 1866)	0	0	1	0	0	0	1	0	0	0	0	1	0	1
Scraptia sericea (Melsheimer 1846)	0	2	2	0	0	0	4	0	0	0	0	0	0	0
Cerambycidae							**246**							**123**
Aegomorphus modestus (Gyllenhal 1817)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Anastrangalia sanguinea (LeConte 1859)	0	0	0	0	1	0	1	0	0	0	0	0	1	1
Anelaphus villosus (Fabricius 1792)	0	2	0	3	0	0	5	0	0	0	0	0	1	1
Anthophylax attenuatus (Haldeman 1847)	0	0	2	0	1	0	3	3	0	0	0	0	0	3
Anthophylax cyaneus (Haldeman 1847)	0	0	0	1	0	0	1	1	3	0	2	0	2	8
Anthophylax viridis (LeConte 1850)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Bellamira scalaris (Say 1826)	0	0	0	1	0	0	1	0	1	0	1	0	0	2
Centrodera decolorata (Harris 1841)	0	0	0	1	0	0	1	0	0	0	0	1	0	1
Cyrtophorus verrucosus (Olivier 1795)	0	10	1	5	0	3	19	1	13	1	11	0	6	32

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Cerambycidae														
Evodinus monticola (Randall 1838)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Gaurotes cyanipennis (Say 1824)	0	0	2	1	1	1	5	1	0	0	2	0	1	4
Glycobius speciosus (Say 1828)	0	4	0	5	1	2	12	0	0	0	0	0	0	0
Grammoptera 1 (Audinet-Serville 1835)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Neoclytus acuminatus (Fabricius 1775)	0	2	0	1	0	1	4	0	0	0	0	0	0	0
Oplosia nubila (LeConte 1862)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Saperda imitans (Felt & Joutel 1904)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Saperda tridentata (Olivier 1795)	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Stenocorus schaumii (LeConte 1850)	0	5	0	0	0	0	5	0	0	0	0	0	0	0
Stenocorus vittiger (Randall 1838)	0	44	0	2	0	11	57	0	0	0	0	1	32	33
Stictoleptura canadensis (Olivier 1795)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Strangalepta abbreviata (Germar 1824)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Strophiona nitens (Forster 1771)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Trachysida mutabilis (Newman 1841))	0	5	6	0	8	1	20	0	5	1	4	3	0	13
Trigonarthris minnesotana (Casey 1913)	0	0	0	2	2	0	4	0	1	0	0	0	0	1
Trigonarthris proxima (Say 1824)	0	1	0	1	1	0	3	0	1	0	4	0	1	6
Urgleptes signatus (LeConte 1852)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Urographis despectus (LeConte 1850)	0	0	0	2	0	0	2	0	0	0	0	0	0	0
Urographis fasciatus (DeGeer 1775)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Xylotrechus colonus (Fabricius 1775)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Chrysomelidae							**10**							**17**
Capraita subvittata (Horn 1889)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Chaetocnema 1 (Stephens 1831)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Cryptocephalus 1 (Geoffroy 1762)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Donacia 1 (Fabricius 1775)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Paria 1 (LeConte 1858)	0	0	0	0	0	0	0	1	0	0	0	0	0	1

