Floristic structure, composition and depth of edge influence at buffer and wildfire edges along small boreal streams

by

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Abstract

Within the boreal forest of northwestern Ontario clearcutting and fire are the most common edge-creating disturbances. Along streams, wild fire consumes the majority of trees to the edge of riparian zones. However, forest management relies on the retention of 30 to 70 m treed buffer reserves beside streams following harvesting. Fundamental knowledge regarding how buffer and fire edges compare near shoreline forests is lacking. The development of sustainable forest management strategies that emulate natural disturbances relies upon such information. The objective of this thesis was to examine and compare the edge influences on overstory structure, near ground microclimate and understory species composition at buffer and wild fire edges adjacent to small streams in northwestern Ontario. The canopy and understory conditions were sampled along (64 m) transects which started at the high water mark of the streams. The depth of edge influence (DEI) was determined by comparing values at different distances from the created edge to values in interior reference forest using repeated measures analysis of variance. Edge orientation had no significant effect at both buffer and fire edges. Fire edges had 33% lower live tree densities and 32% lower canopy cover with three times higher rates of mortality than buffer edges. Understory species compositional changes at the fire edge were lower than buffer edges mainly due to higher abundances of riparian species which are highly adapted to disturbance. The DEI for species composition of buffer edges extended up to 20 m and was greatly reduced within 5 m of fire edges mainly due to the moderating effects of streams on the microclimate. Although species compositions of the edges were different, the magnitude of edge influence (MEI) was the same indicating that the dominant species at the edges were resistant to both types of disturbance.

Acknowledgements

Table of Contents

List of Tables

List of Figures

List of Appendices

Introduction

Natural landscapes are composed of a patchwork of habitats that vary in size, shape and location. Wherever two or more habitat types abut, they form an edge or ecotone, defined by ecological gradients created by the transition from one fairly homogeneous habitat to another. Human manipulation of terrestrial ecosystems has dramatically increased the abundance and altered the nature of these ecotones as well as created new edge effects (Sisk, 2007). The term edge effect refers to the changes in physical and biological conditions that occur at an ecosystem boundary and within adjacent ecosystems (Wilcove, 1986). As a consequence of combined contributions of anthropogenic and natural disturbances (clarcutting and wildfires) a large portion of the landscape may be experiencing edge effects.

The ecological effects of edges have been studied for many decades. Clements (1907) identified the importance of edges by introducing the term 'ecotone'. Decades later, Leopold's (1933) observation that edges increased species diversity and abundance lead many wildlife managers to create more edge habitat, only to realize that the edges also have many negative effects (Bayne & Hobson, 1997). As the amount of habitat fragmentation has increased due to land management such as forest harvesting, road building and other developments, the amount of edge has increased exponentially and the negative effects of edges on biodiversity have become well recognized (Murcia, 1995; Robinson et al., 1995).

In a major review of edge effects in forest landscapes, Harper et al. (2005) classified edge effects as either "primary response that arise directly from edge creation or secondary responses that arise indirectly as a result of edge creation". Primary responses include damage to canopy vegetation, disruption of forest floor and soil, altered nutrient cycling and decomposition,

changed evaporation, and altered pollen and seed dispersal. Secondary responses (or indirect responses) include patterns of plant growth, regeneration and mortality, and are manifested as altered patterns of vegetation structure and species composition. Edge influences can also be classified as abiotic or biotic. Abiotic edge effects refer to microclimatic changes such as increased temperature or light (Chen et al., 1991); or decreased humidity (Stewart and Mallik, 2006), as well as changes in nutrient cycling (Sizer et al., 2000; Weathers et al., 2001; Elemans, 2004). Biotic edge effects include the biological factors that affect ecological communities across a boundary, such as diseases (Hannson, 1994; Benitez-Malvido, 2004; Chapman et al., 2006), alien plant species and weed invasions (Stamps et al., 1987; Cadenasso and Pickett, 2001; Pauchard and Alaback, 2006) and predators (King et al., 1998; Dijak and Thompson, 2000; Sisk and Battin, 2002) Changes in plant communities are well documented at edges, which can significantly affect reproduction (Chen, 1995), growth (Honnay et al., 2002a), seed dispersal (Janzen 1983; Fletcher et al. 2007), soil seed banks (Lin & Cao, 2009) and mortality (Brothers and Springran, 1992). Thus, edge effects can significantly affect the distribution and abundance of species that inhabit vegetation remnants (Fletcher, 2005).

The degree of influence of a given edge has on interior forest conditions are largely determined by the edge type (e.g. maintained agricultural edges versus regenerating edges after forest harvesting) as well as geographical location. Harper et al. (2005) stated that a lower edge influence can be expected in boreal and sub-boreal forest edges compared with tropical forest edges. Edge influence tends to be lower in boreal ecosystems because they are subjected to frequent natural fires, and therefore a large component of the species present may exhibit strong adaption to disturbance and consequently a lower response to edge creation than of other forests (Johnson, 1992). Furthermore, the longer optical path of the sun may reduce contrast with

2

interior forest conditions, resulting in lower magnitude and depth of edge influence at high-latitude edges (Harper et al., 2005).

Riparian buffers are areas beside streams and lakes, left undisturbed to protect sensitive areas from the edge effects (Hylander et al. 2004). Much of the past and present research and management efforts have focused on the lateral properties of riparian areas, particularly their translation into buffers left after harvesting (Lee, 2004). Typically, a belt of treed vegetation is left undisturbed to mediate the negative effects of forest harvesting on aquatic communities and water quality. The width of buffer varies depending on slope and provincial jurisdictions (Mallik, 2006). The retention of riparian plant communities in the form of buffers has been recommended for controlling erosion and sedimentation (Osborne and Kovacic, 1993; Steedman and France, 2000), moderating temperature and light (Johnson and Jones; Macdonald et al., 2003; Moore et al., 2005; Wilkerson et al., 2006), inputting fine and large organic debris (France et al., 1996; Chen et al., 2005; Kreutzweiser et al., 2005), filtering and retention of nutrients (Vought et al., 1994) and maintaining near-shore vegetation (Harper and MacDonald, 2001). In addition studies have shown buffer strips are critical in maintaining invertebrate (Richardson, 2004), fish, bird, and mammalian communities (Lee et al., 2004). Also, the complex vegetation and plant species distributions within riparian corridors influence plant species diversity patterns at both local and regional scales and further reflect both anthropogenic and natural disturbances (Ward, 1998). Because of these characteristics, riparian zones are often the ecosystem level components that are most sensitive to changes within the surrounding environment; they provide early indications of environmental change (Decamps, 1993). Therefore, identifying the key underlying gradients, abiotic conditions and major soil influences on vegetation patterns is essential in formulating plans to protect riparian habitat and function.

The buffer edges left after clearcutting in the upland are subjected to typical edge influences that effect biophysical properties and ecological processes of the buffer (Harper et al. 2005). Altered vegetation structure and microclimate at the clearcut edge of forest harvesting in the upland and the natural ecotonal effects (i.e. from the riparian/upland transition) can interact to have a combined effect on the biophysical properties of riparian buffers (Stewart & Mallik 2006). Studies from Washington (Chen et al., 1995), Maine (Hagan and Whitman, 2000), Sweden (Oerlander and Lagvall, 1993), New Zealand (Davis-Colley et al., 2000), British Columbia (Spittlehouse et al., 2004), Alberta (Harper and Macdonald, 2001) and Quebec (Mascarúa-López et al. 2006) have shown that much of the change in canopy structure and microclimate takes place within about one tree height (15 to 40m) of the buffer edge.

The majority of research has been focused on anthropogenically created edges. However, in the boreal forest, edges are also created by wild fires (Harper et al., 2004). Historically in the boreal forest of Canada, wildfire has been a major stand replacing disturbance (Carleton, 2000). In managed boreal forests most of the edge is the combined effects of forest harvesting and wildfire. The extent to which clearcut edges maintain the structural, microclimatic and species compositional attributes of fire edges could be an important concern for biodiversity conservation in the boreal forest of northern Ontario, where increased forest harvesting has become the predominant agent of instantaneous forest biomass loss (Carleton, 2000).

In contrast to clearcutting, forest fires do not leave buffer zones along streams (Lamb et al., 2003). In the boreal forest, fires typically burn directly to the edge of the riparian zone leaving few if any trees intact and then either stop or jump the wet habitat (stream) barrier. Riparian zones have high soil moisture and consequently different understory vegetation, fuel loads, ratio of live-to-dead material (flammability) and fuel moisture than in the uplands (Dwire

4

and Kauffman, 2003). Also, riparian microclimates are characterized by cooler air temperatures, lower daily maximum air temperature, and higher relative humidity than that of adjacent uplands (Brososfske et al., 1997; Danehy and Kirpes, 2000). The moister, cooler conditions likely contribute to the higher moisture content of live and dead fuels and riparian soils relative to the uplands, presumably lowering the intensity, severity and frequency of fire in the riparian areas. Furthermore, the predominantly broad-leaved riparian zone vegetation is less flammable than the conifer dominated upland vegetation (Johnson, 1992). Lamb et al. (2003) suggested that the high temperatures could damage the foliage and upper stems of some shrubs, but it is unlikely to be severe enough to dame the rhizome systems of riparian plants due to the wet soils.

Forest management in northwestern Ontario has resulted in many streams having a variable width buffer zone with riparian and upland vegetation adjacent to streams (Lamb et al., 2003). Some authors have argued that buffers are artificial anthropogenic structures with a much different functionality than the remnants left by wildfire. In Ontario, the policy of natural disturbance pattern emulation has lead to the development of the Forest Management Guide for Natural Disturbance Pattern Emulation (OMNR, 2001). With increased forest harvesting in the future, the abundance of edge influences and altered ecotonal properties are likely to increase. As a result of this increasing interface and a shift to ecosystem management with the aim of emulating natural disturbance, it is imperative to determine how fire affects the riparian communities of small headwater systems that tend to show the greatest change in abiotic and biotic structure when subjected to disturbance (Minshall et al., 1997; Nitschke, 2005). Coupling the fact that there is an incomplete knowledge of the ecological functions of headwater systems (Moore and Richardson, 2003), it has become essential to determine the resilience of these systems to clearcutting and fire. Perea and Buse (2004) stated that the body of scientific

knowledge on disturbance regimes must be increased to contribute the development management strategies that emulate natural disturbance with the greatest congruency.

Natural (ecotonal) edges have been less well studied than anthropogenically created edges. The inherent variability of processes leading to edge formation at fire edges are more gradual with larger depth of edge influence (DEI) than the sharper gradients found at clearcut edges (Harper et al., 2004). Responses of individual species to edge influence may also differ between the two edge types; for example near ground microclimate, soil moisture and nutrients after fire and clearcutting vary significantly (Nguyen-Xuan et al., 2000; McRae et al., 2001). The majority of riparian research has been concerned with aquatic components of streams, namely water quality (e.g. Carignan et al., 2000; Lamontagne et al., 2000; Steedman, 2000; Prepas et al., 2001) and far less attention has focused on the vegetation structure and composition that directly control the physical and chemical constituents of streams. Also, very few studies have compared the effects of clearcutting and fire on riparian plant communities. In a comparative study by Lamb et al. (2003) the effects of fire and clearcutting were found to have no significant difference on riparian plant communities. The structural attributes of fire and created edges along streams are not well documented. Comparing the biological legacy (canopy structure and floristic species composition) of forest harvesting and wildfire in the riparian zone and adjacent riparian buffers would help determine "resilience thresholds" of these systems (Nitschke, 2005). This understanding will in turn help achieve the goal of forest management by emulating natural disturbance.

