**EFFECTS OF WILDFIRE AND CLEAR-CUTTING ON GROUND LEVEL
SPIDER ASSEMBLAGES IN A BOREAL FOREST**

By

Maxim Larivée

A thesis submitted to

The Faculty of Graduate Studies and Research

In partial fulfilment of

The requirements for the degree of

Masters of Science

Department of Biology

Carleton University

Ottawa, Ontario

March 2003

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**EFFECTS OF WILDFIRE AND CLEAR-CUTTING ON GROUND LIVING
SPIDER ASSEMBLAGES IN A BOREAL FOREST**

Submitted by

Maxim Larivée, B.Sc.H.

in partial fulfillment of the requirements
for the degree of Masters of Science

Chair, Department of Biology

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ABSTRACT

I examined how ground living spider assemblages respond to recently burned and recently cut Black Spruce (*Picea mariana*, (Mill.)) forest stands inside the disturbed stands and at their edges in eastern Québec, Canada. My main objective was to determine if wildfire and clear-cutting affect the structure of the forest floor in different ways, resulting in different spider assemblages inside the disturbed stands and at their edges. Firstly, environmental variables associated with the structure of the forest floor were measured at every site and beside each trap along the edge transects. Over 9000 individuals from 124 species of spiders were identified throughout the study. Burned stands had higher shrub cover, less coarse woody debris and thicker litter and moss-lichen substrate with an overall more homogenous forest floor structure than the clear-cut stands. Correspondence analysis showed that the composition of the spider assemblages was different in all three types of stand. Spider abundance and alpha diversity (mean richness per site) of hunting spiders were the highest in the clear-cut sites, which also had the lowest species turnover between stands. Environmental variables at the edges of wildfire and clear-cut stands did not differ in their response to the edge but a slightly more abrupt change in the ground level spider assemblages was detected at the clear-cut edges through correspondence analyses. The depth of edge influence created by both disturbances on open and interior habitat spider guilds was approximately 30 m on each side of the edge. I discuss the implication of my results in the context of sustainable management of the boreal forests.

ACKNOWLEDGEMENTS

I am most grateful in the realization of this thesis to my directors, Dr. Lenore Fahrig and Dr. Pierre Drapeau who were able to put all their trust and belief in me in order to make this study a reality. The invaluable guidance and knowledge that they shared with me over the past two years have made me grow into a more accomplished ecologist and person. Needless to say, the financial support they provided me was essential in the success of this thesis and my personal life. I also would like to thank Dr. Stewart Peck and Dr. Francis Pick for their ideas, comments and guidance throughout my masters.

I never knew how lucky I was to end up in Dr. Fahrig's Landscape Ecology Lab before I set foot in it and realized that not only I had found a great director but also I was now part of one the keenest labs in the country for ecology. Through all my interactions with the other members of the Lab, I was able to extend and deepen my understanding of biological and statistical concepts in ways that I had not expected. To all those who were present during my two years (and a bit more!) please accept my warmest gratitude. It was a joy to work with you guys everyday!

Special thanks to the Canadian National Collection of arthropods, Dr. Charles Dondale, Pierre Paquin, Raymond Hutchinson and Don Buckle for all their help during the spider identification period. Without them, I would probably still be identifying my spiders! Also, thank you very much Michel St-Germain for your help during fieldwork and sharing your views and opinions with me regarding my work and science in general but also for becoming a very good friend of mine. Many thanks to Dave Omond for your precious help in preparing the fieldwork and to Marie-Pierre Brunet and Annie-Ève

Gagnon for their assistance during fieldwork. Also thanks to all the others partners who contributed to the realization of my project. They are the Chaire industrielle en aménagement forestier, the Québec Ministers of Natural Resources, the Groupe de Recherche en Aménagement Forestier Inter-universitaire (GREFI) and Abitibi-Consolidated for access to their sites and Société de la Faune et des Parcs for use of their facilities.

Merci beaucoup mes parents for being my mentors but also my number one fans and for supporting me all the way in everything I do.

Lastly, to my Fiancé Michelle who gives me everyday unconditional love and support to carry on my passions and goals, I love you!

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GENERAL INTRODUCTION

Boreal forest dynamics are governed by natural fire disturbance regimes (Attiwill 1994, Esseen *et al.* 1997, Bergeron *et al.* 1999). Within a sustainable development framework, the forestry industry is orienting its management efforts towards developing forest harvesting practices based on the patterns and processes that occur under this disturbance regime (Niemela 1999). Current forestry practices aim at disturbing the ecosystem in a similar fashion to natural disturbances under the premise that organisms living in the disturbed systems are pre-adapted to these disturbances. For this management approach to succeed the assumptions of similar effects created by logging and natural disturbances must hold true.

Comparative studies regarding the effects of logging and wildfires on several taxa and abiotic components of the boreal forest have been conducted in the last decade (Mackay and Robinson 1987, Wei and Kimmins 1998, Imbeau *et al.* 1999, Nguyen-Xuan *et al.* 2000, Carignan *et al.* 2000, Drapeau *et al.* 2002). In all those cases different responses resulting from the forestry practices and the wildfires were found. Buddle *et al.* (2000) produced the only comparative study on arthropods in Canada, in the mixed-wood boreal plains of Alberta. They found that web-building spiders associated with mature forests either survived or re-colonized recently cut stands more rapidly than burned stands. In addition, spider assemblages found in burned stands early in succession had a more even distribution and contained spider species found only in the burned stands.

Spiders (Araneae) are a sensitive taxonomic group well suited for studies comparing disturbed ecosystems by clear-cuts and wildfire. First, they are very abundant (Huhta 1971, Coyle 1981, Jennings *et al.* 1988, Coddington *et al.* 1996, Buddle *et al.* 2000). Also, individual species distribution and abundances are mediated by structural attributes of the habitat (Uetz 1992, Hatley and Macmahon 1980). Spiders act as predators and prey in the food chain (Atlegrim and Sjoberg 1995, Nyffeler 2000), and contribute to nutrient recycling and fixation in recently disturbed habitats (Hodkinson *et al.* 2001).

This study examines how ground living spider assemblages respond to recently burned and recently cut Black Spruce (*Picea mariana*, (Mill.)) forest stands inside disturbed stands and at their edges. Wildfires and clear-cuts kill the tree layer (and remove it in the case of clear-cuts) and disturb the forest floor creating changes in the structure and the microclimatic conditions of the forest floor (Matlack 1993, Chen *et al.* 1995, Nguyen-Xuan *et al.* 2000). These environmental changes affect the structure and composition of the spider assemblages, their diversity and abundance. I examined the differences in forest floor conditions in stands recently submitted to a wildfire and stands recently clear-cut to see if those differences mediate a differential response of the ground level spider assemblages found in these stands. In addition, I analysed how the spider assemblages are affected by the changes in environmental conditions at the edges of both disturbances.

CHAPTER 1

COMPARING THE EFFECTS OF WILDFIRE AND CLEAR-CUTTING ON GROUND LEVEL SPIDER ASSEMBLAGES OF A BOREAL ECOSYSTEM

1.1 Introduction

Over the last decade research on sustainable management of the boreal forests has aimed towards developing forest harvesting practices that are based on our understanding of patterns and processes that occur under natural disturbance regimes (Attiwill 1994, Anglestam 1998, Bergeron *et al.* 2001, Simberloff 2001). The strategy is to conduct logging such that the disturbance pattern created on the landscape mimics the spatial and temporal disturbance pattern created by natural disturbances such as a wildfire. The organisms associated with the boreal forest have evolved in the presence of natural disturbances over long time periods (Bergeron *et al.* 2001, OMNR 2001). Therefore it is assumed that they are pre-adapted to management regimes that mimic landscape patterns and processes occurring under natural disturbances, and that they will persist under such management regimes. However, this depends on the assumption that the effects of logging on boreal forest organisms and ecological processes are equivalent to the effects of natural disturbances (Niemela 1999, Nguyen-Xuan *et al.* 2000).

Since the early 1980's several comparative studies have examined the effects of wildfires and clear-cuts on various components and organisms of boreal forests. Differences in the effects of the two disturbances were found for nitrogen levels (Wei and

Kimmins 1998), understory plants and shrubs (Johnston and Elliott 1996, Nguyen-Xuan et al. 2000), lake water quality (Carignan et al. 2000), and bird communities (Hobson and Schieck 1999, Imbeau et al. 1999, Schieck and Hobson 2000, Simons et al. 2002, Drapeau et al. 2002). Small mammal abundances, on the other hand, were not affected differently by clear-cuts and wildfires (Simon et al. 2002). Only one comparative study looked at the effects of wildfire and wood harvesting on arthropods. In mixed-wood forests of the boreal plain in Alberta, Buddle et al. (2000) found that web-building spiders associated with mature forests either survived or re-colonized recently cut stands faster than burned stands and they also showed that spider assemblages originating from burned stands early in succession had a more even distribution and contained several spiders species found only in the burned stands.

Spiders (Araneae) are particularly well suited for comparing the effects of clear-cuts and wildfire at the ground level. They are generally found in high abundance (Huhta 1971, Coyle 1981, Jennings et al. 1988, Coddington et al. 1996, Buddle et al. 2000). Individual species distributions and abundances are tightly linked to the structural attributes of the habitat (Uetz 1979, Hatley and Macmahon 1980). They are key elements of the food chain (Atlegrim and Sjoberg 1995, Nyffeler 2000), and are known to be involved in nutrient recycling and fixation in early stages of succession in disturbed habitats (Hodkinson et al. 2001).

The objective of this study was to investigate the ground level spider assemblage response to fire and logging in Black Spruce (*Picea mariana* (Mill.)) dominated stands in

the boreal forest of Québec, Canada. I hypothesized that wildfires and clear-cuts affect the structure of the forest floor in different ways, resulting in different spider assemblages. I expected that unburned patches of ground surviving the wildfire, and the varying severity of burned ground at the local scale would enhance the structural complexity of the forest floor present after a wildfire, producing richer spider assemblages than the harvested sites.

1.2 Material and Methods

Study area

The study was conducted during the summer of 2001 in the Park des Grands-Jardins (the "park") (latitude 47° 41' N longitude -70° 41' O) and in the Réserve Faunique des Laurentides (the "reserve"), both located in the Charlevoix region, 150 km northeast of Québec city, Canada (Figure 1.1). For both the park and the reserve, Black spruce is the dominant tree species with Balsam fir (*Abies balsamea*, (L.)) being the co-dominant species. The ground vegetation is dominated by dwarf shrubs that consist of *Ledum groenlandicum* Retzius, *Salix* sp., *Vaccinium angustifolium* Ait. and *V. vitis-idea* L. and *Kalmia angustifolia* L.. The ground layer is dominated by sphagnum mosses and lichens such as *Cladonia* sp. is also common. In late spring of 1999, a 5197 hectare area of the park burned in wildfires. During the same period, several sites in the reserve were clear-cut using the Cutting with Protection of Regeneration and of Soils (CPRS) forestry practice.