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Chrysomelidae														
Xanthonia decemnotata (Say 1824)	3	1	2	1	0	1	8	0	0	0	0	0	0	0
Xanthonia 1 (Baly 1863)	0	0	0	0	0	0	0	2	0	1	1	3	5	12
Xanthonia 2 (Baly 1863)	0	0	1	0	0	0	1	0	0	0	0	0	1	1
Anthribidae							**0**							**1**
Choragus sayi (Leconte 1876)	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Brachyceridae							**0**							**1**
Procas lecontei (Bedel 1879)	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Brentidae							**3**							**3**
Arrenodes minutus (Drury 1773)	0	1	0	1	0	1	3	0	0	0	0	0	3	3
Curculionidae							**485**							**353**
Acalles carinatus (LeConte 1876)	0	0	0	0	7	0	7	0	0	0	0	3	0	3
Anisandrus obesus (LeConte 1868)	0	0	0	0	0	0	0	6	0	0	0	0	0	6
Anisandrus sayi (Hopkins 1915)	0	0	0	0	1	0	1	2	0	2	0	0	0	4
Barypeithes pellucidus (Boheman 1834)	0	0	0	0	25	0	25	1	0	0	0	18	0	19
Curculio 1 (Linnaeus 1758)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Hexarthrum ulkei (Horn 1873)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Hylesinus aculeatus (Say 1824)	0	0	0	0	0	0	0	0	1	0	0	0	1	2
Monarthrum mali (Fitch 1855)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Orchestes 1 (Illiger 1798)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Otiorhynchus 1 (Germar 1822)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Phloeophagus minor (Horn 1873)	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Phloeophagus 1 (Schönherr 1838)	0	1	0	2	2	1	6	0	1	0	1	0	0	2
Phyllobius oblongus (Linnaeus 1758)	35	10	17	1	66	8	137	76	21	18	12	49	14	190
Rhyncolus 1 (Germar 1817)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Sciaphilus asperatus (Bonsdorff 1785)	43	0	45	0	43	0	131	29	0	47	1	28	0	105
Stenoscelis brevis (Boheman 1845)	3	12	1	4	7	4	31	0	2	0	2	0	2	6

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Curculionidae														
Tychius picirostris (Fabricius 1787)	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Xylosandrus spp 1 (Reitter 1913)	26	0	62	4	51	0	143	0	0	0	0	0	0	0
Xyloterinus politus (Say 1826)	0	0	0	0	0	0	0	6	0	0	0	0	0	6
Nanophyes marmoratus (Fabricius 1787)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Curculionidae 21	0	0	0	0	0	0	0	1	0	3	0	0	0	4
Curculionidae 44	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Unknown Family							**0**							**13**
Unknown Family 17	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Unknown Family 25	0	0	0	0	0	0	0	0	0	0	1	1	1	3
Unknown Family 38	0	0	0	0	0	0	0	0	0	0	0	5	0	5
Unknown Family 62	0	0	0	0	0	0	0	2	0	0	0	0	0	2
Unknown Family 82	0	0	0	0	0	0	0	0	0	0	0	0	2	2
Total	**989**	**395**	**1028**	**336**	**1440**	**435**	**4623**	**863**	**239**	**691**	**342**	**738**	**445**	**3318**

Araneae	2016							2017						
	Forest management/Height							Forest management/Height						
	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
Family/*Species*	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Agelenidae							**1**							**0**
Agelenopsis utahana (Chamberlin & Ivie 1933)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Amaurobiidae							**99**							**91**
Amaurobius borealis (Emerton 1909)	0	0	1	0	0	0	1	5	0	0	0	0	0	5
Callobius bennetti (Blackwall 1846)	0	0	1	0	1	0	2	0	0	0	0	0	0	0
Callobius nomeus (Chamberlin 1919)	0	0	0	0	0	0	0	0	0	2	0	2	0	4
Coras juvenilis (Keyserling 1881)	1	0	3	0	0	0	4	0	0	0	0	0	0	0
Wadotes calcaratus (Keyserling 1887)	18	0	21	0	23	0	62	24	0	25	0	22	0	71
Wadotes hybridus (Emerton 1890)	6	0	10	0	14	0	30	4	0	4	0	3	0	11
Araneidae							**10**							**19**
Araneus guttulatus (Walckenaer 1841)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Araneus marmoreus (Clerck 1757)	1	0	2	0	2	0	5	1	0	1	0	0	0	2
Araneus nordmanni (Thorell 1870)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Araneus saevus (L. Koch 1872)	0	0	0	0	0	1	1	0	0	0	1	0	0	1
Araniella displicata (Hentz 1847)	0	0	1	0	0	0	1	0	3	2	5	0	3	13
Eustala anastera (Walckenaer 1841)	0	1	0	0	0	0	1	0	0	0	1	0	0	1
Neoscona arabesca (Walckenaer, 1841)	1	0	0	0	0	0	1	0	0	0	0	0	1	1
Clubionidae							**28**							**22**
Clubiona canadensis (Emerton 1890)	2	0	1	1	1	0	5	1	1	4	0	0	0	6
Clubiona kastoni (Gertsch 1941)	1	0	0	0	0	0	1	1	0	0	0	0	0	1
Clubiona obesa (Hentz 1847)	0	0	0	0	2	1	3	1	1	0	0	1	0	3
Clubiona spiralis (Emerton 1909)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Elaver excepta (L. Koch 1866)	1	0	7	5	2	4	19	0	3	1	5	1	1	11
Corinnidae							**13**							**0**
Castianeira cingulata (C. L. Koch 1841)	8	0	2	0	3	0	13	0	0	0	0	0	0	0
Dictynidae							**44**							**29**
Cicurina arcuata (Keyserling 1887)	6	0	4	0	5	0	15	6	0	2	0	3	0	11
Cicurina brevis (Emerton 1890)	3	0	10	0	1	0	14	3	0	1	0	2	0	6
Cicurina itasca (Chamberlin & Ivie 1940)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Cicurina pallida (Keyserling 1887)	0	0	0	0	0	0	0	1	0	1	0	1	0	3