In this study, I compared the location, overstory structure, near ground microclimate and understory species composition of edges created by wildfire and harvesting along small streams to that of undisturbed reference forest. I asked the following questions: (1) how does the

location, structure and microclimate of buffer and wildfire edges differ?; (2) what impact does the given disturbance type have on the understory species composition of these edges? and; (3) what is the detectable depth of edge influence from fire and buffer edges? I hypothesized that: (1) because of the greater structural damage at the fire edge, species composition will have a stronger MEI than buffer edges; and (2) in general, the MEI and DEI at fire edges will be weaker and shorter than buffer edges because the fire edge is located closer to streams.

Methods

Study area

The study area was located in the boreal mixedwood forest northeast of Thunder Bay, Ontario, Canada (Figure 1). This area has rolling relief with a bedrock substrate overlain by glacial tills. Temperatures for the months of January and July range from -20 to -26°C and 22 to 25°C respectively. Annual precipitation ranges between 700 and 850 mm (Baldwin et al., 2000). The forest cover in the area is dominated by conifers such as *Picea mariana, Pinus banksiana* and *Abies balsamea* as well as deciduous species such as *Populus tremuloides* and *Betula papyrifera*. The sites selected for this study had a dominant overstory vegetation of *Picea mariana* with inter-dispersed *Picea glauca, Pinus banksiana, Abies balsamea, Populus tremuloides* and *Betula papyrifera*. The riparian vegetation was typically either a swamp thicket dominated by *Alnus incana*, or a grass and sedge-dominated meadow marsh. The common riparian species include *Alnus incana, Cornus stolonifera, Calamagrostis canadensis, Thalicutrum dayscarpum, Mertensia paniculata* and *Athyrium filix-femina*. The ecotone between the riparian zone and upland forest was marked by a very rapid shift in species composition. The common upland understory species include *Ledum groenlandicum, Acer spicatum, Aster macrophyllus, Aralia nudicalus, Lycopodium annotinum* and *L. dendroideum*. Table 1 outlines site specific habitat parameters. Additional data on site conditions can be found in Appendix I.
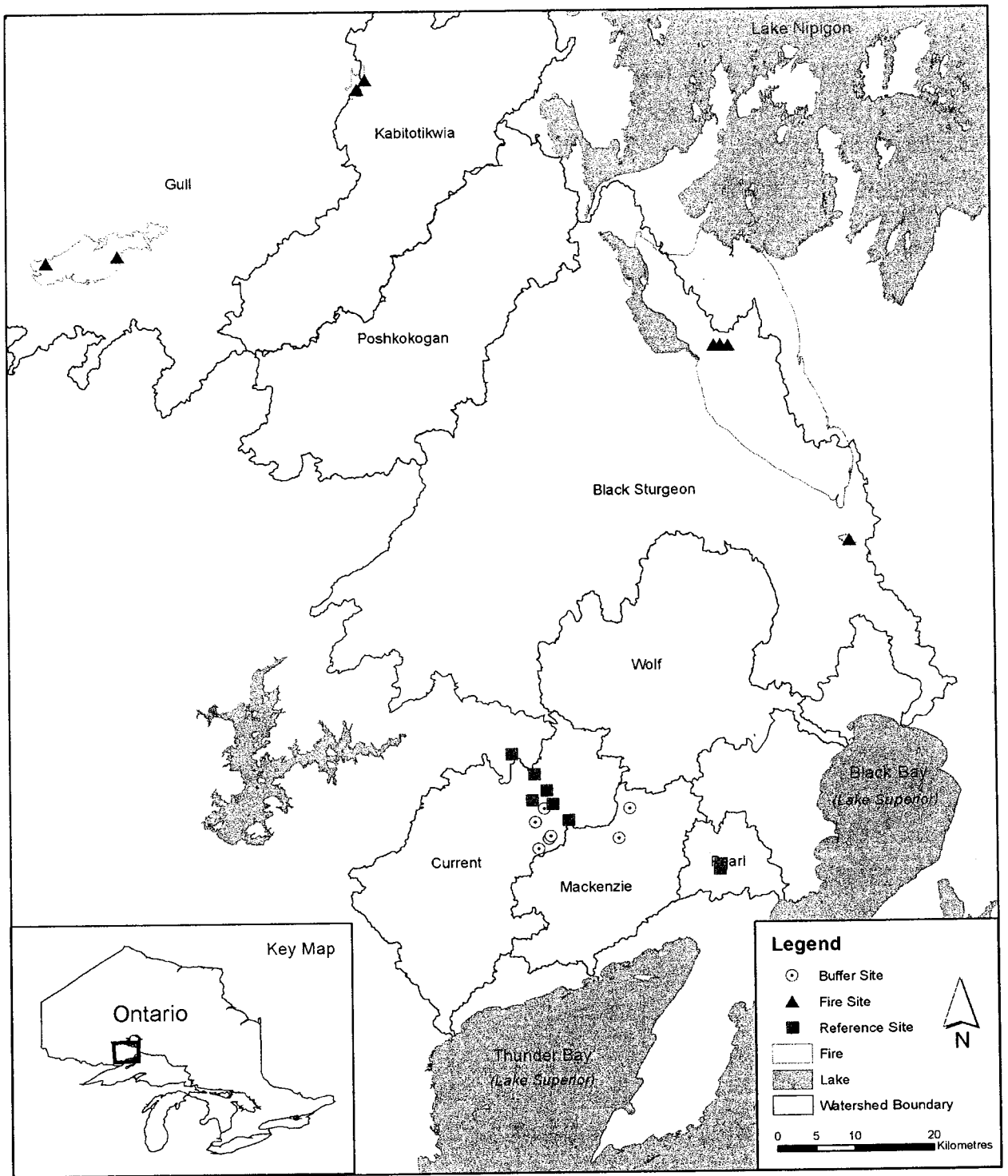
Figure 1. Map of the study area in northeast of Thunder Bay, Ontario, Canada. Location of reference (squares), buffer (circles) and fire (triangles) sites are distributed in six watersheds with comparable soil condition and vegetation composition.

Table 1. General biophysical conditions (mean ± standard error) of the streams selected within the (a) reference, (b) buffer, and (c) fire sites.

Site Code	Micro. (n)[a]	VPD (pKa)[b]	OMD (cm)[c]	Soil temp. (°C)	Soil moisture (m³/m³)	Structure (n)[d]	Canopy cover (%)	Live stem density[e]	Downed stem density[e]	Snag stem density[e]	Windthrow (%)	Mortality (%)
a) Reference												
LFP21	30	1.55 ± 0.02	12.2 ± 0.8	22.0 ± 0.3	13.7 ± 1.2	18	90.0 ± 1.2	2166.7 ± 196.2	11.1 ± 11.1	77.8 ± 23.6	1.1 ± 1.1	4.6 ± 1.4
WTT23	29	1.13 ± 0.02	12.8 ± 1.0	20.1 ± 0.3	14.1 ± 1.7	17	89.1 ± 0.7	1223.5 ± 116.2	70.6 ± 29.4	176.5 ± 61.6	3.9 ± 1.5	13.1 ± 3.7
RSW24	29	1.45 ± 0.06	11.3 ± 0.9	15.4 ± 0.3	19.0 ± 0.9	17	78.7 ± 1.6	1070.6 ± 128.3	117.6 ± 38.6	223.5 ± 59.1	9.9 ± 3.7	25.1 ± 5.2
AM4	30	0.48 ± 0.07	13.6 ± 1.0	17.8 ± 0.2	30.4 ± 2.7	18	91.9 ± 2.7	2255.6 ± 151.5	300.0 ± 70.9	133.3 ± 36.2	10.9 ± 2.1	16.1 ± 2.2
RN5	30	1.35 ± 0.04	12.1 ± 0.9	14.5 ± 0.2	25.1 ± 2.3	18	92.4 ± 1.3	1966.7 ± 179.5	433.3 ± 102.6	222.2 ± 55.8	15.8 ± 3.6	25.5 ± 3.5
LIL3	30	1.59 ± 0.06	11.8 ± 0.7	13.6 ± 0.5	33.2 ± 0.4	18	67.7 ± 6.7	1344.4 ± 147.1	33.3 ± 24.3	255.6 ± 50.6	3.6 ± 2.8	18.1 ± 3.1
MAC4	30	0.68 ± 0.08	11.9 ± 0.9	15.6 ± 1.4	31.4 ± 0.7	18	13.4 ± 1.5	1500.0 ± 158.6	88.9 ± 49.1	366.7 ± 63.1	3.2 ± 1.8	22.1 ± 3.8
LIL2	30	1.43 ± 0.08	11.2 ± 0.8	19.0 ± 1.1	30.2 ± 0.8	18	87.2 ± 1.5	1622.2 ± 66.4	22.2 ± 15.2	111.1 ± 36.9	1.3 ± 0.9	7.4 ± 2.2
Average	**238**	**1.21 ± 0.03**	**12.1 ± 0.3**	**17.2 ± 0.3**	**24.7 ± 0.7**	**142**	**76.2 ± 2.3**	**1650.7 ± 61.9**	**135.2 ± 21.6**	**195.8 ± 18.7**	**6.2 ± 0.9**	**16.5 ± 1.3**
b) Buffer												
BEA4	30	1.00 ± 0.06	11.7 ± 0.7	17.7 ± 0.4	15.8 ± 0.7	16	66.0 ± 5.4	1062.5 ± 191.2	287.5 ± 91.2	400.0 ± 126.5	24.6 ± 8.0	41.2 ± 6.6
RRD1	29	2.74 ± 0.07	8.8 ± 0.6	22.4 ± 0.3	24.8 ± 3.0	15	67.2 ± 3.2	1546.7 ± 275.0	160.0 ± 65.3	466.7 ± 100.8	7.1 ± 3.1	26.4 ± 5.0
RRD2	29	1.76 ± 0.07	9.9 ± 0.6	18.3 ± 0.3	10.4 ± 1.6	14	68.3 ± 3.4	1342.9 ± 229.1	385.7 ± 103.7	557.1 ± 120.8	25.1 ± 7.5	46.7 ± 5.3
MBT2	29	1.79 ± 0.09	10.5 ± 0.8	19.4 ± 0.6	15.7 ± 0.9	11	55.1 ± 6.2	1309.1 ± 241.8	400.0 ± 97.2	36.4 ± 36.4	20.3 ± 4.9	21.7 ± 5.1
RC1	29	1.59 ± 0.10	11.2 ± 1.0	19.7 ± 0.5	15.5 ± 0.8	16	54.0 ± 5.7	1362.5 ± 200.2	312.5 ± 79.5	350.0 ± 71.9	20.9 ± 6.4	39.1 ± 7.1
HB11	28	1.90 ± 0.08	9.6 ± 0.6	20.5 ± 0.4	14.2 ± 0.4	12	55.4 ± 6.6	1116.7 ± 140.3	316.7 ± 116.7	83.3 ± 29.7	22.9 ± 7.5	27.6 ± 6.9
MBT3	30	1.08 ± 0.09	11.8 ± 0.8	17.2 ± 0.4	15.6 ± 0.7	14	66.3 ± 3.3	1542.9 ± 205.9	242.9 ± 63.5	142.9 ± 32.7	16.5 ± 5.0	27.3 ± 7.8
EEL1	28	1.93 ± 0.07	10.8 ± 0.9	22.7 ± 0.3	35.5 ± 1.0	13	58.8 ± 7.5	1307.7 ± 204.6	369.2 ± 100.9	369.2 ± 74.6	21.6 ± 6.6	40.6 ± 6.8
Average	**232**	**1.72 ± 0.04**	**10.6 ± 0.3**	**19.7 ± 0.2**	**18.4 ± 0.7**	**111**	**61.7 ± 1.9**	**1326.1 ± 75.8**	**304.5 ± 31.5**	**315.3 ± 34.4**	**19.8 ± 2.3**	**34.4 ± 2.4**
c) Fire												
NIP52	28	1.14 ± 0.03	3.3 ± 0.4	16.2 ± 0.2	20.5 ± 0.8	17	13.3 ± 1.5	11.8 ± 11.8	776.5 ± 152.3	1011.8 ± 137.7	40.8 ± 5.9	99.3 ± 0.7
NIP101	30	2.52 ± 0.11	4.8 ± 0.7	21.6 ± 0.6	44.8 ± 4.3	18	9.3 ± 1.6	33.3 ± 24.3	1577.8 ± 144.5	333.3 ± 62.6	81.3 ± 2.9	97.7 ± 1.7
NIP102	30	0.78 ± 0.10	2.1 ± 0.3	18.3 ± 0.2	38.7 ± 0.9	18	6.4 ± 1.3	0.0 ± 0.0	1322.2 ± 161.8	455.6 ± 91.2	72.9 ± 5.5	100.0 ± 0.0
NIP103	30	2.54 ± 0.11	3.4 ± 0.6	21.6 ± 0.6	42.3 ± 2.4	18	11.7 ± 1.6	22.2 ± 15.2	1400.0 ± 102.3	644.4 ± 44.4	66.8 ± 2.7	98.8 ± 0.8
THU16S	29	1.17 ± 0.03	3.5 ± 0.4	16.1 ± 0.2	20.9 ± 0.9	18	14.7 ± 1.5	33.3 ± 18.1	500.0 ± 56.6	1200.0 ± 118.8	30.3 ± 3.2	97.9 ± 1.1
THU16N	29	1.31 ± 0.05	3.8 ± 0.7	16.5 ± 0.4	17.8 ± 0.6	17	15.3 ± 1.2	0.0 ± 0.0	552.9 ± 84.9	1411.8 ± 67.5	27.3 ± 3.4	100.0 ± 0.0
THU576	28	1.61 ± 0.05	3.3 ± .05	16.2 ± 0.1	19.7 ± 0.5	16	10.8 ± 1.0	0.0 ± 0.0	575.0 ± 85.4	1162.5 ± 84.1	32.9 ± 4.6	100.0 ± 0.0
THU577	29	1.80 ± 0.05	3.8 ± 0.7	16.9 ± 0.2	19.7 ± 0.8	17	12.1 ± 0.8	11.8 ± 11.8	658.8 ± 81.8	1211.8 ± 100.7	34.4 ± 3.3	99.3 ± 0.7
Average	**233**	**1.61 ± 0.05**	**3.5 ± 0.2**	**17.9 ± 0.2**	**29.0 ± 1.1**	**139**	**11.7 ± 0.5**	**14.4 ± 4.9**	**930.9 ± 53.1**	**919.4 ± 45.1**	**48.9 ± 2.2**	**99.1 ± 0.3**