Spider diversity

In the spring of 2001, 6 burned sites were selected in the park, 6 clear-cut sites in the reserve and 6 undisturbed sites (4 in the park and 2 in the reserve) for a total of 18 sites (Figure 1.1). In each of these sites a 400 m² area was delimited. Within the 400 m², three 10 cm diameter pitfall traps, 10 m apart (Figure 1.2) were inserted in the ground with the tip of the traps flush with the ground level. Each trap was composed of two plastic cups one within the other to facilitate the removal of the trap from the forest floor and minimize damage to the soil surrounding the trap during the weekly sampling. In

each trap was poured 100 ml of a 50% ethylene glycol solution. A square 10 cm by 10 cm plastic roof supported by nails in its extremities was placed 5cm above the traps. The traps were active for six weeks from June 6th, 2001 to July 17th, 2001. All traps were emptied weekly. All spiders were identified to species (Dondale and Redner 1978, Dondale and Redner 1982, Dondale and Redner 1990, Platnick and Dondale 1992, Pierre Paquin pers. comm.) and compared with specimens from the National Collection at Agriculture Canada in Ottawa. Classification followed Platnick (1993, 1997). A voucher specimen collection of the species collected during the study will be deposited at the Insectarium René-Martineau at the Centre de Foresterie des Laurentides in Quebec city. In order to account for traps damaged mostly due to mammal predation during the sampling period, I converted abundance and richness values to values per trap per week.

I calculated alpha, beta and gamma diversity of the spider assemblages. Alpha diversity was the mean spider richness per site across the three traps, beta diversity was the species turnover between sites within a treatment (burned, clear-cut or undisturbed) and gamma diversity was the overall spider richness found at the treatment level. Beta diversity was estimated following Drapeau *et al.* (2000) by calculating the gradient length occupied by the sites of a disturbance type on the first axis of a detrended correspondence analysis (DCA) since units on the axes of a DCA represent standard deviations.

Habitat structure

Five 1 m² quadrats were systematically laid out within each 400m² site (see Figure 1.2). In each quadrat, I measured the percent cover at the ground level of moss-

lichen substrate, bare soil, leaves (including needles), shrubs and woody debris (WD). Litter depth (A and O layers) and tree canopy cover were also measured. The canopy cover corresponded to the canopy created by the mature dominant trees at each site and was measured following the procedure of James and Shugart (1970) with the exception that measurements (presence of canopy or not) with the site tube were taken at every metre for a total of ten metres in each cardinal direction. These values were then converted into a percentage of canopy cover. Species richness inventories of all vascular plants were made. In addition, the number of snags, stumps and fallen trees were counted for each 400 m² area. I also calculated a forest floor heterogeneity index for each site using plot-scale environmental variables (1m² quadrat measurements) that were directly related to ground level structure characteristics associated with each site. First, I calculated the coefficient of variation across the five plots within the site for each plot-scale environmental variables. The heterogeneity index for each site was the average of these coefficients of variation. Sites that were more heterogeneous at the local scale had larger values of the index.

Statistical Analyses

I performed one-way ANOVA's to compare the environmental characteristics of the three treatments, burned, clear-cut and undisturbed. Alpha diversity and abundance (# of individuals/trap/week) of spiders were also compared between treatments using one-way ANOVA's. We conducted a correspondence analysis (CA) (Jongman et al. 1995, Legendre and Legendre 1998) to assess the variation in species composition of spider assemblages both within and between the three treatments. The data structure of spider

assemblages was then related to the environmental variables using a canonical correspondence analysis (CCA) (ter Braak 1986, Legendre and Legendre 1998, McGarigal *et al.* 2000) using CANOCO 4.0 for Windows[®] (ter Braak and Smilauer, 1998). For the CCA, the environmental variables measured at the plot scale were averaged to obtain a value for each site. These included percent cover at the ground level of leaves, mosses and lichen, bare soil and coarse woody debris, as well as the percent cover of the shrub layer and the tree canopy. Variables measured at the site level - number of stumps, treefall, snags and our calculated heterogeneity index - were also included in the CCA. All the values obtained for the environmental variables were standardised. The spider matrix used for the CCA was composed of all the species represented by more than one individual.

Finally, to determine which environmental variables were related to the presence of the most abundant spider species from the hunter and the web-builder guilds, a CCA was performed with the 10 most abundant spiders from each guild as the response variables and the environmental variables as the predictors. The hunter guild of spider families (Agelenidae, Amaurobiidae, Clubionidae, Gnaphosidae, Lycosidae, Philodromidae, Salticidae, Thomisidae) hunt their prey or use a sit-and-wait ambush technique and are generally associated with open, well-lit habitats that have a more variable climate. The web-builders (families Araneidae, Dictynidae, Hahniidae, Linyphiidae (subfamilies Linyphiinae and Erigoninae), Tetragnathidae, Theridiidae) use webs to catch their prey and are associated with moist mature closed canopy forest containing a deep moss or lichen layer on the ground, especially in the boreal ecosystems. The classification of the spider species into guilds, hunters and web-builders, has been

widely used in other studies (Huhta 1971, Uetz 1975, Uetz 1976, Uetz 1977, Coyle 1981, Pajunen et al. 1995, Collins et al. 1996, Buddle et al. 2000). Therefore, we expected these two groups to respond differently to the environmental changes caused by clear-cutting and fire disturbances.

1.3 Results

Overall 2935 individual spiders were collected, of which 2655 were adults. There were 95 species from 13 families and 59 genera. Hunting spiders were numerically dominant (2336 individuals) while web-builders were represented by 599 individuals. Linyphiids were the most diverse family represented by 50 species but they accounted only for 16.3 percent of all adult spiders. The family represented by the second largest number of species was the wolf spider (Lycosidae) family with 15 species. Lycosids were the most abundant family, with 2054 individuals (77% of the total). Six species found in the Lycosid genus *Pardosa* accounted for 72% of the total abundance. Thirty-two species of spiders were represented by only one individual in our study.

Forest floor structure and heterogeneity

There were significant differences among the disturbance categories in the structure and composition of the ground surface (Table 1.1). First, the forest floor heterogeneity index was significantly higher in the clear-cut sites than in the burned sites. The percent cover of ground vegetation was significantly higher in the burned sites than in the clear-cut sites. The percent cover of moss and lichen and the litter depth were significantly higher while the amount of coarse woody debris was significantly lower in the undisturbed and burned sites than the clear-cut sites. Finally canopy cover and depth of the moss-lichen substrate were significantly higher in the undisturbed sites (Table 1.1) than in the clear-cut and burned sites. While not significantly different, it is interesting to note that there was, on average, 13% canopy cover in the burned sites compared to 0% in the clear-cuts.

Total Abundance and Diversity Indices

Differences in the total abundance of the spider assemblages were observed between the different disturbance types. Abundance values (abundance/trap/week) obtained were significantly higher in the clear-cut sites followed by the burned sites, with the undisturbed sites showing the lowest abundances of spiders (Figure 1.3). When we separate the spider assemblages into the hunting and web-building guilds the hunting guild values for abundance/trap/week were significantly different among all the treatments, while I did not detect any significant differences in abundance between disturbances for the web-building spiders (Figure 1.3). Overall Alpha diversity, the mean spider richness per site, was not different among disturbance types. At the guild level, the alpha diversity of the hunting guild was significantly different among treatments, being the highest in the clear-cut sites followed by the burned sites, while it did not differ among treatments for the web-building guild (Table 1.2). The species turnover among sites represented by the beta diversity estimates was the smallest in the clear-cut sites (0.42) followed by the burned sites (0.78) and the highest species turnover ratio (1.04) was among the undisturbed sites (Table 1.2). The overall diversity, represented by the gamma diversity was at its lowest in the disturbance resulting from the wildfire (Table 1.2).

Results of one-way ANOVA's for the effect of the disturbance category on the 10 most abundant species of the Hunter and Web-builder's guilds are shown in Table 1.3. In the hunter guild, 8 of the 10 most abundant hunters were Lycosids and 7 of them responded significantly. Of these, 5 were species of the genus *Pardosa*. *Pardosa hyperborea* (Thorell) was significantly more abundant in the burned sites, while *Pardosa*

moesta Banks and *Pardosa uintana* (Gertsch) were significantly more abundant in the clear-cut sites. *Pardosa mackenziana* (Keyserling) and *Pardosa xerampelina* (Keyserling) were significantly more abundant in the disturbed sites compared to the control sites (Table 1.3). For the web-builders, 4 species showed a significant response to the type of disturbance. The Erigoninae *Pocadicnemis americana* Millidge was significantly more abundant in undisturbed and burned sites while another Erigoninae, *Sisicottus montanus* (Emerton), and the Hahniidae *Cryphoea montana* Emerton were significantly more abundant in the clear-cut sites. Web-building spiders *Wabasso cacuminatus* Millidge (Table 1.3) and *Walckenaeria minuta* Emerton (Appendix 1.1) were found only in burned sites. Finally, clear-cuts had 12 rare species (more than 1 individual but fewer than 10 found throughout the study) that were not found in burned stands while 8 rare species found in burned stands were not found in clear-cuts (see Appendix 1.1).

Effect of disturbances on spider assemblage species composition

Forty percent of the variation in species composition was explained by the first two axes of the correspondence analysis (CA) performed on the spider assemblage data matrix (Figure 1.4). The spider assemblages associated with each disturbance are well clustered together in different parts of the diagram and both show a different composition from the spider assemblages found at the undisturbed sites. Spider assemblages found in the undisturbed sites also show more variability in their composition (not well clustered) than those found in the disturbances.

The CCA shown in Figure 1.5 indicates which environmental variables are most associated with spider assemblages from each treatment. The first canonical axis from the CCA explained 25.3 percent of the variance between the spider assemblages and the environmental variables. It portrays a gradient of environmental variables associated with dry, open, disturbed habitats (percent cover of WD, bare soil and dead leaves) on the left side of the axis to a closed moist old growth habitat (high percentages of moss and lichens on the ground, litter depth, snags and canopy cover) on the right of the axis. The second canonical axis, explaining 19.2 percent of the variance, represents mainly a gradient of cover dominance from trees to shrubs (Figure 1.5). The environmental variables influencing the spider assemblages found in clear-cut sites were the percent of coarse woody debris, the percent of bare soil and clear-cut artefacts like number of fallen trees and stumps. In the wildfire sites, the environmental variables mostly associated to the species composition of the spider assemblage were the percent cover of the ground vegetation and litter depth (Figure 1.5). The spider assemblages in the undisturbed sites were mainly associated with old growth attributes such as percent canopy cover, and depth and percent cover of moss and lichen. Also interesting is that the spider assemblages in the undisturbed sites were much more spread apart in Figure 1.5 than were the assemblages in either of the two disturbances.