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UN	UP	UN	UP	UN	**Total**
Dictynidae														
Cicurina robusta (Simon 1886)	6	0	4	0	4	0	14	7	0	1	0	0	0	8
Emblyna sublata (Hentz 1850)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Gnaphosidae							**2**							**4**
Drassyllus eremitus (Chamberlin 1922)	0	0	0	0	**0**	**0**	0	0	0	0	0	1	0	1
Drassyllus fallens (Chamberlin 1922)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Zelotes fratris (Chamberlin 1920)	0	0	0	0	1	0	1	2	0	1	0	0	0	3
Hahniidae							**292**							**43**
Neoantistea magna (Keyserling 1887)	93	0	100	0	99	0	292	14	0	14	0	15	0	43
Linyphiidae							**227**							**56**
Agyneta serrata (Emerton 1909)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Bathyphantes pallidus (Banks 1892)	8	0	0	0	1	0	9	0	0	0	0	0	0	0
Centromerus persolutus (O. P.-Cambridge 1875)	0	0	0	0	0	0	0	0	0	2	0	0	0	2
Centromerus sylvaticus (Blackwall 1841)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Ceraticelus fissiceps (O. P.-Cambridge 1874)	2	0	0	0	1	0	3	0	2	1	0	0	0	3
Collinsia oxypaederotipus (Crosby 1905)	11	0	7	0	2	0	20	5	1	1	0	2	0	9
Diplocephalus subrostratus (O. P.-Cambridge 1873)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Drapetisca alteranda (Chamberlin 1909)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Floricomus praedesignatus (Bishop & Crosby 1935)	4	0	0	0	3	0	7	0	0	0	0	0	0	0
Grammonota angusta (Dondale 1959)	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Lepthyphantes intricatus (Emerton 1911)	2	0	0	0	0	0	2	3	0	1	0	0	0	4
Lepthyphantes turbatrix (O. P.-Cambridge 1877)	1	0	0	0	0	0	1	0	0	1	0	0	0	1
Mermessus maculatus (Banks 1892)	0	0	3	0	1	0	4	0	0	2	0	0	0	2
Microneta viaria (Blackwall 1841)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Neriene clathrata (Sundevall 1830)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
*Neriene radiata (*Walckenaer 1841)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Pityohyphantes costatus (Hentz 1850)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Pityohyphantes limitaneus (Emerton 1915)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Pityohyphantes subarcticus (Chamberlin & Ivie 1943)	1	0	0	1	0	0	2	0	0	0	0	0	0	0
Pocadicnemis americana (Millidge 1976)	28	0	26	0	38	0	92	3	0	3	0	2	0	8