[a]Microclimate sample size for each site. [b]VPD = vapour pressure deficit. [c]OMD = organic matter depth. [d]Structure sample size for each site. [e]Stem density = stems/ha.

10

Site Selection

All streams considered for this study had i) approximately 1 km^2 catchment area, ii) shoreline slopes ≤ 15%, iii) north to south (or vice versa) flow directions, and iv) stands dominated by *Picea mariana*.

GIS maps derived from digital elevation models (also called flow accumulation models) were used to identify streams with approximately 1 km^2 catchment areas (Wilson and Gallant, 2000). Buffer zones are an integral component of Ontario's "Timber Management Guidelines for the Protection of Fish Habitat" (Ontario Ministry of Natural Resources (OMNR, 1988)). Under these guidelines, forest managers are directed to delineate a zone called an "area of concern" (AOC) or riparian buffer reserve, along all streams large enough to be shown on a 1:50,000 scale map. Typically, 1 km^2 catchment area streams are the smallest size recognizable on such maps and make up very large components of watershed area. Within this document the strip of upland reserve retained following harvesting will be referred to as a "buffer". I choose streams with shoreline slopes ≤ 15% for consistency of buffer width. Edge orientation has been found to have a profound effect on stand structure (Harper et al., 2005) as well as vegetation composition (Hylander 2000). Consequently, streams with north to south flow direction were selected to reconcile aspect influences. Forest stands of similar canopy species and total stem density were selected. Appendix II provides a detailed comparison of stand composition and Appendix III provides information of stem density between sites.

I sampled a total of 24 streams, 8 streams with a 28 to 52 m buffer retained on either side of the stream 2 to 6 years post-harvest, 8 streams recovering from a natural fire that occurred 2 to 7 years previously, and 8 streams in undisturbed forest (Figure 2).
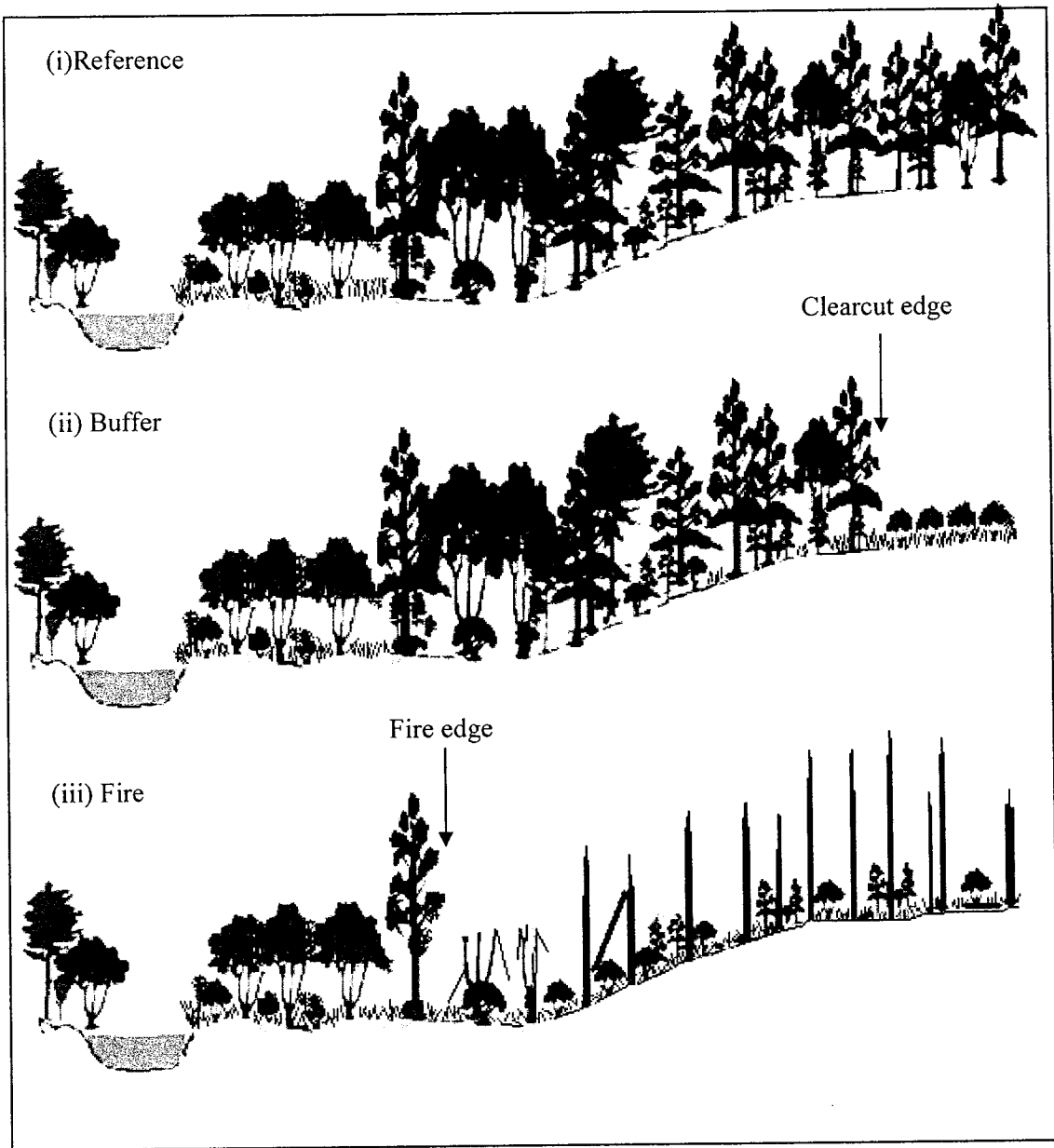
Figure 2. Generalized profile diagrams of streams, (i) in reference (undisturbed) forest, (ii) riparian buffers (28 – 60 m) and (iii) subject to natural fires.

The date of forest harvesting along buffered streams was verified within the 2001-2005 age class interval using harvest depletion maps provided by the OMNR. Only sites with buffers on both sides of the stream were selected. Site specific harvest dates are provided in Appendix I. Streams subjected to wildfire were selected within approximately the similar age class (2 to 7 years post-fire). Fire start dates ranged from April to August with fire weather index ranging 2 to 15 (Raman, 2008). The undisturbed mature forest streams were considered "reference" streams where no land use activity had occurred within 500 m. Most boreal studies have found edge influences disappear within 50 m of forest edge (Harper et al. 2005). The reference sites were located within approximately 90 to 100 years old fire regenerated forest.

Vegetation surveys

I used belt transects to determine plant community composition along a 64 m lateral gradients starting at stream edge. Four transects were used per site. Two transects ~50 m apart were laid on either side of the stream and all transects were considered independent of each other. Edge effects are not considered to be monotonic, therefore, a fine enough scale of sampling must be used to give precise estimates and identify significant fluctuations (Murica, 1995). Accordingly, along each transect 1 m^2 quadrats were placed at 4 m intervals for a total of 17 quadrats per transect. In instances where the 4 m interval did not match the ecotonal or buffer edge location, additional quadrats were placed at the identified edge. Within each of the 1244 quadrats I determined the percent cover of all vascular plant and bryophyte species (with the exception of liverworts and lichens) by ocular method. Plant samples difficult to identify in the field were brought to the laboratory for identification using taxonomic keys and herbarium specimens.

Habitat parameters

Within each of the vegetation survey quadrats, I measured the following habitat parameters in three random locations: soil moisture and temperature using a HH2 Moisture Meter with a WET Sensor type WET-2 (Delta T Devices, Cambridge, UK), air temperature and relative humidity using a traceable hygrometer (Model 35519-050, Control Company, Friendswood, Texas, USA), and organic matter depth by digging small soil pits. For each sampling plot vapour pressure deficit (VPD) was calculated from the mean air temperature and mean relative humidity using the following equation:

$$VPD = vpsat(T) \times (100 - RH) / 100$$

Where RH is the relative humidity (in percentages) and vpsat(T) is the saturated vapor pressure (in kilopascals) at air temperature T (in degrees Celsius).