Effects of environmental variables on abundances of the 10 most abundant species of each guild are shown in Figure 1.6. The scores of the most numerous spiders of the web-building guild (stars in Figure 1.6) are related to greater tree canopy cover, depth and percentage of cover of mosses and lichens, and the ground vegetation cover. Hunters (black triangles in Figure 1.6) are mainly located in the lower part of the second CCA

axis. The ordination shows that openness of the area, the percent of exposed bare soil, woody debris, and leaves on the ground are the factors that have the largest influence, within our study sites, on the distribution and abundance of the dominant spiders from the hunter's guild. Of the dominant spiders from each guild, the hunting spiders *Haplodrassus signifera* (C.L. Koch), *Pardosa hyperborea*, *Pardosa mackenziana* and the web-builders *Neoantistea agilis* (Keyserling), *Tunagyna debilis* (Banks) and *Wabasso cacuminatus* were the most influential spiders in the ordination of the burned site (Figure 1.6). For the clear-cut sites, mainly three species had a very strong influence on the ordination of the sites; *Pardosa moesta* and *Pardosa uittana* from the hunter's guild and the web-builders *Cryphoeca Montana* and *Sisicottus montanus* (Figure 1.6). Two wolf spiders *Arctosa alpegina* (Doleschall) and *Pardosa xerampelina* were dominant in both disturbances. The ordination of the undisturbed sites was mainly dictated by web-building spiders and only one hunting spider *Trochosa terricola* Thorell (Figure 1.6).

1.4 Discussion

Spider assemblage responses to the disturbances

The correspondence analysis (Figure 1.4) clearly showed that the spider assemblages associated with each treatment were different. The main hypothesis behind this study was that the wildfire and clear-cuts disturbed the structure and heterogeneity of the forest floor in different ways, resulting in different spider assemblages at the ground level. Our CCA ordination (Figure 1.5) showed that spider assemblages associated with the different disturbance types reacted to different environmental variables. Several of the environmental variables strongly associated with the CCA ordination axes, such as litter depth and percentage cover of moss-lichen, coarse woody debris, canopy cover and ground vegetation, were significantly different between wildfire and clear-cut sites (Table 1.1). Woody debris (mainly logging residues), openness of the habitat, soil exposure and the amount of dead needles covering the ground were the main forest floor characteristics structuring the spider response in the clear-cut stands. These forest floor attributes are closely linked with the habitat requirements of hunting spiders and explain their significantly higher abundances in the clear-cuts stands (Figure 1.5). The literature describes them as opportunistic spiders that hunt on open forest floors and that rapidly colonize recently disturbed habitats (Aitchison-Bennell 1994, Uetz 1975). Other studies identify them as open habitat specialists that are generally the dominant guild in recently logged areas (Huhta 1971, Coyle 1981, Jennings *et al.* 1988, Pajunen *et al.* 1995, Buddle *et al.* 2000).

I noted in our clear-cut sites small undisturbed patches of forest floor that contained characteristic attributes of undisturbed forest. Nguyen-Xuan et al. (2000) also mentioned that cuts protecting the regeneration and soils (CPRS) leave behind many patches of undisturbed forest floor to facilitate natural regeneration. These patches can contain a few spider species that are associated with mature undisturbed stands and this could explain why clear-cuts and control stands yielded the highest gamma diversity values (Table 1.2). This phenomenon was also mentioned by Huhta (1971) who noted that spider assemblages in recently clear-cut forest stands still show some characteristics of old-growth forest spider assemblages due to the persistence of some mature forest species surviving in remnants of mature forest micro-habitats found in the recent clear-cuts. In recently harvested sites in the boreal mixed woods of western Canada, Buddle et al. (2000) found a large increase in linyphiid abundance, which are generally associated with mature forest stands (Huhta 1971, Coyle 1981, Pajunen et al. 1995). They attributed this in part to the possibility that these web-builders were able to survive by hiding deep in the litter. In the same area of Canada, Spence et al. (1996) found that ground beetle (Carabidae) assemblages in recently harvested sites were more similar to mature forest ground beetle assemblages than those found in older clear-cut sites.

Low species turnover among the clear-cut sites (Table 1.2) can be attributed to the fact that 6 spider species of the genus *Pardosa* of the Lycosidae family accounted for 84 percent of the abundance and were found in every clear-cut site. The recurrence and numerical dominance of these species has a major effect on the species turnover that overshadows the species of web-builders that are found in the clear-cuts. Low species

turnover among clear-cut stands could be viewed as a sign of similar type of heterogeneity of the forest floor produced by clear-cuts, although within clear-cut sites the local forest floor heterogeneity is high due to the presence of both bare ground and undisturbed patches of forest floor resulting from the CPRS silvicultural operations.

Spider assemblages from the burned sites responded to high percent cover of ground vegetation, low forest floor heterogeneity index and deep litter resulting from the addition to the litter of an abundance of fire-killed moss-lichen substrate (Figure 1.5). Although stands recently disturbed by wildfires and clear-cuts are newly created open habitats, the different forest floor characteristics explain why overall spider abundances were significantly lower in burned stands compared to clear-cuts. The lower overall gamma diversity obtained in burned stands (Table 1.2) probably arises from the less heterogeneous forest floor measured in our study (Table 1.1). This lower value of our heterogeneity index in the burned stands could also be related to the high severity (5 out of 5 on the fire severity index) of the fire event where we sampled. Interestingly though, species turnover among burned stands was twice as high as among the clear-cut sites, suggesting that the wildfire provided more environmental variability among the burned stands. A higher variation in the heterogeneity of the forest floor was also found between burned stands (0.72) compared to harvested stands (0.42) when I calculated a heterogeneity index reflecting between stand variation of forest floor heterogeneity.

Species and guild-level responses

The most numerous spider species from the two guilds showed different responses to the environmental variables (Figure 1.6). Most web-builders were associated with variables pertaining to mature forest stands like canopy cover, the moss-lichen component on the forest floor and dense ground vegetation cover which are required to attach their webs (Hutha 1971, Palmgren 1977). Hunters were associated with variables representing openness of the forest floor such as high percentages of exposed soil, dry coniferous needles and in the case of burned habitats high percentages of burned litter. These forest floor characteristics facilitate prey hunting and egg sunning for lycosid females (Coyle 1981, Buddle 2000). Their morphology (Coyle, 1981) and bigger body size (Larivée *et al.*, in prep.) have been suggested as adaptations to endure the larger temperature variations that occur in open habitats.

Many spider species were found only or mostly in logged or burned sites (see Table 1.3 and Appendix 1.1). These differences in the species composition of the spider assemblages found at the ground level in the two disturbances most likely arise from the structural differences of the forest floor (Table 1.1). Hunting spiders *Pardosa hyperborea*, *Alopecosa aculeata* (Clerck) and web-building spiders *Wabasso cacuminatus*, and *Walckenaeria minuta* that were found mainly or only in burned stands are signs that the wildfire offered forest floor characteristics that are not provided by the clear-cut stands like a denser shrub cover, abundant desiccated moss-lichen substrate still covering the ground and a thicker litter layer. Interestingly, Buddle *et al.* (2000) reported significantly higher captures of *P. hyperborea* in recently burned Trembling Aspen stands

of western Canada. Buddle *et al.* (2000) and Koponen (1993) also suggest that wolf spider species *Arctosa alpegina* Doleschall and *Pirata bryantae* Kurata as being aided by conditions prevailing after a wildfire. These two species were found in similar numbers in harvested stands and burned stands (See Annexe 1). As spiders are all predators and their presence in a particular habitat rely mostly on the structure and the abiotic factors found in that habitat, any interpretation concerning the dependence to fire by certain spider species should rather be viewed as a response to certain post-disturbance characteristics offered by burned habitats that the current harvesting approach do not provide.

Although most spider species responded in accordance with the general habitat specifications found in the literature for their guilds, *Trochosa terricola*, a hunting spider, was found mainly in undisturbed forests and web-building spiders *Cryphoeca montana* and *Neoantistea agilis* in the family Hahniidae were found mostly in open habitats and showed significant habitat associations (Table 1.3 and Figure 1.6) that portrayed habitat preferences contrary to what is generally recognized for their guilds. This indicates that generalizations about habitat affinities of spider guilds, while informative, do not hold for all species in the guilds.

1.5 Conclusion

I found that forest stands that were recently clear-cut using methods that protect the soil did not yield ground level spider assemblages similar to those found in recently burned stands. These results are in accordance with other studies that have found differences in the effects of wildfires and wood harvesting on biotic and abiotic components of boreal ecosystems (Johnston and Elliott 1996, Wei and Kimmins 1998, Hobson and Schieck 1999, Imbeau *et al.* 1999, Carignan *et al.* 2000, Nguyen-Xuan *et al.* 2000, Schieck and Hobson 2000, Simons *et al.* 2002). I suggest that controlled prescribed burning in clear-cut stands could reduce the differences found in the forest floor of burned and clear-cut stands by triggering the response of fire adapted shrubs and reducing the number of undisturbed forest floor patches left behind following the CPRS cutting approach used in Black Spruce boreal forests. Another outcome of this study was the higher spider species turnover between burned stands which suggests a varying type of heterogeneity occurs between burned stands compared to the heterogeneity found in clear-cut stands. Therefore, a diversification of harvesting techniques to enhance differences in forest floor heterogeneity between harvesting blocks could provide the variation in heterogeneity found between burned stands and increase the spider species turnover between harvested stands.