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Linyphiidae														
Praestigia kulczynskii (Eskov 1979)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Tapinocyba simplex (Emerton 1882)	1	0	0	0	1	0	2	0	0	0	0	0	0	0
Taranucnus ornithes (Barrows 1940)	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Tenuiphantes zebra (Emerton 1882)	6	0	4	0	7	0	17	6	0	1	0	0	0	7
Tmeticus ornatus (Emerton 1914)	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Wabasso quaestio (Chamberlin 1949)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Walckenaeria atrotibialis (O. P.-Cambridge 1878)	3	0	2	0	10	0	15	0	0	1	0	3	0	4
Walckenaeria communis (Emerton 1882)	9	0	3	0	18	0	30	0	0	1	0	4	0	5
Walckenaeria digitata (Emerton 1913)	1	0	2	0	1	0	4	0	0	0	0	1	0	1
Walckenaeria directa (O. P.-Cambridge 1874)	0	0	0	0	0	0	0	3	0	1	0	2	0	6
Walckenaeria minuta (Emerton 1882)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Walckenaeria pallida (Emerton 1882)	3	0	0	0	4	0	7	0	0	0	0	0	0	0
Walckanaeria spiralis (Emerton 1882)	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Liocranidae							**89**							**22**
Agroeca ornata (Banks 1892)	17	0	13	0	4	0	34	3	0	2	0	3	0	8
Phrurotimpus alarius (Hentz 1847)	13	0	15	0	27	0	55	4	0	4	0	6	0	14
Lycosidae							**581**							**139**
Pardosa moesta (Banks 1892)	0	0	0	0	0	0	0	1	0	1	0	0	0	2
Pardosa xerampelina (Keyserling 1877)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pirata montanus (Emerton 1885)	173	0	134	0	271	0	578	30	0	26	0	78	0	134
Schizocosa ocreata (Hentz 1844)	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Schizocosa saltatrix (Hentz 1844)	0	0	0	0	0	0	0	0	0	2	0	0	0	2
Trochosa ruricola (De Geer 1778)	0	0	0	0	2	0	2	0	0	0	0	0	0	0
Trochosa terricola (Thorell 1856)	0	0	0	0	0	0	0	0	0	1	0	0	0	1
Philodromidae							**1**							**4**
Philodromus praelustris (Keyserling 1880)	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Philodromus rufus vibrans (Dondale 1964)	0	0	0	0	1	0	1	0	0	0	1	0	1	2
Philodromus vulgaris (Hentz 1847)	0	0	0	0	0	0	0	0	1	0	0	0	0	1

	2001-2005		1989-1994		Never cut			Strip cut		Selection cut		Never cut		
	UN	UP	UN	UP	UN	UP	**Total**	UN	UP	UN	UP	UN	UP	**Total**
Salticidae							**21**							**7**
Chinattus parvulus (Banks 1895)	0	0	0	0	3	0	3	0	0	0	0	0	0	0
Neon nelli (Peckham & Peckham 1888)	1	0	0	0	1	0	2	0	0	2	0	0	0	2
Pelegrina flavipes (Peckham & Peckham 1888)	1	0	0	2	0	1	4	0	0	0	0	0	0	0
Pelegrina insignis (Banks 1892)	1	0	0	0	1	0	0	0	0	0	1	1	0	2
Pelegrina proterva (Walckenaer 1837)	2	0	4	2	3	1	12	1	0	1	1	0	0	3
Theridiidae							**30**							**20**
Dipoena nigra (Emerton 1882)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Enoplognatha latimana (Hippa & Oksala 1982)	0	0	0	0	0	0	0	0	0	1	0	1	0	2
Enoplognatha ovata (Clerck 1757)	0	0	0	0	2	0	2	0	0	0	0	0	0	0
Robertus riparius (Keyserling 1886)	2	0	9	0	10	0	21	7	0	3	0	0	0	10
Theridion frondeum (Hentz 1850)	1	0	0	0	0	0	1	0	0	1	0	0	0	1
Theridion murarium (Emerton 1882)	0	1	0	0	0	1	2	0	0	0	1	0	3	4
Yunohamella lyrica (Walckenaer 1841)	1	0	1	1	1	0	4	1	0	0	1	0	0	2
Thomisidae							**18**							**16**
Bassaniana utahensis (Gertsch 1932)	1	0	0	0	0	0	1	0	1	0	0	0	1	2
Ozyptila distans (Dondale & Redner 1975)	5	0	1	0	8	0	14	7	0	1	0	3	0	11
Xysticus elegans (Keyserling 1880)	0	0	0	0	3	0	3	1	0	1	0	0	0	2
Xysticus ferox (Hentz 1847)	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Total	**449**	**2**	**394**	**12**	**589**	**10**	**1456**	**149**	**14**	**124**	**17**	**158**	**10**	**472**