Vegetation structure

The influence of tree canopy structure on near ground microclimate which influence understory species cannot be ignored. To quantify canopy structure, I laid out 5 x 10 m contiguous quadrats along each transect on either side of the stream. I recorded diameter of all trees > 5 cm in each 50 m^2 quadrat at breast height diameter (BHD, at 1.3 m from the ground). To determine the extent of windthrow and mortality I recorded the number of trees fallen on the ground as "downed" trees. Downed treefalls were those fallen since logging/fire occurred, not those incorporated into the forest floor, rotting or covered with moss. Dead standing trees were recorded as "snags". Windthrow was calculated as the ratio of downed: total stem density. Mortality was calculated as the ratio of downed + snags: total stem density. The basal area (BA) of each tree was calculated by multiplying the BHD by 0.00007854. I used a convex spherical

densiometer (Model A, Forest Densiometers, Oklahoma, USA) to quantify the canopy closure by taking measurements in four directions at three random points within each quadrat.

Data analysis

Edge location

Transects were laid perpendicular to stream starting at high water mark to a length of 64 m. The position of ecotonal edge (riparian width) was considered to be the point at which mature canopy trees were present. Also this was considered the point at which there was recognizable shift in vegetation from riparian obligate species to those typical of the upland understory. The exact position of the buffer edge was considered to be the canopy dip line of mature uncut trees.

A Leven's and Kolmogorov-Smirnov tests were initially run on edge location data (distance from stream in meters) to test for homogeneity of variances and normality respectively. Next, a univariate General Linear Model (GLM) coupled with a Tukey Honestly Significant Differences (HSD) post hoc test was run on the ecotonal distance data to identify if there was a detectable difference in the location across the treatments. Finally, a univariate GLM was applied to the clearcut edge and fire edge location data to highlight their difference in distance from stream edge. GLMs were chosen because they incorporate a number of different statistical models including: ANOVA, ANCOVA and MANOVA, as well are very robust to failures of normality (McDonald, 2008). All statistical analysis was conducted using SPSS version 16 (SPSS Inc., Chicago, Illinois).

Analysis of edge effects

Three general groups of response variables were tested for edge effects: (1) structural variables including; stem density, basal area, windthrow, mortality and canopy cover; (2)

microclimatic variables including; soil temperature, soil moisture, vapour pressure deficit and organic matter depth; (3) understory vegetation composition variables including; species percent cover and richness. Understory species were further divided into groups based on life form and functional types such as: tree saplings, shrubs, herbs, grasses, sedges, ferns, horsetails, clubmosses and mosses. Other groups were further divided based on physiognomy, such as conifer and deciduous saplings, tall and low-growing shrubs and acrocarpous, pluerocarpous and sphagnum mosses. The herb group was further divided based on shade-tolerance (Stewart, 2004).

I evaluated edge effects on each response variable by three indices: significance of edge influence (SEI), depth of edge influence (DEI), and magnitude of edge influence (MEI). The following describes each in more detail.

Significance of edge influence (SEI)

The calculation of SEI has become a common practice for edge related studies (Burton, 2002; Harper et al., 2005). SEI reflects the statistical difference between response variables measured at the edge compared with interior forest (Chen et al. 1995). Following Burton (2002), an ANOVA was used to calculate SEI with plots nested within edge type (blocking factor). This ANOVA was used to determine if a response variable in the plot immediately adjacent to the clearcut/fire edge was significantly different from the mean reference forest value of corresponding distances (Figure 3). This approach was used because I assumed that natural gradients existed from the riparian zone to upland. Individual transects were analyzed separately, because differing buffer widths (as well as fire edge distances) were used. Stewart and Mallik (2006) found that for structural parameters, the depth of edge influence at clearcut edges penetrated 10 m. Therefore, it was assumed that forest structure at the ecotonal edge location of buffer and interior reference forest would not differ.

(a) Reference streams

Transect

Distance from stream (m)

Stream

Quadrats analyzed at comparable distances

(b) Buffer / fire streams

Clearcut / fire edge

Transect

Distance from stream (m)
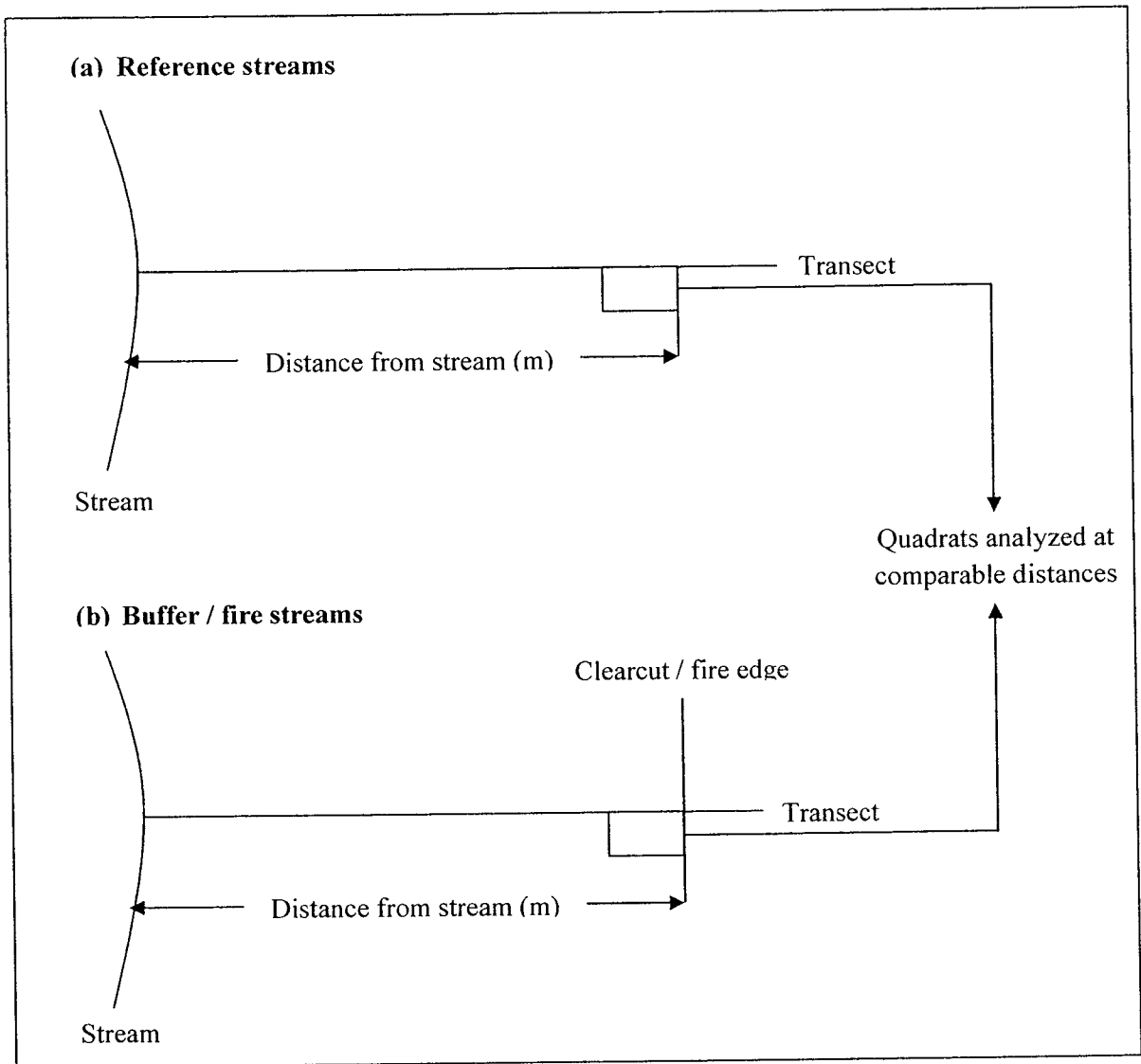
Stream

Figure 3. Schematic diagram of the quadrats within; (a) reference sites, and (b) buffer/ streams fire sites, used for the determination of SEI and MEI.

As multiple investigations have shown (Harper et al. 2005; Stewart and Mallik, 2003; Hylander, 2002) that edge orientation can have a significant effect on the measured response variable. Therefore, east and west-facing edges were analyzed separately. Again, an ANOVA with edge plots (i.e. quadrats immediately adjacent to the edge) nested within edge orientation was conducted to determine if there was significance difference between edges of east and west orientation.

Magnitude of edge influence (MEI)

Harper et al. (2005) defined and explained that both MEI and SEI can be used as a measure of the extent to which a given vegetation and environmental parameter differs at the edge, as compared with the reference "nonedge" (undisturbed forest). To avoid confusion, MEI and SEI are used to distinguish between the magnitude of an ecological effect and its statistical significance, respectively. Following Harper et al. (2005) MEI was calculated as $(e - r)/(e + r)$, where e equals the value of the parameter at the edge and r equals the value of the parameter in the reference forest. The MEI thus, varies between -1 and +1 and is equal to 0 when there is no edge influence.

Depth of edge influence (DEI)

DEI is considered the set of distances from the edge into adjacent community over which there is a statistically significant edge influence; also known as distance of edge influence (Esukirchen et al., 2001), and edge width (Forman and Godron, 1986). Although there is generally accepted notion that edge effects are detrimental for forest fragments, there is no generally accepted statistical method for quantifying the spatial scale of edge influences (Ewers and Didham, 2006). Furthermore, the assessment DEI is essential for determining the effective size of interior forest fragments and buffer strips (Harper and Macdonald, 2001).

Early edge related research assumed that edge effects vary monotonically with distance from edge and therefore plotted linear functions assuming that the interception point of the response variable with interior conditions would represent the DEI (e.g., Ranney et al., 1981). Later studies found peaks and depressions in edge effects at intermediate distances from the edge (Hester and Hobbs, 1992; Palik and Murphy, 1990), which dismissed the notion of monotonicity. Other studies have compared parameter values to an arbitrary proportion of the value obtained in the habitat interior (Chen et al., 1992; Brand and George, 2001; Hylander, 2005) and at other times a subjective, visual inspection of graphs (Chen et al., 1995). A more rigorous approach (critical values approach) has been introduced to compare parameter values near edges to the range of variation in that parameter that occurs in the habitat interior (Laurance et al., 1998; Harper and MacDonald, 2001: Harper et al., 2005).However, statistical significance of this procedure is based on comparison to critical values calculated through randomization tests on interior data and therefore is not applicable to situations where natural gradients are established within the interior forest condition (e.g. riparian buffer strips). Most recently, Cancino (2005) has proposed the use of piecewise, or breakpoint, regression to determine the location of ecological thresholds such as DEI. However, as Ewers and Didham (2006) explain that this approach aims to identify discrete (compartmentalized) changes in response rates across boundaries, when in fact the rates of change of most response functions are continuous.

The use of repeated measures ANOVA has become an accepted and reputable approach for the determination of DEI (e.g. Polhman et al., 2009; Boudreault et al., 2008; Mascarúa-Lopez et al., 2006) I adopted this approach for this study. Prior to any statistical analysis, the MEI of the response variables (structural, microclimatic, and compositional) were plotted against distance to verify a trend extended into the interior from from edge location. Response variables with very

erratic behaviour (many spikes and valleys at multiple locations from edge) or those showing little departure from zero (indicating little difference than reference forest) were removed from further analysis. The repeated measures ANOVA was run with treatment as the grouping variable and plot distance along the transects as the repeated measure. A post hoc test cannot be run under this model because the error term has zero degrees of freedom. Therefore, for variables deemed significant by the repeated measures, a factorial ANOVA with distance as the fixed factor was run to attain Tukey HSD results (used to determine DEI). DEI was considered the point at which two consecutive plots were significantly different (i.e. $p \leq 0.05$) than the reference forest.