Table 1.1: Mean values and standard error of the environmental variables measured at each site. One-way ANOVAs of the forest floor environmental variables measured at all sites of the different treatments. Post-Hoc Scheffe's Tests were conducted for the environmental variables that were significantly different between treatments where different letters indicates significant differences between treatments. Levels of significance are as follow: * <0.05; ** <0.01; *** <0.001

Environmental Variables	Undisturbed	Wildfire	Clear-cut	Sign.
Moss depth (cm)	5.9 ± 1.6 b	0 ± 0 a	0.2 ± 0.2 a	***
Litter depth A+O layers (cm)	10.0 ± 1.4 a	13.8 ± 0.5 a	8.9 ± 1.0 b	**
Soil (% cover/m ²)	3.2 ± 1.2	14.8 ± 7.0	14.7 ± 4.26	ns
Leaves (% cover/m ²)	8.1 ± 1.3	10.2 ± 3.2	13.8 ± 2.7	ns
Moss-Lichen (% cover/m ²)	75.8 ± 3.0 a	57.7 ± 9.8 a	21.2 ± 5.5 b	***
Woody Debris (% cover/m ²)	11.2 ± 2.0 a	13.5 ± 4.0 a	50.5 ± 4.2 b	***
Ground veg (% cover/m ²)	30.0 ± 10.0 ab	39.8 ± 4.1 a	9.3 ± 3.7 b	**
Canopy cover (% cover/m ²)	55.8 ± 9.3 a	13.3 ± 3.7 b	0 ± 0 b	***
Heterogeneity Index	0.53 ± 0.10 ab	0.41 ± 0.15 a	0.68 ± 0.18 b	*
Plant richness (m ²)	7.2 ± 0.9	4.2 ± 0.9	6.3 ± 1.0	ns
Snag (nb/400m ²)	6.7 ± 3.7	2.3 ± 1.0	1.7 ± 0.6	ns
Stump (nb/400m ²)	9.5 ± 3.2	24.2 ± 17.4	41.2 ± 2.9	ns
Treefall (nb/400m ²)	23.2 ± 6.9	31.8 ± 11.8	49.3 ± 3.5	ns

Table 1.2: Descriptive statistics and results of One-way ANOVAs on comparisons of community structure parameters for the spider assemblages found in undisturbed, burned and clear-cut lichen-woodlands of Parc des Grands-Jardins and the Réserve fauniques des Laurentides, Québec. Treatment effects were performed on the alpha diversity values with One-way ANOVAs. Post-Hoc Scheffe's Test was conducted for the alpha diversity values obtained at each treatment where different letters indicates significant differences between treatments.

	Alpha Diversity			Beta diversity			Gamma diversity		
	Hunters	Web-builders	Total	Hunters	Web-builders	Total	Hunters	Web-builders	Total
Undisturbed	1.4 ± 0.31 a	2.47 ± 0.89 a	3.87 ± 1.03 a	2.17	2.03	1.04	18	39	57
Burned	1.93 ± 0.30 b	2.1 ± 0.68 a	4.03 ± 0.37 a	0.71	1.05	0.78	21	28	49
Clear-cut	2.5 ± 0.28 c	2.27 ± 0.48 a	4.77 ± 0.37 a	0.36	1.13	0.42	28	31	59

Table 1.3: Total abundance of the 10 most abundant spider species of each guild. One-way ANOVAs testing for disturbance effects were performed on abundance (nb.ind/trap/week) values. Post-Hoc Scheffe's Test was conducted for spiders who responded significantly to the treatment effect. Values with different letters indicate significant differences between treatments.

	Species	Undisturbed	Burned	Clear-cutted	Total	Sign.
Hunters	<i>Cybaeopsis tibialis</i>	20	2	13	35	ns
	<i>Haplodrassus signifer</i>	2	4	6	12	ns
	<i>Alopecosa aculeata</i>	13	33	10	56	ns
	<i>Arctosa alpegina</i>	0	10	14	24	*
	<i>Pardosa hyperborea</i>	76a	257b	84a	417	*
	<i>P. mackenziana</i>	1b	72a	40a	113	***
	<i>P. moesta</i>	1a	8a	233b	242	***
	<i>P. uintana</i>	46a	170a	653b	869	***
	<i>P. xerampelina</i>	1b	119a	152a	272	**
	<i>Trochosa terricola</i>	20a	6b	11ab	37	*
Web-builders	<i>Cryphoea montana</i>	2a	0a	30b	32	***
	<i>Neoantistea agilis</i>	0b	19a	8ab	27	**
	<i>Diplocentria bidentata</i>	21	17	33	71	ns
	<i>D. rectangulata</i>	16	3	4	23	ns
	<i>Pocadicnemis americana</i>	19	24	6	49	*
	<i>Sisicottus montanus</i>	21ab	1b	29a	51	*
	<i>Tunagyna debilis</i>	10	13	7	30	ns
	<i>Wabasso cacuminatus</i>	3	14	0	17	ns
	<i>Walckenaeria directa</i>	12	3	9	24	ns
	<i>W. tricornis</i>	9	13	5	27	ns

Figure 1.1: Location of the sampling sites in the Grands-Jardins National Park and the Réserve faunique des Laurentides in Québec, Canada.

Figure 1.2: Environmental variable sampling design and pitfall trap layout in the 400m² area used at each site.

Figure 1.3: Spider abundance values (catches/trap/week) and standard errors obtained in undisturbed, burned and clear-cut lichen-woodlands of Parc des Grands-Jardins and the Réserve fauniques des Laurentides, Québec. One-way ANOVAs testing for treatment effects were performed on the abundance values at the guild level. Post-Hoc Scheffe's Test was conducted for the Hunter's guild values obtained at each treatment where different letters indicates significant differences between treatments.

Figure 1.4: Ordination of the 18 sampling sites obtained from a correspondence analysis (CA) of the spider assemblages found at each site. Values aside the canonical axes are the percentages of explained variance by the canonical axis.

Figure 1.5: Ordination of the 18 sites obtained from a canonical correspondence analysis (CCA) according to the spider assemblages and environmental variables measured at the sites. The percentages of explained variance by the canonical axis are in brackets.

Figure 1.6: Canonical correspondence analysis (CCA) of the 10 most abundant species from the hunter guild (black triangles) and the web-builder guild (stars). The arrows represent the environmental variables while their length indicates the importance of their effect on the spider species. Aloacu = *Alopecosa aculeata*, arcalp = *Arctosa alpegina*, crymon = *Cryphoea montana*, cybtib = *Cybaeopsis tibialis*, dipbid = *Diplocentria bidentata*, diprec = *Diplocentria rectangulata*, hapsig = *Haplodrassus signifer*, noeagi = *Neoantistea agilis*, parhyp = *Pardosa hyperborea*, parmac = *Pardosa mackenziana*, parmo = *Pardosa moesta*, paruin = *Pardosa uintana*, parxer = *Pardosa xerampelina*, pocame = *Pocadicnemis americana*, sismon = *Sisicottus montanus*, troter = *Trochosa terricola*, tundeb = *Tunagyna debilis*, wabcac = *Wabasso cacuminatus*, waldir = *Walkenaeria directa*, waltri = *Walckenaeria tricornis*.

Appendix 1.1: List of all the spider species collected during the study (Chapter One) with their total abundance in each treatment. One-way ANOVAs testing for disturbance effects were performed on species with total abundance values of 20 individuals or more. Values used for the ANOVAs were converted to nb.ind/trap/week values. Post-Hoc Scheffe's Test was conducted for spiders who responded significantly to the treatment effect. Values with different letters indicate significant differences between treatments.

Family	Species	Undisturbed	Burned	Clear-cutted	Total	
Amaurobiidae	<i>Callobius bennetti</i> (Black.)	5	1	1	7	
	<i>Cybaeopsis tibialis</i> (Em.)	20	2	13	35	ns
	2	25	3	14	42	
Clubionidae	<i>Clubiona canadensis</i> (Em.)	0	0	1	1	
	<i>C. kulczinskii</i> Lessert	1	1	0	2	
	2	1	1	1	3	
Dictynidae	<i>Cicurina brevis</i> (Em.)	0	0	1	1	
Gnaphosidae	<i>Drassodes neglectus</i> (Keys.)	0	2	0	2	
	<i>Gnaphosa muscorum</i> (C.L.K.)	0	5	2	7	
	<i>G. microps</i> (Holm)	0	0	2	2	
	<i>Haplodrassus eunis</i> Cham.	0	1	0	1	
	<i>H. hiemalis</i> Em.	0	0	1	1	
	<i>H. signifer</i> (C.L.K.)	2	4	6	12	ns
	<i>Micaria aenea</i> Thor.	2	3	0	5	
	<i>M. pulicaria</i> (Sund.)	0	1	2	3	
	<i>Zelotes fratris</i> Cham.	0	0	1	1	
	9	4	16	14	34	
Hahniidae	<i>Cryphoea montana</i> Thor.	2a	0a	30b	32	***
	<i>Hahnina cinera</i> Em.	0	1	0	1	
	<i>Neoantistea agilis</i> (Keys.)	0b	19a	8ab	27	**
	3	2	20	38	60	
Linyphiidae	<i>Agyneta allosubtilis</i> Loksa	0	0	3	3	
	<i>A. olivacea</i> (Em.)	0	0	1	1	
	<i>A. simplex</i> (Em.)	4	0	0	4	
	<i>Aphileta misera</i> (O.P.C)	0	0	2	2	
	<i>Bathypantes pallidus</i> (B.)	1	0	5	6	
	<i>Carorita limnaea</i> (C.&B.)	3	1	0	4	
	<i>Centromerus furcatus</i> (Em)	6	0	2	8	
	<i>C. longibulbus</i> (Em.)	4	2	0	6	
	<i>Ceratinella brunnea</i> Em.	3	3	5	11	
	<i>Ceratinops</i> sp#1	1	0	0	1	
	<i>Cnephalocotes obscurus</i> (Black.)	0	2	0	2	
	<i>Diplocentria bidentata</i> (Em)	21	17	33	71	ns
	<i>D. rectangulata</i> (Em.)	16	3	4	23	ns
	CNC Species #64	2	1	1	4	
	<i>Gonatium crassipalpum</i> (Bryant)	0	0	1	1	
	<i>Halorates plumosus</i> (Em.)	0	0	1	1	
	<i>Hilaira herniosa</i> (Thor.)	1	0	0	1	