DEI is largely affected by edge type and geographic location. For the terms of comparisons, "maintained" (i.e. agricultural) edges, and edge studies conducted in tropical and western North America were avoided. The primary comparisons of this study were with regenerating forest edges in boreal and eastern North America. Appendix IV provides synthesis of edge influences on forest structure, near ground microclimate and composition from published referred studies considered by this study.

Plant community analyses

Multivariate community level analyses were performed using Multiple Response Permutation Procedure (MRPP) and Indicator Species Analysis. MRPP is a nonparametric procedure for testing the hypothesis of no difference between two or more groups of entities (McCune and Grace, 2002). It calculates the observed delta, which is a linear combination for all groups of the average pair-wise distances between each sample within a particular group. The observed delta is compared to the expected delta, which is the null distribution of all possible permutations of samples into groups of the same size. A test statistic T is calculated from a

Pearson type III distribution to derive the probability of having (p-value) a delta the same size or smaller than the observed delta. The test statistic describes the separation between the groups. The more negative is T, the stronger the separation (McCune and Grace, 2002). An A statistic (chance-corrected within-group agreement) is also calculated from the observed and expected deltas and describes within-group homogeneity, compared with random expectation.

Dividing complex datasets in various ways allows a number of different questions to be answered. Locations along transects were divided into the following microhabitats; (1) riparian, (2) ecotonal, (3) upland, (4) clearcut edge, and (5) clearcut/burn matrix (Figure 4). Because of the variable nature of the riparian zone and buffer widths, quadrats along each transect were classified under the following protocol (based on field observations) to attain a balanced design: the first two from stream edge were considered the riparian zone, three quadrats were classified as the ecotone/fire edge (i.e. one on either side of the ecotonal edge), two quadrats interior to the clearcut edge were considered the upland zone; three quadrats were grouped as clearcut edge (one on either side of the identified cut edge), final three quadrats of the transects were consider the clearcut/burn matrix. MRPP was used to test the following null hypotheses: (i) no understory species compositional difference between site type at the 5 microhabitats, and (ii) no life form compositional difference between the clearcut and fire edges. I used the Sørensen distance measure and default weighting of groups $(C(I) = n(I)/sum(n(I)))$ for the MRPP tests. I chose Sørensen distance measure because it is less prone to exaggerate the influence of outliers (McCune and Grace 2002).
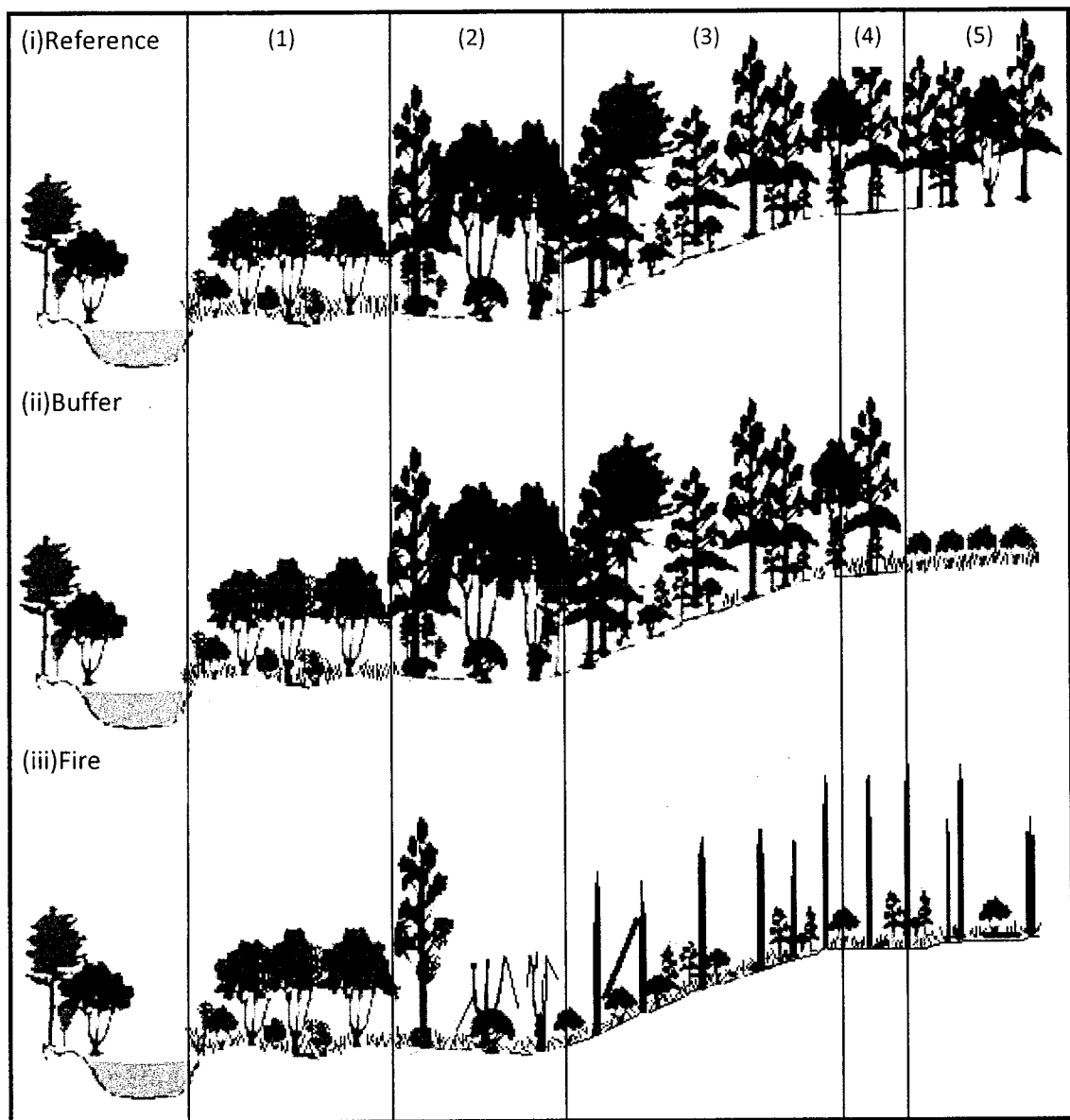
Figure 4. . Generalized profile diagrams of streams: (i) in reference (undisturbed) forest, (ii) in riparian buffers, and (iii) subject to natural fires. Numbers represent spatial divisions of regions along the transects, (1) riparian, (2) ecotone, (3) upland, (4) clearcut edge, and (5) clearcut/fire matrix.

One of the aims of plant community analysis is to identify different species that can detect and describe environmental conditions, levels of disturbance or experimental treatments (McCune and Grace 2002). Indicator species analysis provides a method for combining information about species abundance and frequency within a particular group (Dufrene and Legendre, 1997). Indicator values were determined for each treatment according to the same divisions of microhabitats used in MRPP tests. A Monte Carlo randomization technique with 5000 runs was used to test the statistical significance of the indicator values. The response variable used in MRPP and indicator species analysis was percent cover of all understory species. All multivariate tests were performed using PC-ORD version 5 (MjM Software Design, 2008, Glenden Beach, Oregon, USA).

Results

Edge location

The ecotonal edge location ranged from 3 to 15 m from the stream across the treatments. The mean ecotonal edge distance among the three treatments did not differ significantly (p = 0.5021; Figure 5). The buffer widths (strip of upland forest) also had a large range from 28 to 60 m with a mean of 40.19 m. The clearcut edge location was significantly (p = 0.0014) higher (30.97 m) than the average distance of the fire edge (9.22 m) starting from stream bank.
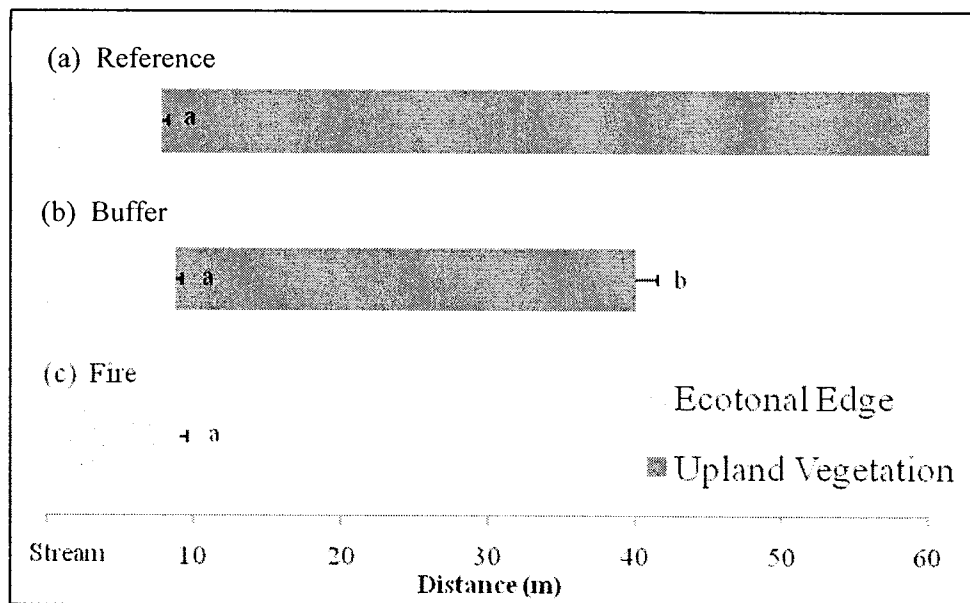
Figure 5. Mean (±SEM) edge distance from stream bank. Different letters means that the edges were significantly different. The reference upland forest cover continued for 100s of meters and therefore does not have a SEM associated with it.

Edge orientation

Forest structure between east compared to west-facing edges

Overall, the structural attributes of east and west-facing edges at the buffer edges were similar. Basal area of snags was 8.7% higher at west-facing buffer edges than the east facing

edge (Figure 6a & b). Also, the west-facing edges had a significantly higher sang stem density than east-facing edges (400 and 250 stems/ha respectively). No detectable difference was identified between east and west-facing reference locations at the averaged distance of the buffer edge (Table 2).

Canopy cover was the only response variable that showed a statistically detectable difference between orientations at fire edges (Figure 6c & d). The west-facing fire edges had almost twice as much canopy cover than the east-facing edges (Figure 6d). Aspect had no effect on any of the response variables at the ecotonal edge location in reference streams (Table 3).

Edge orientation and structure of reference forest

When edge orientation of both buffer and fire were compared individually (east-facing clearcut edge to east-facing reference forest of comparable distance, etc.), both buffer edge orientations showed a significantly lower live stem density and canopy cover and a higher downed stem density as well as percentage of windthrow and mortality (Table 4). Also, compared to reference forest condition both fire edge orientations responded in a similar manner (Table 5). Since the east and west facing edge data of structural variables responded similarly, it was pooled together to increase the power of the subsequent analyses by increasing the sample size.
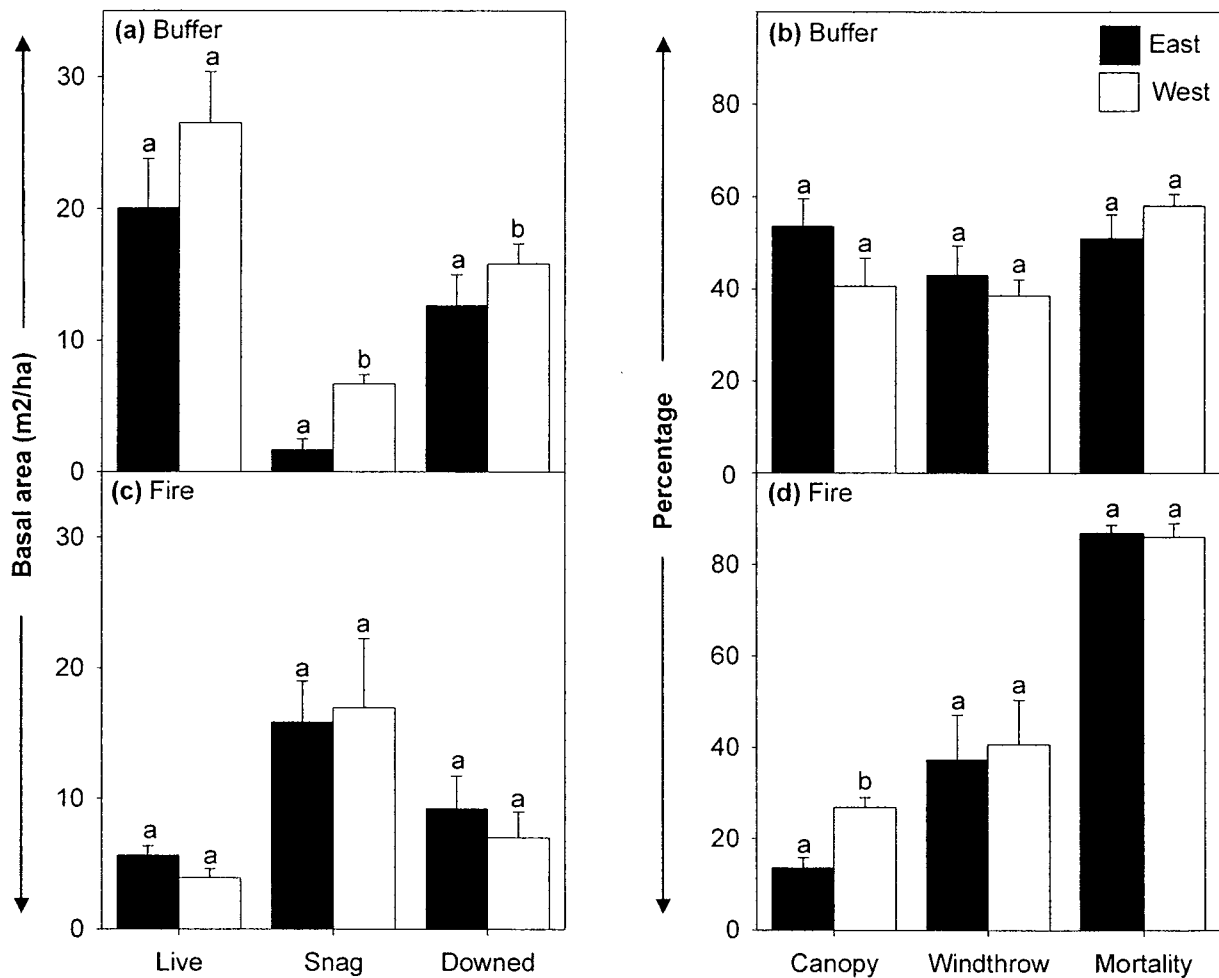
Figure 6. Mean (±SEM) for basal area (m^2/ha), canopy cover (% covered sky), windthrow (%) and mortality (%) at east and west-facing buffer (a & b) and fire edges (c & d). Significant difference between edge orientations (p≤0.05) are indicated by differing letters (refer to Tables 2 & 3 for exact values).

Table 2. Mean (±SEM) structural, microclimatic, and compositional variables at east versus west-facing buffer edges and comparable distances in reference forest. An ANOVA with plots nested within edge orientation was used to test the effect of orientation (refer to methods for more detail).

Structure Attribute	Reference East-Facing (n = 8)	Reference West-Facing (n = 8)	F-ratio	Sig. (Prob.>F[a])	Buffer East-Facing (n = 8)	Buffer West-Facing (n = 8)	F-ratio	Sig. (Prob.>F[a])
Live SD[b]	1857.1 ± 331.6	1675 ± 354	0.0237	0.8799	975.0 ± 148.5	925 ± 113	0.8604	0.3693
Snag SD[b]	285.7 ± 122.3	175 ± 95.90	0.0739	0.7897	150.0 ± 62.7	400 ± 37.80	4.1718	**0.0406**
Down SD[b]	200.0 ± 130.9	325 ± 141.11	0.0738	0.4255	825.0 ± 116.1	850 ± 111.80	0.0214	0.8790
Total SD[b]	2342.9 ± 337.3	2175 ± 378.79	0.0567	0.8747	1950.0 ± 135.0	2175 ± 201.56	4.2889	0.0573
Live BA[c]	36.5 ± 4.7	36.06 ± 6.43	0.3122	0.5852	20.1 ± 3.7	26.51 ± 3.85	2.7420	0.1200
Snag BA[c]	3.1 ± 1.5	2.72 ± 1.11	0.2805	0.6047	1.7 ± 0.8	6.70 ±0.70	5.4530	**0.0349**
Down BA[c]	7.5 ± 5.7	10.20 ± 6.86	0.0000	0.9977	12.7 ± 2.3	15.84 ± 1.53	2.1019	0.1691
Total BA[c]	47.1 ± 7.1	48.99 ± 8.72	0.4870	0.4967	34.4 ± 4.0	49.05 ± 3.91	11.6670	**0.0042**
Ave. live BA[c]	4.2 ± 0.6	4.7 ± 0.7	0.0707	0.7942	3.5 ± 0.2	4.8 ± 0.6	1.4451	0.2493
Ave. snag BA[c]	4.7 ± 0.7	4.7 ± 0.7	0.1843	0.6743	4.4 ± 0.8	6.1 ± 0.9	1.2971	0.2739
Ave. down BA[c]	1.8 ± 1.3	2.8 ± 1.3	0.3592	0.5585	3.1 ± 3.4	4.0 ± 0.4	21.1510	**0.0004**
Ave. tree BA[c]	2.8 ± 1.4	1.8 ± 1.1	0.0020	0.9653	2.3 ± 0.8	3.6 ± 0.5	1.8269	0.2050
Windthrow (%)[d]	16.0 ± 3.9	17.16 ± 6.56	2.5682	0.1313	43.1 ± 6.3	38.57 ± 3.48	0.3892	0.5427
Mortality (%)[d]	22.8 ± 5.3	24.51 ± 6.95	0.1280	0.7259	51.1 ± 5.1	58.09 ± 2.56	1.5286	0.2367
Canopy Cover (%)[d]	89.0 ± 3.8	80.79 ± 6.04	1.3195	0.2699	53.6 ± 6.0	40.59 ± 6.17	2.2994	0.1517
Microclimate Attribute	East-facing (n = 24)	West-facing (n = 24)	F-ratio	Sig. (Prob.>F[a])	East-facing (n = 24)	West-facing (n = 24)	F-ratio	Sig. (Prob.>F[a])
OMD (cm)[e]	10.93 ± 0.87	10.72 ± 0.86	0.0300	0.9000	9.30 ± 0.60	9.16 ± 0.65	0.0133	0.9098
Soil temp. (°C)[d]	17.20 ± 0.56	16.41 ± 0.73	0.7258	0.7086	1.75 ± 0.11	1.61 ± 0.11	0.2955	0.5953
Soil moist. (m³/m³)[d]	24.69 ± 2.04	24.43 ± 3.60	0.4002	0.9028	20.30 ± 0.50	18.99 ± 0.54	1.5333	0.236
VPD (kPa)[f]	1.22 ± 0.10	1.32 ± 0.12	0.4272	0.6355	18.18 ± 2.36	17.21 ± 1.82	0.0374	0.8495
Composition Attribute	East-Facing (n = 48)	West-Facing (n = 48)	F-ratio	Sig. (Prob.>F[a])	East-Facing (n = 48)	West-Facing (n = 48)	F-ratio	Sig. (Prob.>F[a])
Abundance[d]	142.77 ± 6.40	148.69 ± 5.91	0.3421	0.5630	186.44 ± 8.62	187.79 ± 8.91	0.0091	0.9246
Richness[d]	9.33 ± 0.26	9.63 ± 0.44	0.1998	0.6581	12.46 ± 0.48	11.94 ± 0.43	0.6452	0.4282

[a] Results of ANVOA. If Prob.>F is less than 0.05, then the difference between edge orientation is considered significant at the 95% confidence level.
[b] SD = stem density (stems/ha). Log transformed for analysis.
[c] BA = basal area (m²/ha). ln transformed for analysis.
[d] Raw data (untransformed) used for analysis.
[e] OMD = organic matter depth. Square root transformed for analysis.
[f] VPD = vapour pressure deficit. Untransformed data used for analysis.

Table 3. Mean (±SEM) structural, microclimatic, and compositional variables at east versus west-facing fire edges and comparable distances in reference forest. An ANOVA with plots nested within edge orientation was used to test the effect of orientation (refer to methods for more detail).