	<i>Hybauchenidium gibbosum</i> (Sor.)	1	0	0	1	
	<i>Improphantes complicatus</i> (Em.)	5	1	0	6	
	<i>Leptyphantes alpinus</i> (Em.)	2	0	6	8	
	<i>Leptyphantes</i> #7 CNC	1	0	0	1	
	<i>Maro amplus</i> (D.&B.)	3	0	0	3	
	<i>Micrargus longitarsus</i> (Em.)	0	0	1	1	
	<i>Oreonetides vaginatus</i> (Thor.)	1	0	3	4	
	<i>Oreophantes recurvatus</i> (Em.)	0	0	1	1	
	<i>Pityohyphantes costatus</i> (Hentz.)	1	0	0	1	
	<i>Pocadicnemis americana</i> (Mill.)	19	24	6	49	*
	<i>Sciastes truncatus</i> (Em.)	5	1	6	12	
	<i>Scironis tarsalis</i> (Em.)	2	0	0	2	
	<i>Scotynotilus pallidus</i> (Em.)	0	1	0	1	
	<i>S. sacer</i> (Crosby)	1	0	1	2	
	<i>Sisicottus montanus</i> (Em.)	21ab	1b	29a	51	*
	<i>Sisis rotundus</i> (Em.)	0	0	1	1	
	<i>Styloctetor stativus</i> (Sim.)	2	0	0	2	
	<i>Tapinocyba bicarinata</i> (Em.)	1	2	0	3	
	<i>T. minuta</i> (Em.)	0	0	2	2	
	<i>T. simplex</i> (Em.)	1	1	0	2	
	<i>Tunagyna debilis</i> (B.)	10	13	7	30	ns
	<i>Wabasso cacuminatus</i> Mill.	3	14	0	17	ns
	<i>Walckenaeria arctica</i> Mill.	2	2	0	4	
	<i>W. atrotibialis</i> (O.P.C)	1	1	0	2	
	<i>W. castanea</i> (Em.)	0	3	0	3	
	<i>W. clavipalpis</i> Mill.	0	0	1	1	
	<i>W. directa</i> (O.P.C)	12	3	9	24	ns
	<i>W. fallax</i> Mill	1	0	0	1	
	<i>W. karpinskii</i> (O.P.C)	0	0	1	1	
	<i>W. minuta</i> (Em.)	1	14	0	15	
	<i>W. tricornis</i> (Em.)	9	13	5	27	ns
	<i>Wubana pacifica</i> (B.)	1	0	0	1	
	<i>Zornella cultrigera</i> (L. K.)	1	1	2	4	
	50	169	124	139	432	
Liocranidae	<i>Agroeca ornata</i> Banks	1	2	0	3	
Lycosidae	<i>Alopecosa aculeata</i> (Cl.)	13	33	10	56	ns
	<i>Arctosa alpegina</i> (Dole.)	0	10	14	24	*
	<i>A. raptor</i> (Kulc.)	0	0	5	5	
	<i>Pardosa concinna</i> (Thor.)	0	0	5	5	
	<i>P. fuscula</i> (Thor.)	0	0	6	6	
	<i>P. hyperborea</i> (Thor.)	76a	257b	84a	417	*
	<i>P. mackenziana</i> (Keys.)	1b	72a	40a	113	***
	<i>P. moesta</i> Banks	1a	8a	233b	242	***
	<i>P. uintana</i> (Gerstch)	46a	170a	653b	869	***
	<i>P. xerampelina</i> (Keys.)	1b	119a	152a	272	**

	<i>Pirata bryantae</i> (Kurata)	2	1	1	4
	<i>P. minutus</i> Em.	0	0	1	1
	<i>P. montanus</i> Em.	0	0	1	1
	<i>P. piraticus</i> (Cl.)	0	0	2	2
	<i>Trochosa terricola</i> Thor.	20a	6b	11ab	37 *
	15	160	676	1218	2054
Philodromidae	<i>Philodromus placidus</i> B.	1	0	0	1
Salticidae	<i>Pellenes montanus</i> (Em.)	0	0	1	1
Tetragnathidae	<i>Tetragnatha viridis</i> (Walc.)	1	0	0	1
Theridiidae	<i>Enoplognatha intrepida</i> (Sor.)	0	1	0	1
	<i>Euryopsis argentea</i> Em.	3	0	0	3
	<i>Robertus fuscus</i> (Em.)	2	1	1	4
	<i>Theonoe stridula</i> Cr.	0	1	0	1
	4	7	3	2	12
Thomisidae	<i>Xysticus britcheri</i> (Gerstch)	1	0	0	1
	<i>X. canadensis</i> Gertsch	3	0	4	7
	<i>X. ellipticus</i> Tumb. et al.	0	1	0	1
	<i>X. luctuosus</i> (Black.)	1	2	1	4
	<i>X. obscurus</i> Collet	0	0	1	1
	5	5	3	6	14
13	95	374	848	1433	2655

CHAPTER 2

COMPARING THE EDGE EFFECTS OF WILDFIRE AND CLEAR-CUTTING ON THE GROUND LEVEL SPIDER ASSEMBLAGES IN A BOREAL FOREST

2.1 Introduction

In the boreal forests of North America wildfires and forest harvesting are two major disturbances (Bergeron *et al.* 2001, Rees and Juday 2002). Recently, many studies have compared their effects on the abiotic (Mackay and Robinson 1987, Carignan *et al.* 2000) and biotic (Johnston and Elliott 1996, Hobson and Schieck 1999, Imbeau *et al.* 1999, Buddle *et al.* 2000, Nguyen-Xuan *et al.* 2000, Drapeau *et al.* 2002, Simon *et al.* 2002, Simons *et al.* 2002, Larivée *et al.* in prep.) components of the forest. The aim of gathering this information is to refine sustainable management efforts of the boreal forest by adjusting forestry practices toward patterns and processes that occur under the natural disturbance regime (Attiwill 1994, Angelstam 1998, Simberloff 2001).

Recently burned or harvested stands differ from the mature forest in many obvious ways, such as; openness of the habitat, light distribution, humidity, temperature, wind and overall microclimate stability (Forman 1995, deMaynadier and Hunter 1998, DeGraaf and Yamasaki 2001). The contrast between the disturbed habitats and the adjacent undisturbed habitats creates an "induced edge" (Laurance and Yensen 1991, Voller 1998). The edge is defined as the part of both habitats that is modified, biotically

and abiotically, through its interactions with the adjacent habitat (Murcia 1995, Hilmo and Holien 2001).

Edge effects on most microclimatic variables in the undisturbed habitat generally vanish within the first 60 m of the edge, with strong effects in the first 10 to 20 m (Matlack 1993, Murcia 1995, deMaynadier and Burton 1998, Davies-Colley *et al.* 2000, Burton 2002; see Chen *et al.* for up to 250 m). The undisturbed habitat can also influence the microclimate to about 20 m into the disturbed habitat (Cadenasso *et al.* 1997, Davies-Colley *et al.* 2000). The responses of biotic components to the edge are generally detected in the first 30-60 m inside the undisturbed habitat (Matlack 1993, Murcia 1995, Peltonen 1997, Esseen and Renhorn 1998, deMaynadier and Hunter 1998, Davies-Colley *et al.* 2000, Degraaf and Yamasaki 2001, Harper and Macdonald 2001, 2002, Hilmo and Holien 2002, Rheault *et al.* 2003), and they can reach 20 m into the disturbed areas (Meiners *et al.* 2002).

It is generally thought that clear-cuts create hard or abrupt edges while wildfires create softer edges (Laurance 1991, Forman 1995, Voller 1998). A hard edge refers to an abrupt transition between the two habitats with very limited penetration of edge effect into the adjacent habitats, while a soft edge is more permeable to edge effects, which penetrate farther into the adjacent habitats (Stamps *et al.* 1987, Forman 1995, Voller 1998).

To my knowledge, no empirical studies have compared edge effects resulting from wildfires and logging in the boreal forest. However, better knowledge on how organisms respond to anthropogenic versus naturally induced edges is important in the context of the increasing density of clear-cut edges in boreal landscapes. This information is needed for forestry practices that aim to approach as much as possible the spatial and temporal patterns created by the natural disturbance regime.

Many attributes of spiders (Araneae) make them a good taxon for the investigation of edge effects. They have high abundance levels (Huhta 1971, Coyle 1981, Jennings *et al.* 1988, Coddington *et al.* 1996, Buddle *et al.* 2000). Species richness levels, individual species abundances and distributions can be associated to the structural attributes and the abiotic factors found in their habitat (Uetz 1979, Hatley and Macmahon 1980). They are important components of the food chain (Atlegrim and Sjoberg 1995, Nyffeler 2000), and are known to be involved in nutrient recycling and fixation in early stages of succession in disturbed habitats (Hodkinson *et al.* 2001).

The aim of this study is to compare edge effects created by a recent wildfire and recent clear-cuts on the ground living spider assemblages in a boreal forest. I have previously shown that ground living spider assemblages in a boreal forest differ between a recent wildfire and recent clear-cuts (Larivée *et al.* in prep.). Here, I hypothesize that environmental conditions at the edges resulting from wildfires are different from those generated by clear-cuts and consequently that ground living spider assemblages respond differently to edges resulting from the two disturbances. More specifically, I make the

following predictions: (1) edges resulting from clear-cuts have a sharper change in environmental variables than those created by wildfires, (2) the transition in the species composition of spider assemblages along wildfire edges is more gradual than across clear-cut edges, (3) clear-cut edges will reduce to a greater extent than wildfire edges the penetration of open habitat spiders in the undisturbed part of the edge, (4) clear-cut edges will reduce to a greater extent than wildfire edges the recolonization of interior habitat spiders in the disturbed part of the edge (Figure 2.1).

2.2 Materials and Methods

Study area

The study was conducted during the summer of 2001 in the Park des Grands-Jardins (the "park") (latitude 47° 41' N longitude -70° 41' O) and in the Réserve Faunique des Laurentides (the "reserve"), both located in the Charlevoix region, 150 km north east of Québec City, Canada (Figure 2.2). For both the park and the reserve, Black Spruce (*Picea mariana* (Mill)) is the dominant tree species with Balsam Fir (*Abies balsamea* (L.)) being the sub-dominant species. The ground vegetation is dominated by dwarf shrubs like *Ledum groenlandicum* Retzius, *Salix* sp., *Vaccinium angustifolium* Ait. and *V. vitis-idea* L. and *Kalmia angustifolia* L.. The ground layer is dominated by sphagnum mosses and *Cladonia* sp. is also common. In late spring of 1999, a 5197 hectare area of the park burned during a highly intense wildfire. During the same period, several sites in the reserve were clear-cut using the Cut with Protection of the Regeneration and of Soils (CPRS) approach. In this method, saplings and small trees are not harvested, leaving some vegetation structure in the stands.

Sampling scheme

In the spring of 2001, twelve 100 m edge transects were established. Six transects crossed from burned to unburned areas while the 6 other transects crossed from clear-cut to undisturbed forest (Figure 2.2). The transects started 50 m inside the disturbance and finished 50 m inside the undisturbed area (Figure 2.3). There was one pitfall trap (10 cm in diameter) every 10 m for a total of 11 traps along each transect. The traps were inserted in the ground with the rim of the traps flush with the ground level. Each trap was

composed of two plastic cups one within the other to facilitate the removal of the inner cup while minimizing damage to the soil surrounding the trap. In each trap I poured 100 ml of a 50% ethylene glycol solution and a few drops of soap. A square 10 cm by 10 cm plastic roof supported by nails was placed 5 cm above the traps. The traps were active for six weeks from June 6th, 2001 to July 17th, 2001. All traps were emptied weekly. To account for traps damaged during the sampling period, abundance and richness values were converted to values per trap per week. All adult spiders and juveniles when possible were identified to species (Dondale and Redner 1978, Dondale and Redner 1982, Dondale and Redner 1990, Platnick and Dondale 1992, Paquin pers. comm.) and compared with specimens from the National Collection at Agriculture Canada in Ottawa. Classification followed Platnick (1993, 1997).