	Reference				Fire			
Structure Attribute	East-Facing (n = 8)	West-Facing (n= 8)	F-ratio	Sig. (Prob.>F^a)	East-Facing (n = 8)	West-Facing (n= 8)	F-ratio	Sig. (Prob.>F^a)
Live SD^b	1450.0 ± 306.5	1550.0 ± 304.1	0.0536	0.8202	225.0 ± 25.0	225.0 ± 45.3	0.0000	1.0000
Snag SD^b	50.0 ± 24.2	125.0 ± 64.8	0.8400	0.3749	850.5 ± 159.2	800.0 ± 207.0	0.0366	0.8509
Down SD^b	125.0 ± 52.6	50.0 ± 50.0	1.0678	0.3190	675.0 ± 192.5	675.0 ± 177.0	0.0000	1.0000
Total SD^b	1625.0 ± 296.3	1725.0 ± 320.6	0.0525	0.8221	1750.0 ± 62.8	1700.0 ± 106.9	0.1628	0.6927
Live BA^c	28.9 ± 7.4	27.3 ± 6.1	0.0263	0.8734	5.7 ± 0.7	3.9 ± 0.7	3.0759	0.1013
Snag BA^c	0.18 ± 0.09	0.8 ± 0.4	1.7683	0.2048	15.8 ± 3.1	17.0 ± 5.3	0.0322	0.8602
Down BA^c	3.1 ± 1.9	0.8 ± 0.8	1.1845	0.2948	9.2 ± 2.5	7.0 ± 2.0	0.4871	0.4967
Total BA^c	32.2 ± 7.0	28.9 ± 6.2	0.1222	0.7319	30.7 ± 1.3	27.9 ± 5.1	0.2931	0.5979
Ave. live BA^c	5.4 ± 1.6	4.1 ± 1.0	0.5060	0.4886	5.2 ± 0.7	3.3 ± 0.6	4.1342	0.0614
Ave. snag BA^c	0.1 ± 0.1	0.4 ± 0.3	2.1544	0.1643	3.7 ± 0.5	3.5 ± 0.9	0.0559	0.8166
Ave. down BA^c	2.1 ± 1.1	0.4 ± 0.4	2.2300	0.1575	2.5 ± 0.4	1.9 ± 0.5	0.7140	0.4123
Ave. tree BA^c	4.8 ± 1.3	3.7 ± 0.8	0.4818	0.4990	3.6 ± 0.2	3.2 ± 0.5	0.3894	0.5426
Windthrow (%)^d	12.6 ± 8.0	2.1 ± 2.1	0.4910	0.4950	37.3 ± 9.8	40.7 ± 9.7	0.0694	0.7961
Mortality (%)^d	15.1 ± 7.8	9.1 ± 3.6	1.6179	0.2241	86.9 ± 1.7	86.0 ± 3.0	0.0799	0.8115
Canopy Cover (%)^d	79.3 ± 5.1	65.4 ± 9.5	1.6743	0.2166	13.6 ± 2.2	26.9 ± 2.2	18.2750	**0.0008**
Microclimate Attribute	East-facing (n = 24)	West-facing (n = 24)	F-ratio	Sig. (Prob.>F^a)	East-facing (n = 24)	West-facing (n = 24)	F-ratio	Sig. (Prob.>F^a)
OMD (cm)^e	16.00 ± 1.38	15.75 ± 0.85	0.0098	0.9225	6.51 ± 0.89	6.64 ± 0.99	0.0072	0.9338
Soil temp. (°C)^d	24.17 ± 1.93	24.37 ± 2.69	0.0015	0.9695	18.13 ± 0.81	17.30 ± 0.70	0.2353	0.6352
Soil moist. (m^3/m^3)^d	17.41 ± 0.49	16.12 ± 0.91	0.5052	0.4889	29.77 ± 1.86	41.87 ± 8.16	1.7116	0.2118
VPD (kPa)^f	1.10 ± 0.09	1.26 ± 0.09	0.5075	0.4879	1.42 ± 0.07	1.70 ± 0.14	1.0270	0.3281
Composition Attribute	East-Facing (n = 48)	West-Facing (n= 48)	F-ratio	Sig. (Prob.>F^a)	East-Facing (n = 48)	West-Facing (n= 48)	F-ratio	Sig. (Prob.>F^a)
Abundance^d	156.21 ± 7.55	160.60 ± 6.71	0.2255	0.6383	133.63 ± 10.20	146.52 ± 8.95	0.5234	0.4750
Richness^d	12.46 ± 0.70	11.33 ± 0.69	1.7437	0.1967	9.75 ± 0.66	9.23 ± 0.59	0.2231	0.6401

^{a} Results of ANVOA. If Prob.>F is less than 0.05, then the difference between edge orientation is considered significant at the 95% confidence level.
^{b} SD = stem density (stems/ha). Log transformed for analysis.
^{c} BA = basal area (m^2/ha). ln transformed for analysis.
^{d} Raw data (untransformed) used for analysis.
^{e}OMD = organic matter depth. Square root transformed for analysis.
^{f}VPD = vapour pressure deficit. Untransformed data used for analysis.

Table 4. Mean (±SEM) structural attributes of: (i) east-facing buffer edges versus east-facing reference plots (of comparable distance) and (ii) west-facing buffer edges versus west-facing reference plots. An ANOVA with plots nested within edge type was used to test the effect of the disturbance (refer to methods for more detail).

Attribute	(i)East-facing edges				(ii)West-facing edges			
	Reference (n = 8)	Buffer (n = 8)	F-ratio	SEI (Prob>F[a])	Reference (n = 8)	Buffer (n = 8)	F-ratio	SEI (Prob>F[a])
Live stem density (stems/ha)[b]	1857.1 ± 331.6	975.0 ± 148.5	6.4536	**0.0246**	1675 ± 354	925 ± 113	4.0645	**0.0234**
Snag stem density (stems/ha)[b]	285.7 ± 122.3	150.0 ± 62.7	1.0534	0.3234	175 ± 95.90	400 ± 37.80	4.7647	**0.0466**
Downed stem density (stems/ha)[b]	200.0 ± 130.9	825.0 ± 116.1	12.8530	**0.0033**	325 ± 141.11	850 ± 111.80	8.5041	**0.0113**
Live basal area (m^2/ha)[c]	36.5 ± 4.7	20.1 ± 3.7	7.7723	**0.0154**	36.06 ± 6.43	26.51 ± 3.85	1.6252	0.2231
Snag basal area (m^2/ha)[c]	3.1 ± 1.5	1.7 ± 0.8	0.7248	0.4100	2.72 ± 1.11	6.70 ± 0.70	9.1392	**0.0091**
Downed basal area (m^2/ha)[c]	7.5 ± 5.7	12.7 ± 2.3	0.7685	0.3966	10.20 ± 6.86	15.84 ± 1.53	0.6420	0.4364
Ave. live basal area (m^2/ha)[c]	4.2 ± 0.6	3.5 ± 0.2	0.7746	0.2734	4.7 ± 0.7	4.8 ± 0.6	1.7332	0.9420
Ave. snag basal area (m^2/ha)[c]	4.7 ± 0.7	4.4 ± 0.8	0.0845	0.7751	4.7 ± 0.7	6.1 ± 0.9	1.5201	0.2379
Ave. log basal area (m^2/ha)[c]	1.8 ± 1.3	3.1 ± 3.4	0.9075	0.3569	2.8 ± 1.3	4.0 ± 0.4	0.7628	0.3972
Windthrow (%)[d]	6.0 ± 3.9	43.1 ± 6.3	23.1620	**0.0003**	17.16 ± 6.56	38.57 ± 3.48	8.3184	**0.0120**
Mortality (%)[d]	22.8 ± 5.3	51.1 ± 5.1	14.7750	**0.0020**	24.51 ± 6.95	58.09 ± 2.56	20.5350	**0.0005**
Canopy cover (%)[d]	89.0 ± 3.8	53.6 ± 6.0	24.8630	**0.0002**	80.79 ± 6.04	40.59 ± 6.17	21.6890	**0.0004**

[a] Results of ANVOA performed with; [b] log, [c] ln, [d] untransformed, and [e] square root transformed data. If Prob.>F is less than 0.05, then the difference between edge orientation is considered significant at the 95% confidence level.

Table 5. Mean (±SEM) structural attributes of: (i) east-facing fire edges versus east-facing reference plots (of comparable distance) and (ii) west-facing fire edges versus west-facing reference plots. An ANOVA with plots nested within edge type was used to test the effect of the disturbance (refer to methods for more detail).

Attribute	East-facing edges				West-facing edges			
	Reference (n = 8)	Fire (n = 8)	F-ratio	SEI (Prob>F[a])	Reference (n = 8)	Fire (n = 8)	F-ratio	SEI (Prob>F[a])
Live stem density (stems/ha)[b]	1450.0 ± 306.5	225.0 ± 25.0	15.8710	0.0014	1550.0 ± 304.1	225.0 ± 45.3	18.5680	0.0007
Snag stem density (stems/ha)[b]	50.0 ± 24.2	850.5 ± 159.2	22.9740	0.0003	125.0 ± 64.8	800.0 ± 207.0	9.6831	0.0077
Downed stem density (stems/ha)[b]	125.0 ± 52.6	675.0 ± 192.5	6.4431	0.0074	50.0 ± 50.0	675.0 ± 177.0	11.5440	0.0043
Live basal area (m²/ha)[c]	28.9 ± 7.4	5.7 ± 0.7	9.7766	0.0074	27.3 ± 6.1	3.9 ± 0.7	14.5130	0.0019
Snag basal area (m²/ha)[c]	0.18 ± 0.09	15.8 ± 3.1	24.9480	0.0002	0.8 ± 0.4	17.0 ± 5.3	9.3351	0.0086
Downed basal area (m²/ha)[c]	3.1 ± 1.9	9.2 ± 2.5	3.7216	0.0742	0.8 ± 0.8	7.0 ± 2.0	8.1948	0.0125
Ave. live basal area (m²/ha)[c]	5.4 ± 1.6	5.2 ± 0.7	0.0110	0.9180	4.1 ± 1.0	3.3 ± 0.6	0.4865	0.4969
Ave. snag basal area (m²/ha)[c]	0.1 ± 0.1	3.7 ± 0.5	52.7170	>0.0001	0.4 ± 0.3	3.5 ± 0.9	9.2137	0.0089
Ave. log basal area (m²/ha)[c]	2.1 ± 1.1	2.5 ± 0.4	0.0857	0.7740	0.4 ± 0.4	1.9 ± 0.5	5.6417	0.0324
Windthrow (%)[d]	12.6 ± 8.0	37.3 ± 9.8	3.7702	0.0726	2.1 ± 2.1	40.7 ± 9.7	15.0590	0.0017
Mortality (%)[d]	15.1 ± 7.8	86.9 ± 1.7	79.7020	>0.0001	9.1 ± 3.6	86.0 ± 3.0	267.9800	>0.0001
Canopy cover (%)[d]	79.3 ± 5.1	13.6 ± 2.2	139.8700	>0.0001	65.4 ± 9.5	26.9 ± 2.2	15.6350	0.0014

[a] Results of ANVOA with; [b] log, [c] ln, [d] untransformed, and [e] square root transformed data. If Prob.>F is less than 0.05, then the difference between edge orientation is considered significant at the 95% confidence level.

Edge orientation and microclimate

There was no significant difference in organic matter depth (OMD), soil temperature (ST), soil moisture (SM), and vapour pressure deficit (VPD) between east and west-facing buffer and fire edges (Table 2 & 3). Also, when the disturbance related edge orientations were compared individually (i.e. east-facing buffer edge to east reference forest of comparable distance, etc.) little differences were identified.

Edge orientation and species composition

Edge orientation had no effect on richness and abundance of life forms sampled at clearcut and fire edges (Table 6 & 7).

Table 6. Mean (±SEM) life form abundance (A) and richness (R) at east versus west-facing buffer edges and comparable distance in reference forest. An ANOVA with plots nested within edge orientation was used to test the effect of orientation (refer to methods for more detail).