Habitat structure

1m² quadrats were placed beside each trap. In each quadrat, I measured litter depth, and the percent cover at the ground level of moss-lichen substrate, bare soil, leaves (needles), shrubs, and coarse woody debris (CWD) which included logging residues in the clear-cut transects. I also measured tree canopy cover following the site tube procedure of James and Shugart (1970), with the exception that measurements were taken every metre for ten metres in each cardinal direction. These values were then converted into a percentage canopy cover.

Statistical Analyses

To detect changes in the environmental conditions along our edge transects, a locally weighted regression smoothing procedure (LOWESS, tension = 0.5, Legendre and Legendre 1998, deMaynadier and Hunter 1998) was conducted with SYSTAT 10[®] (Wilkinson et al. 1992) for each environmental variable in the two disturbance types to help detect asymptotic responses of the variables.

To test for differences in the abruptness of the changes in our environmental variables at the wildfire and clear-cut edges a t-test was performed to compare the 12 slope coefficients (6 from burned edge transect and 6 from clear-cut edge transects) obtained through simple regressions of each variable and distance to edge for each transect.

To assess how changes in species composition of spiders assemblages were affected by edge types, I conducted two separate correspondence analyses (CA) (ter Braak and Smilauer 1998, ter Braak 1986, Legendre and Legendre 1998, McGarigal et al. 2000) for the spider data sets of the wildfire transects and for the clear-cut transects. Species occurring only once in the data set were not included in this analysis. To facilitate the interpretation of the CA biplots, Gaussian bivariate ellipses representing three distance categories from our edge transects were superposed on the ordinations using SYSTAT 10[®]. Each ellipse is centered on the sample means of the x and y variables and represents one standard deviation from the sample means (Wilkinson et al. 1992). The distance categories were *cut* or *burn* (50 m-20 m in the disturbance), *edge* (10 m in

the disturbance to 10 m in the undisturbed part) and *undisturbed* (20 m to 50 m in the undisturbed part). These categories were chosen following the results on the change in environmental conditions at the edge, a literature review of the abiotic factors response to edges and preliminary results obtained with the spider data set. Canonical correspondence analyses were then performed on each edge data set to determine the combination of environmental variables that were associated to CA ordination axes (ter Braak and Smilauer 1998).

To measure and compare the depth of edge influence (DEI) into the undisturbed and disturbed areas, we identified spider species that were open habitat specialists and another set that were interior habitat specialists. These groups were created in order to conduct our analyses on the spider that were potentially more sensitive to edge effects. Therefore, for each species we calculated the ratio of its abundance in the last three traps inside the disturbed area (30 m to 50 m) to its abundance in the last three traps inside the undisturbed area (30 m to 50 m). If the ratio was greater than or equal to 1.5 the species was classified as an open habitat specialist and if it was less than or equal to 0.5 the species was classified as an interior habitat specialist. Thus, the cut level ratios were set to select spiders that were captured 75% or more in the open or interior part of the edge. The open habitat species were determined separately for the clear-cut transects and the wildfire transects since the spider assemblages differ between wildfire areas and clear-cuts (Larivée et al. in prep.). Abundance values for each group of species (open habitat and interior habitat) were then pooled together for each trap along the transects. Species richness of each group was also calculated for each trap.

I used a locally weighted regression smoothing procedure (Wilkinson *et al.* 1992) (LOWESS, tension = 0.5, Legendre and Legendre 1998, deMaynadier and Hunter 1998) to locate and compare asymptotic responses of abundances and richness of open and interior habitat spiders to the two types of edges.

To test for differences in the abruptness of the changes in spider abundance at the wildfire and clear-cut edges a t-test was performed on the slope coefficients obtained through simple regressions of spider abundance and distance to edge for each transect. The same procedure was repeated for the richness of each group.

2.3 Results

Overall 6222 individuals totalling 124 species from 67 genera and 15 families were captured throughout the sampling period. The wildfire transect samples contained 97 spider species and 2649 individuals, while 86 species and 3573 individuals were found in the clear-cut transects. Hunting spider species from the genus *Pardosa* (Araneae: Lycosidae) were numerically dominant through out both the wildfire (1301 individuals) and the clear-cut (2072 individuals) edge transects. Web-building spiders Linyphiidae (Linyphiinae and Erigoninae) had the highest species richness in both the burned edges and the clear-cut edges with 49 species and 47 species respectively, while Lycosids (hunting spiders) had the second highest richness, with 12 species in the wildfire edges and 13 species in the clear-cut edges.

Response of forest floor attributes and vegetation across the edges

Changes in the environmental conditions generally occurred from 30 m in the disturbed area to 30 m in the undisturbed part of the transects (Figure 2.4). The only variable that did not change at the edges of the disturbances was the percentage of leaves covering the ground. The percent cover of shrubs, coarse woody debris, and of exposed soil differed in their response to clear-cut and wildfire edges. The percent cover of coarse woody debris barely changed along the wildfire edges but showed a sharp drop ($t = 5.23$, $df = 7$, $p = 0.001$) along the clear-cut edges up to 20 m inside the forest (Figure 2.4). The percent of exposed soil showed a somewhat similar response to WD but the change was only marginally more abrupt ($t = 2.24$, $df = 8$, $p = 0.054$) along the clear-cut edges. The cover of shrubs increased 20% in the first 20 metres of the interior part of the clear-cut

transects while it diminished gradually along the wildfire edges. These changes in shrub cover were marginally significant between the two types of edge ($t = 2.03$, $df = 9$, $p = 0.09$) but the responses were in directions, preventing us from any conclusions regarding differences in the abruptness of the change in shrub cover (See Figure 2.4). While we detected no difference in the abruptness of the change in canopy cover ($t = 0.95$, $df = 9$, $p = 0.36$) there was no canopy cover in the clear-cut transects until the edge where it rose sharply. Overall, edges from the clear-cuts and wildfire are less different than expected (rejection of prediction 1).

Spider assemblages' response to the edges

Spider assemblages' edge responses are shown in Figures 2.5a and 2.5b. The first axis of both CA's represents a distance gradient of the assemblages found inside the disturbances to those found in the forest interior part of the transects. Notice the concentrated small size of the confidence ellipse of the clear-cut assemblages (spiders sampled between 20 m to 50 m in the clear-cuts) which depicts low variability amongst the species composition of the spider assemblages found from one site to another (Figure 2.5a). In addition, the ellipse representing the clear-cuts assemblages is totally embedded within the edge ellipse indicating that the spider assemblages found in the clear-cut section of the transects are a subset of the assemblages found at the edges of the clear-cuts. The lack of overlap between the ellipses from the forest interior of the clear-cut transects and the other two ellipses shows that species composition of assemblages in the forest interior is distinct and more variable (larger size of the ellipse) from that of clear-cut edges (Figure 2.5a). As for the response of the spider assemblages to the wildfire

edges (Figure 2.5b), there is a high overlap between the ellipses of spider assemblages found in the wildfire and those found at the edge, whereas the overlap between the edge and the interior ellipses is smaller. Also, the position of the ellipses in the ordination plot shows a transition in the composition of the spider assemblages from burned, to edge, to the interior part of the transects. Hence, at the community level, the variation in species composition of spider assemblages to fire edges appears more gradual than in clear-cut edges (support of prediction 2).

The spider species that are the most influential in the correspondence analyses are shown in Table 2.1. Spiders driving the structure of the ordination of the clear-cut and wildfire transects are, on the left side of the first axis, all lycosid species, members of the hunting guild while those associated with the right side of the axis are mainly linyphiid spiders which are web-building spiders. The first axis of both ordinations portrays from left to right a gradient from open habitat spiders to mature interior habitat spiders. Correlations of environmental variables with the first axis of both ordinations were highest for percent canopy cover and the depth of the lichen-moss carpet. Litter depth only correlated with the first axis of the wildfire edges ordination whereas the percent cover of the lichen-moss substrate correlated only with the first axis of the clear-cut edges ordination (Table 2.2). The percent cover of coarse woody debris and of exposed soil were the strongest environmental correlates of the second axis of the clear-cut edges ordination whereas the percent cover of shrubs was the highest correlate for the second canonical axis of the wildfire edges ordination (Table 2.2).

Guild response across the edges

Seventeen spider species were categorized as open habitat spiders in both disturbances and fourteen species were categorized as interior habitat species. There was a high variability in abundance and richness patterns of both open habitat and interior forest specialists along transects of both wildfire and clear-cut edges (Figures 2.6 and 2.7). Open habitat specialists decreased in abundance from 30 m inside the disturbed area to 20 to 30 m inside the undisturbed area, for both disturbance types (Figure 2.6). One can also notice a slightly more abrupt drop in abundance of the open habitat species across the clear-cut edges than across the wildfire edges, supporting prediction 3. In addition, slope coefficients acting as surrogates for changes in abundance of open habitat spiders over distance to the edge were significantly more negative i.e., more abrupt, for the clear-cut edges than wildfire edges ($t = 2.85$, $df = 9$, $p = 0.02$). Interior specialists dropped in abundance between 30 m in the undisturbed area to 30 m inside the disturbances (Figure 2.6). There was no difference in abruptness of this change between the clear-cut and wildfire transects ($t = 0.56$, $df = 9$, $p = 0.59$; rejection of prediction 4). Open habitat species richness did not change more abruptly ($t = 0.009$, $df = 9$, $p = 0.99$) between clear-cut and wildfire transects. Similarly, the change in richness of interior habitat spiders was not significantly different ($t = 1.02$, $df = 9$, $p = 0.34$) at the two types of edges (Figure 2.7). Thus, only the abundance of open habitat spiders changed more abruptly at the clear-cut edges providing little evidence to support to the idea that clear-cut edges are less permeable to movements of open habitat species than wildfire edges (prediction 3) (Figure 2.7).

2.4 Discussion

Response of environmental variables and spiders to edges

Five out of eight environmental variables responded in a similar fashion to both types of edges with no difference in the abruptness (non-significant t-tests of their slope coefficients) of their responses. Only the percent cover of CWD and exposed soil diminished more abruptly approaching the clear-cut edges.

On the other hand, there is some evidence for a slightly more abrupt shift in the spider assemblages at clear-cut rather than fire edges, since the ellipses on the CA ordination of the clear-cut transects did not overlap (Figure 5a) while those of the CA ordination from the wildfire transects did show some overlap (Figure 5b), indicating a more gradual change in the composition of the spider assemblages along our wildfire edge transects (prediction 2). Also, from the smoothed regressions (Figures 6 and 7), only the open habitat spiders showed a significant slightly more abrupt response in their abundance to the clear-cut edges (Figures 6).

Overall, our results do not support my first prediction (Figure 1) or the literature predicting that edges created by clear-cuts are hard in comparison to edges created by fire (Forman 1995, Voller 1998). There is little or no evidence for a more abrupt shift in environmental variables at the clear-cut edges than the burn edges. Perhaps the high severity (index of 5 out of 5) of the wildfire that occurred in the park created fire edges that were more similar to clear-cut edges than wildfires with a lower severity index would.

There is, however, a small difference between the edge types in the abruptness of the response of the spider community to the edges (prediction 2). Changes in species composition of spider assemblages were indeed more gradual (higher overlap) along edge transects of wildfire than those of clear-cut edges. Hence, even though these changes were more subtle than expected, they nevertheless suggest that spiders are a sensitive taxonomic group even for very weak differences in environmental conditions between clear-cut and wildfire edges.

I did not find that some generalist spider species were more abundant or had higher richness at the edges of both disturbances as in other arthropod studies (Helle and Muona 1985, Jokimaki *et al.* 1998, Magura *et al.* 2001). Such species or groups of species were labelled as edge specialists in these studies.

Extent of edge influence

The influence of wildfire and clear-cut edges on ground living spider abundance and richness extended no more than 30 m on both sides of the edge (Figures 2.6 and 2.7). These distances are consistent with our results for the environmental variables (Figure 2.4) and with most previous work on the response of plants to edges (Matlack 1993, Peltonen 1997, Esseen and Renhorn 1998, Harper and Macdonald 2001, 2002, Hilmo and Holien 2002, Meiners *et al.* 2002, Burton 2002, Rheault *et al.* 2003) or animals (Murcia 1995; a review, deMaynadier and Hunter 1998, Degraaf and Yamasaki 2001, see Spence *et al.* 1996 for up to 80 m). Thus, the ecotone, defined as the area where spider

assemblages associated with both the disturbed and the undisturbed habitats intermix (Forman 1995), was approximately 60 metres wide for both disturbances in our study.

Although it is recognized that edge effects occur on both of the adjacent habitats (Forman 95, Baker *et al.* 2002), most studies measuring the extent of edge influences focus on the undisturbed habitat (Laurance 1991 Laurance and Yensen 1991, Matlack 1993, Chen *et al.* 1995, Cadenasso *et al.* 1997, Donovan *et al.* 1997, Whitaker *et al.* 1997, Davies and Margules 1998, deMaynadier and Hunter 1998, Peltonen 1999, Degraaf and Yamasaki 2002, Hilmo and Holien 2002). However, it has been shown the undisturbed habitat to exert an influence on the disturbed habitat by affecting abiotic factors (lower air and soil temperatures, wind, irradiance and higher moisture) of up to 20 m (Cadenasso *et al.* 1997, Davies-Colley *et al.* 2000). This could explain why a reduction in abundance of open habitat spiders occurs around 30 m from the edge and why the interior habitat spiders only reach their expected low levels in the disturbed habitats at around the same distance of 30 m (Figures 2.6 and 2.7).

2.5 Conclusion

Edges have always been a part of boreal ecosystems as recurrent wildfires that create edges between the burned habitats and the unburned parts of the habitat have been the most important disturbance in boreal ecosystems for millennia (Niemela 1999, Bergeron *et al.* 1999, 2001). My results show that both the clear-cuts and the wildfire produced edge effects on ground living spider assemblages. There was no evidence to support my first prediction that clear-cut edges are more abrupt and therefore less permeable to spider movement between the adjacent habitats (predictions 3 and 4) than are wildfire edges. However, one of the most important characteristic that distinguishes forest harvesting from wildfires is the varying severity of wildfires (Kafka *et al.* 2001, Bergeron *et al.* 2002) which can create a variety of edge types ranging from highly contrasted to quite permeable. All of the wildfire edge transects were within the same fire event which was of high severity. Therefore before extrapolating our findings to all fires, comparative studies between edges from wood harvest and wildfires of various degrees of severity must be conducted.

The 30 m distance of the depth of edge influence created by wildfire and clear-cut can serve as relevant information in the determination of minimal width of forest remnants, size of riparian or road buffer and forest strips between cut-blocks. I suggest that a minimal width of 100 m is necessary for woody buffers and forest remnants to retain forest interior spider assemblages. Evidently, this measure is only relevant to ground living spider assemblages and could possibly change when looking at the response of other taxa.

Finally, this study was conducted two years after the disturbances. Previous studies have shown that edge effects are dynamic through time and will tend to diminish as time since the creation of the edge increases (Maltack 1993, deMaynadier and Hunter 1998, Harper and McDonald 2002). Therefore, further comparisons on the edge effects at various times since creation of the edges and on edges resulting from different silvicultural practices are needed.

Table 2.1: Correspondence analysis loadings of the 10 spiders contributing the most to the structure of the first two axes for the clear-cut and wildfire edge ordinations.

a) Clear-cut edges	AX1	AX2	WEIGHT
Eigenvalues	0.5063	0.3801	
<i>Pardosa uintana</i> (Gerstch)	-0.4783	-0.4781	158.6
<i>Pardosa moesta</i> Banks	-0.6468	-0.9713	71
<i>Pardosa hyperborea</i> (Keyserling)	-0.4439	1.8514	50.32
<i>Pardosa xerampelina</i> (Keyserling)	-0.5762	-0.7033	43.77
<i>Cybaeopsis tibialis</i> (Emerton)	1.8713	0.3592	24.53
<i>Diplocentria bidentata</i> (Emerton)	1.0765	-0.1979	22.12
<i>Sisicottus montanus</i> (Emerton)	2.5466	-0.4333	20.4
<i>Pardosa mackenziana</i> (Hentz)	-0.4677	-0.4494	18.03
<i>Pocadicnemis americana</i> Millidge	-0.2764	2.4378	11.47
<i>Improphantes complicatus</i> (Emerton)	2.6092	-0.1248	10.92
b) Wildfire Edges	AX1	AX2	WEIGHT
Eigenvalues	0.4042	0.3118	
<i>Pardosa hyperborea</i> (Thorell)	-0.4227	-0.5457	112.43
<i>Pardosa uintana</i> (Gerstch)	0.3617	1.8219	64.62
<i>Pardosa xerampelina</i> (Keyserling)	-0.7728	1.006	37.3
<i>Cybaeopsis tibialis</i> (Emerton)	2.25	-0.3057	20.67
<i>Alopecosa aculeata</i> (Clerck)	-0.4408	-0.1987	20.17
<i>Pardosa mackenziana</i> (Keyserling)	-0.2221	1.0506	17.47
<i>Pocadicnemis americana</i> Millidge	-0.0038	-0.6515	16.7
<i>Pardosa moesta</i> Banks	-1.8797	-0.897	13.82
<i>Walckenaeria tricornis</i> (Emerton)	1.568	-1.0243	11.98
<i>Diplocentria bidentata</i> (Emerton)	0.8938	-0.4138	8.45

Table 2.2: Correlation of the environmental variables with the first two axes of the canonical correspondence analysis performed on the spider species matrix constrained by the environmental variables. The first canonical axis ($p < 0.005$) and all the canonical axes ($p < 0.005$) were significant through a Monte-Carlo randomization. The degree of correlation of each environmental variable is related to its contribution to the percentage of explained variance between the spider matrix and the environmental variable matrix.

	Wildfire edges		Clear-cut edges	
	AX1	AX2	AX1	AX2
Species-environment correlations	0.90	0.75	0.92	0.81
Variation explained (%) of the species-envir. var. matrices	40.6	17.8	43.5	24.5
Environmental variables				
Lichen-moss depth	0.35	-0.04	0.51	-0.62
Litter depth	0.44	0.03	0.14	-0.18
Exposed soil	0.09	0.11	-0.24	0.39
Leaves % of cover	0.20	0.35	-0.10	0.28
Lichen-moss % of cover	0.12	-0.13	0.39	-0.60
Shrubs % of cover	-0.46	-0.46	0.18	-0.37
Woody Debris	0.00	0.07	-0.41	0.45
% of Canopy cover	0.83	-0.22	0.91	0.04

Figure 2.1: Expected shift in abundance or richness in response to wildfires (a) and clear-cut (b) edges by open and interior habitat spiders.

Figure 2.2: Location of the sampling sites in the Grands-Jardins National Park and the Réserve Faunique des Laurentides in Québec, Canada.

Figure 2.3: Trap layout along a transect.

Figure 2.4: Changes in the values of the forest floor and vegetation attributes measured along the edge transects, using the LOWESS (tension = 0.5) smoothing technique. Values on the x-axis correspond to the location on the transects.

Figure 2.5: a) Ordination of the 66 traps of the six 100m transects across clear-cut edges based on a correspondence analysis (CA) of spider species composition. b) Ordination of the 66 traps of the six 100m transects across wildfire edges based on a correspondence analysis (CA) of spider species composition. 67% concentration ellipses for the distance groups identified are shown.

Figure 2.6: Distribution of the open and interior-habitat species across the clear-cut-forest and wildfire-forest edge transects. The LOWESS (tension = 0.5) smoothing techniques fits the pooled abundance (/trap/week) of each group of species across the transects. Values on the x-axis correspond to the location on the transects.

Figure 2.7: Distribution of the open and interior-habitat species richness across the clear-cut-forest and wildfire-forest edge transects. The LOWESS (tension = 0.5) smoothing techniques fits the richness (/trap/week) of each group of species across the transects. Values on the x-axis correspond to the location on the transects.

Appendix 2.1: List of all the spiders species collected during the study with the number of individuals found in each type of edge. Numbers in bold refer to the total number of species found.

Family	Species	Wildfire edges	Clear-cuts edges
Agelenidae	<i>Agelenopsis utahana</i> (Chamb. & Iv.)	0	3
	1	0	1
Amaurobiidae	<i>Callobius bennetti</i> (Blackwall)	4	12
	<i>Cybaeopsis euopla</i> (B. & C.)	1	0
	<i>Cybaeopsis tibialis</i> (Emerton)	103	133
	3	3	2
Araneidae	<i>Hypsosinga rubens</i> (Hentz)	3	0
	1	1	0
Clubionidae	<i>Clubiona canadensis</i> (Emerton)	2	2
	<i>Clubiona kulczinskii</i> Lessert	1	0
	2	2	1
Dictynidea	<i>Cicurina brevis</i> (Emerton)	3	1
	1	1	1
Gnaphosidae	<i>Gnaphosa microps</i> Holm	9	27
	<i>Gnaphosa muscorum</i> (C.L. Koch)	11	3
	<i>Haplodrassus eunis</i> Chamberlain	6	0
	<i>Haplodrassus signifer</i> (C.L. Koch)	14	11
	<i>Micaria aenae</i> Thorell	15	4
	<i>Micaria pulicrina</i> (Sundevall)	4	0
	<i>Zelotes fratris</i> Chamberlain	1	0
	7	7	4
Hahniidae	<i>Cryphoeca montana</i> Thorell	7	41
	<i>Hahn timer cinera</i> Emerton	8	5
	<i>Hahn timer glacialis</i> (Sorensen)	0	10
	<i>Neoantistea agilis</i> (Keyserling)	43	39
	<i>Neoantistea magma</i> (Keyserling)	1	1
	5	4	5
Linyphiidae	<i>Agyneta allosubtilis</i> Loksa	0	21
	<i>Agyneta olivacea</i> (Emerton)	7	24
	<i>Agyneta simplex</i> (Emerton)	11	5
	<i>Bathypantes concolor</i> (Wider)	1	0
	<i>Bathypantes pallidus</i> (Banks)	1	21
	<i>Bathypantes reprob us</i> (Kulczinski)	1	5
	<i>Bathypantes simillimus</i> (L.Koch)	1	10
	<i>Carorita limnaea</i> (C. & B.)	5	1
	<i>Centromerus furcatus</i> (Emerton)	5	8
	<i>Centromerus longibulbus</i> (Emerton)	8	1
	<i>Ceraticelus atriceps</i> (O.P.C)	1	0
	<i>Ceraticelus similis</i> (Banks)	0	1
	<i>Ceratinella alaska e</i> (Chamb. & Iv.)	0	1
	<i>Ceratinella brunnea</i> Emerton	11	14
	<i>Ceratinops crenatus</i> (Emerton)	1	0
	<i>Diplocentria bidentata</i> (Emerton)	44	124

	<i>Diplocentria rectangulata</i> (Emerton)	12	11
	<i>Eperigone trilobata</i> (Emerton)	1	2
	<i>Estrandia grandaeva</i> (Keyserling)	0	2
	<i>Floricomus plumalis</i> (Crosby)	1	0
	<i>Gonatium crassipalpus</i> (Bryant)	4	1
	<i>Hilaira herniosa</i> (Thorell)	1	6
	<i>Hybauchenidium gibbosum</i> (Sorensen)	3	0
	<i>Impropantes complicatus</i> (Emerton)	17	55
	<i>Incestophantes washingtonii</i> (Zorsch)	4	0
	<i>Leptyphantes alpinus</i> (Emerton)	9	3
	<i>Leptyphantes</i> sp#7 CNC	0	3
	Linyphiidae sp#64 CNC	0	4
	<i>Micrargus longitarsus</i> (Emerton)	2	6
	<i>Oreonetides flavescens</i> (Crosby)	3	1
	<i>Oreonetides rotundus</i> (Emerton)	1	0
	<i>Oreonetides vaginatus</i> (Thorell)	6	10
	<i>Oreophantes recurvatus</i> (Emerton)	0	2
	<i>Pelecopsis moesta</i> (Banks)	1	0
	<i>Pityohyphantes subarcticus</i> (Chamb. & Iv.)	2	0
	<i>Pocadicnemis americana</i> Millidge	94	62
	<i>Sciastes truncatus</i> (Emerton)	18	15
	<i>Scironis tarsalis</i> (Emerton)	18	3
	<i>Scotynotilus sacer</i> (Crosby)	5	8
	<i>Scotynotilus pallidus</i> (Emerton)	1	5
	<i>Sisicottus montanus</i> (Emerton)	15	90
	<i>Sisicus penusifer</i> (B. & C.)	1	0
	<i>Sisis rotundus</i> (Emerton)	0	1
	<i>Styloctetor stativus</i> (Simon)	3	1
	<i>Tapinocyba bicarinata</i> (Emerton)	11	0
	<i>Tapinocyba minuta</i> (Emerton)	2	0
	<i>Tapinocyba simplex</i> (Emerton)	1	5
	<i>Tunagyna debilis</i> (Banks)	24	24
	<i>Wabasso cacuminatus</i> Millidge	2	2
	<i>Walckenaeria arctica</i> Millidge	2	1
	<i>Walckenaeria atrotibialis</i> (O.P.C.)	0	2
	<i>Walckenaeria castanea</i> (Emerton)	6	15
	<i>Walckenaeria clavicornis</i> (Emerton)	2	1
	<i>Walckenaeria clavipalpis</i> Millidge	0	1
	<i>Walckenaeria directa</i> (O.P.C.)	34	38
	<i>Walckenaeria fallax</i> Millidge	0	1
	<i>Walckenaeria karpinski</i> (O.P.C.)	1	9
	<i>Walckenaeria minuta</i> (Emerton)	38	13
	<i>Walckenaeria palustris</i> Millidge	1	0
	<i>Walckenaeria redneri</i> Millidge	2	0
	<i>Walckenaeria tricornis</i> (Emerton)	60	61
	<i>Zornella cultrigera</i> (L.Koch)	2	10
	61	52	48
Liocranidae	<i>Agroeca ornata</i> Banks	5	3
	1	1	1
Lycosidae	<i>Alopecosa aculeata</i> (Clerck)	111	30

	<i>Arctosa alpegina</i> (Doleschall)	7	36
	<i>Pardosa distincta</i> (Blackwall)	0	2
	<i>Pardosa fuscata</i> (Thorell)	0	3
	<i>Pardosa hyperborea</i> (Thorell)	621	273
	<i>Pardosa mackenziana</i> (Keyserling)	91	105
	<i>Pardosa milvina</i> (Hentz)	0	2
	<i>Pardosa moesta</i> Banks	80	414
	<i>Pardosa uirtana</i> (Gerstch)	328	1022
	<i>Pardosa xerampelina</i> (Keyserling)	181	251
	<i>Pirata bryantae</i> (Kurata)	3	6
	<i>Pirata minutus</i> Emerton	3	1
	<i>Pirata montanus</i> Emerton	1	0
	<i>Schizocosa</i> sp. 1	1	0
	<i>Trochosa terricola</i> Thorell	41	26
	15	12	13
Philodromidae	<i>Tibellus oblongus</i> (Walckenaer)	2	0
	1	1	0
Salticidae	<i>Neon nellii</i> Peckham & Peckham	0	2
	<i>Sitticus finschii</i> (L.Koch)	1	0
	<i>Sitticus raniera</i> (Peckham & Peckham)	1	0
	3	2	1
Tetragnathidae	<i>Tetragnathidae</i> sp.	1	0
	1	1	0
Theridiidae	<i>Euryopsis argentea</i> Emerton	5	0
	<i>Robertus borealis</i> (Kaston)	2	0
	<i>Robertus fuscus</i> (Emerton)	10	7
	<i>Rugathodes sexpunctatus</i> (Emerton)	1	4
	<i>Theonoe stridula</i> Crosby	5	5
	<i>Theridion differens</i> Emerton	0	1
	6	5	4
Thomisidae	<i>Xysticus britcheri</i> (Gerstch)	1	0
	<i>Xysticus canadensis</i> Gerstch	9	9
	<i>Xysticus ellipticus</i> Turnbull et al.	7	0
	<i>Xysticus emertoni</i> Keyserling	0	1
	<i>Xysticus luctuosus</i> (Blackwall)	7	1
	<i>Xysticus obscurus</i> Collet	6	1
	6	5	4
15	Total # of spp.	124	86
	Total # of ind.	5481	3207

GENERAL CONCLUSION

Overall, my results have shown that ground living spider assemblages responded differently to the recent wildfire and clear-cuts. Firstly, a correspondence analysis showed that the composition of the spider assemblages were distinct in a recently burned habitat from those associated with clear-cut habitats. Clear-cut stands had a higher alpha diversity of hunting spiders at the stand level than wildfire stands but spider assemblages found in the clear-cut stands had the lowest species turnover between stands, reflecting a more similar type of heterogeneity at the forest floor level in the clear-cut stands. These differences in the response of the spider assemblages to the two disturbances were mainly driven by different forest floor attributes dominating each disturbance. Burned stands were characterized with high percent cover of shrubs and of moss-lichen substrate remaining after the fire and a thicker litter. Clear-cut stands had a more complex forest floor and high levels of coarse woody debris. Also, certain spider species (e.g.: *Pardosa hyperborea*, *Pocadicnemis americana*, *Wabasso cacuminatus*) were found only or more abundantly in the burned stands implying that recently burned stands harbour certain forest floor attributes that are unique to recently burned habitats.

Contrary to my prediction there was no difference in the abruptness of changes in environmental conditions of the edges created by the two disturbances. Only woody debris had a significantly more abrupt diminution (higher slope coefficients in simple regressions of percent cover in relation to distance from the edge) at clear-cut edges. Nevertheless, the composition of the ground living spider assemblages did change slightly more gradually along the wildfire edges than it did in clear-cut edges, suggesting

that this taxonomic group may be sensitive to subtle changes in the environmental conditions of edges induced by natural and anthropogenic disturbances. Hence, while my results do not allow one to conclude that clear-cut edges are more abrupt, it would be premature to conclude that edges from the two disturbances have similar effects on the ground living spider assemblages. The fire in this study was severe, which may have resulted in abrupt edges, in comparison with other fires of lower severity (Bergeron *et al.* 2002).

Edges induced by the wildfire and the clear-cuts created changes in the environmental conditions and the composition of the ground living spider assemblages on both sides of the edges at distances up to 30 m. Open habitat spiders penetrated up to 30 m inside the undisturbed part of the edges while the interior habitat spiders reached their richness and abundance levels 30 m inside both disturbances.

Black Spruce forests are dominated by severe fires (Bergeron *et al.* 1999) which kill the established regeneration and the organic layers of the soil, producing a stand initiation dominated by fire adapted dwarf shrubs. Stands submitted to Cuts that Protect the established Regeneration and the Soil (CPRS), the present logging strategy utilized in Black Spruce forests of Ontario and Quebec differed, in my study, from the burned stands by having a thinner litter and moss-lichen cover, a smaller percentage of shrub cover and by having a large amount of coarse woody debris as a consequence of logging residues. Therefore, I suggest that controlled prescribed burning of the forest floor in recently harvested Black Spruce stands could increase similarities with burned stands in their

forest floor attributes and consequently their spider assemblages. This would increase the chance that recently harvested stands provide similar habitats to those occurring after a wildfire which would allow the natural succession characteristic of boreal ecosystems dominated by Black Spruce.

Lastly, my findings on the depth of edge influence allow me to recommend that buffers and stands remaining after forest harvesting should have a minimal width of 100 m to prevent these habitats from being barren of ground living spider assemblages associated with the forest interior and also to provide those spider assemblages with sufficient habitat for their survival. Of course, this measure is only from the stand point of spider assemblages; it is subject to change when addressing the depth of edge influence on other organisms. See Rheault *et al.* (2003) for edge effects on epiphytic lichens in Black Spruce feather moss forests. Therefore, sustainable management efforts of the boreal forests must take into account the various possible responses of the different organisms living in it.

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