Life form		Reference East-Facing (n = 48)	West-Facing (n =48)	F-ratio	Sig. (Prob.>F^a)	Buffer East-Facing (n = 48)	West-Facing (n = 48)	F-ratio	Sig. (Prob.>F^a)
Conifer[b]	A	10.92 ± 2.87	4.67 ± 2.04	2.2463	0.1561	2.50 ± 1.48	0.52 ± 0.40	1.6219	0.2236
	R	0.79 ± 0.16	0.33 ± 0.10	2.7954	0.1167	0.25 ± 0.09	0.08 ± 0.06	2.0000	0.1792
Deciduous[c]	A	1.19 ± 0.68	1.77 ± 0.87	0.2948	0.5957	20.79 ± 4.81	8.10 ± 2.05	2.1484	0.1648
	R	0.21 ± 0.10	0.17 ± 0.08	0.1111	0.7438	1.17 ± 0.21	0.75 ± 0.16	1.1290	0.3060
Tall shrubs	A	6.56 ± 1.43	15.23 ± 2.53	0.0654	0.0504	24.65 ± 3.85	37.98 ± 4.28	3.3636	0.0880
	R	0.79 ± 0.15	1.42 ± 0.26	0.1474	0.0949	1.88 ± 0.16	2.50 ± 0.28	3.1818	0.0961
Low shrubs	A	35.60 ± 5.06	38.48 ± 4.70	0.0654	0.8019	49.13 ± 4.65	44.90 ± 3.48	3.3636	0.0880
	R	3.33 ± 0.33	3.63 ± 0.32	0.1474	0.7068	4.25 ± 0.31	4.46 ± 0.25	0.1547	0.7000
ST herbs[d]	A	28.42 ± 2.77	20.92 ± 3.27	2.1061	0.1688	23.71 ± 2.65	30.83 ± 2.93	2.1563	0.1641
	R	3.83 ± 0.24	2.63 ± 0.37	3.9378	0.0672	3.42 ± 0.28	3.71 ± 0.24	0.3749	0.5502
SIT herbs[e]	A	4.75 ± 1.67	6.75 ± 2.07	0.2403	0.6316	23.75 ± 2.79	23.02 ± 2.45	0.0222	0.8837
	R	0.67 ± 0.19	0.79 ± 0.21	0.1011	0.7552	2.79 ± 0.36	2.75 ± 0.35	0.0027	0.9590
Grasses	A	0.00 ± 0.00	0.73 ± 0.46	2.8824	0.1117	7.56 ± 2.23	7.15 ± 1.44	0.0191	0.8920
	R	0.00 ± 0.00	0.13 ± 0.07	4.2000	0.0596	0.92 ± 0.20	0.92 ± 0.16	0.0000	1.0000
Sedges	A	0.00 ± 0.00	0.27 ± 0.19	1.0000	0.3343	5.44 ± 2.10	4.81 ± 2.82	0.0245	0.8778
	R	0.00 ± 0.00	0.08 ± 0.08	1.0000	0.3343	0.63 ± 0.20	0.50 ± 0.19	0.1611	0.6942
Horsetails	A	0.08 ± 0.08	0.27 ± 0.19	0.4378	0.5189	0.00 ± 0.00	0.27 ± 0.15	3.9043	0.0682
	R	0.04 ± 0.04	0.08 ± 0.06	0.2000	0.6616	0.00 ± 0.00	0.13 ± 0.07	4.2000	0.0596
Club mosses	A	8.98 ± 1.84	5.40 ± 1.29	0.9550	0.3450	4.27 ± 1.33	6.42 ± 0.99	1.0599	0.3207
	R	1.21 ± 0.18	0.83 ± 0.16	1.0759	0.3172	0.67 ± 0.18	1.38 ± 0.19	3.2894	0.0912
Ferns	A	0.54 ± 0.39	3.25 ± 1.94	1.6402	0.2211	0.67 ± 0.34	0.33 ± 0.24	0.4308	0.5223
	R	0.08 ± 0.06	0.33 ± 0.18	1.2353	0.2851	0.17 ± 0.08	0.08 ± 0.06	0.3684	0.5536
Bryophytes	A	45.40 ± 4.65	50.25 ± 4.74	0.3126	0.5849	14.88 ± 2.15	16.90 ± 3.62	0.1102	0.7448
	R	4.33 ± 0.26	4.83 ± 0.46	0.5040	0.4894	2.83 ± 0.29	2.08 ± 0.29	2.2235	0.1581
Acrocarpous[f]	A	8.65 ± 1.35	12.19 ± 2.09	1.3943	0.2573	7.40 ± 1.65	5.17 ± 1.34	0.4864	0.4970
	R	1.63 ± 0.22	2.25 ± 0.32	2.2277	0.1577	1.67 ± 0.29	1.04 ± 0.20	1.7138	0.2116
Pleurocarpous[g]	A	36.75 ± 5.09	37.31 ± 4.42	0.0046	0.9471	7.48 ± 2.00	11.73 ± 3.65	0.4685	0.5049
	R	2.71 ± 0.19	2.46 ± 0.29	0.2897	0.5989	1.17 ± 0.20	1.04 ± 0.25	0.0928	0.7651
Sphagnum[h]	A	0.00 ± 0.00	0.75 ± 0.50	2.1892	0.1611	0.00 ± 0.00	0.00 ± 0.00	n/a	n/a
	R	0.00 ± 0.00	0.13 ± 0.07	2.0323	0.1759	0.00 ± 0.00	0.00 ± 0.00	n/a	n/a

[a] Results of ANVOA. If Prob.>F is less than 0.05, then the difference between edge orientation is considered significant at the 95% confidence level. [b]Conifer seedlings. [c]Deciduous seedlings. [d]Shade-tolerant herbs. [e]Shade-intolerant herbs. [f]Acrocarpous mosses. [g]Pleurocarpous mosses. [h]Sphagnum mosses.

Table 7. Mean (± SEM) life form abundance (A) and richness (R) at east versus west-facing fire edges and comparable distance in reference forest. An ANOVA with plots nested within edge orientation was used to test the effect of orientation (refer to methods for more detail).

Life form		Reference				Fire			
		East-Facing (n = 48)	West-Facing (n =48)	F-ratio	Sig. (Prob.>F[a])	East-Facing (n = 48)	West-Facing (n = 48)	F-ratio	Sig. (Prob.>F[a])
Conifer[b]	A	7.50 ± 2.19	7.48 ± 3.33	0.0183	0.9968	7.08 ± 2.18	2.29 ± 0.64	3.5778	0.0794
	R	0.54 ± 0.15	0.50 ± 0.17	0.0000	0.8944	0.67 ± 0.18	0.42 ± 0.12	1.3404	0.2663
Deciduous[c]	A	0.00 ± 0.00	1.19 ± 0.85	4.2000	0.1677	2.71 ± 1.34	4.38 ± 1.54	0.9218	0.3533
	R	0.00 ± 0.00	0.13 ± 0.07	2.1170	0.0596	0.29 ± 0.13	0.38 ± 0.10	0.3590	0.5586
Tall shrubs	A	20.44 ± 3.85	19.10 ± 3.80	0.3164	0.8395	30.17 ± 7.25	21.65 ± 3.83	0.8338	0.3766
	R	1.42 ± 0.22	1.75 ± 0.35	0.4260	0.5827	1.83 ± 0.26	1.71 ± 0.19	0.0796	0.7819
Low shrubs	A	27.21 ± 3.27	26.63 ± 4.27	0.4172	0.9377	16.96 ± 1.90	17.60 ± 2.57	0.0225	0.8830
	R	3.88 ± 0.41	3.17 ± 0.38	0.0063	0.4172	2.00 ± 0.17	2.04 ± 0.15	0.0326	0.8594
ST herbs[d]	A	20.56 ± 2.27	19.23 ± 2.65	1.2190	0.2882	18.04 ± 2.35	24.65 ± 3.79	1.6746	0.2166
	R	3.54 ± 0.35	2.88 ± 0.39	0.1340	0.7198	2.71 ± 0.34	2.63 ± 0.33	0.0190	0.8922
SIT herbs[e]	A	11.58 ± 2.30	14.85 ± 3.93	0.8248	0.8248	22.46 ± 4.13	26.56 ± 5.19	0.1527	0.7018
	R	2.33 ± 0.41	2.13 ± 0.54	0.2470	0.6269	3.08 ± 0.48	3.50 ± 0.61	0.7262	0.7262
Grasses	A	6.67 ± 2.44	4.04 ± 1.29	0.9787	0.3393	5.60 ± 5.27	5.60 ± 5.27	0.0365	0.8513
	R	0.71 ± 0.18	0.58 ± 0.13	0.2593	0.6186	1.00 ± 0.16	1.04 ± 0.15	0.0283	0.8687
Sedges	A	3.42 ± 1.21	2.23 ± 1.29	0.2532	0.6227	4.44 ± 1.37	4.44 ± 1.37	1.5688	0.2309
	R	0.50 ± 0.16	0.25 ± 0.11	1.2353	0.2851	0.71 ± 0.20	1.08 ± 0.27	0.7061	0.4149
Horsetails	A	2.31 ± 0.70	3.65 ± 1.26	0.9300	0.3512	0.00 ± 0.00	0.00 ± 0.00	n/a	n/a
	R	0.38 ± 0.10	0.42 ± 0.12	0.0476	0.8304	0.00 ± 0.00	0.00 ± 0.00	n/a	n/a
Club mosses	A	4.33 ± 1.39	8.96 ± 2.17	1.3736	0.2608	3.50 ± 0.67	3.50 ± 0.67	0.7504	0.4010
	R	0.58 ± 0.15	1.00 ± 0.22	1.3736	0.2608	0.58 ± 0.10	0.29 ± 0.09	4.1325	0.0615
Ferns	A	3.31 ± 1.45	2.08 ± 1.39	0.2430	0.6297	3.75 ± 1.24	3.75 ± 1.24	3.7607	0.0729
	R	0.54 ± 0.54	0.38 ± 0.17	0.1824	0.6758	0.50 ± 0.15	0.79 ± 0.16	1.1320	0.3054
Bryophytes	A	45.83 ± 6.22	51.10 ± 6.12	0.3634	0.5563	23.67 ± 2.94	23.67 ± 2.94	0.9890	0.3369
	R	5.50 ± 0.81	5.33 ± 0.60	0.0333	0.8577	3.08 ± 0.42	3.33 ± 0.46	0.1680	0.6881
Acrocarpous[f]	A	14.94 ± 3.78	11.60 ± 3.47	0.3986	0.5380	19.67 ± 2.51	19.67 ± 2.51	2.7189	0.1214
	R	2.79 ± 0.54	2.38 ± 0.49	0.0137	0.9085	2.29 ± 0.23	2.63 ± 0.31	1.1256	0.3067
Pleurocarpous[g]	A	28.71 ± 4.43	34.35 ± 5.31	0.5704	0.4626	3.92 ± 1.82	3.92 ± 1.82	0.9534	0.3454
	R	2.50 ± 0.39	2.46 ± 0.29	0.0137	0.9085	0.75 ± 0.31	0.63 ± 0.18	0.1349	0.7189
Sphagnum[h]	A	2.19 ± 1.34	5.15 ± 2.37	1.0972	0.3126	0.08 ± 0.08	0.08 ± 0.08	0.6117	0.4472
	R	0.21 ± 0.10	0.10 ± 0.21	1.1951	0.2928	0.04 ± 0.04	0.08 ± 0.06	0.3684	0.5536

[a] Results of ANVOA. If Prob.>F is less than 0.05, then the difference between edge orientation is considered significant at the 95% confidence level. [b]Conifer seedlings. [c]Deciduous seedlings. [d]Shade-tolerant herbs. [e]Shade-intolerant herbs. [f]Acrocarpous mosses. [g]Pleurocarpous mosses. [h]Sphagnum mosses.

Edge Structure

Structure of buffer edges versus reference forest

At the buffer edge, there was 32.5 and 20.3% lower live stem density (Table 8) and basal area respectively (Figure 7a). Tree canopy cover, windthrow and mortality significantly differed at buffer edges compared to reference forest (Figure 7b). Canopy cover was 37.8% lower due to high at mortality buffer edges. This increase in mortality was primarily caused by windthrow which was 29.6% above reference forest (Figure 7b).

Structure of fire edges versus reference ecotonal edge

Wild-fire created a significant edge influence for all measured parameters (Figure 7c & d; Table 8). The fire edge had a 75.6% less live stem density and 42.6% increase in snag density compared to same location in reference forest. Furthermore, canopy tree mortality at the fire edge was more than seven times higher than the reference (Table 8).
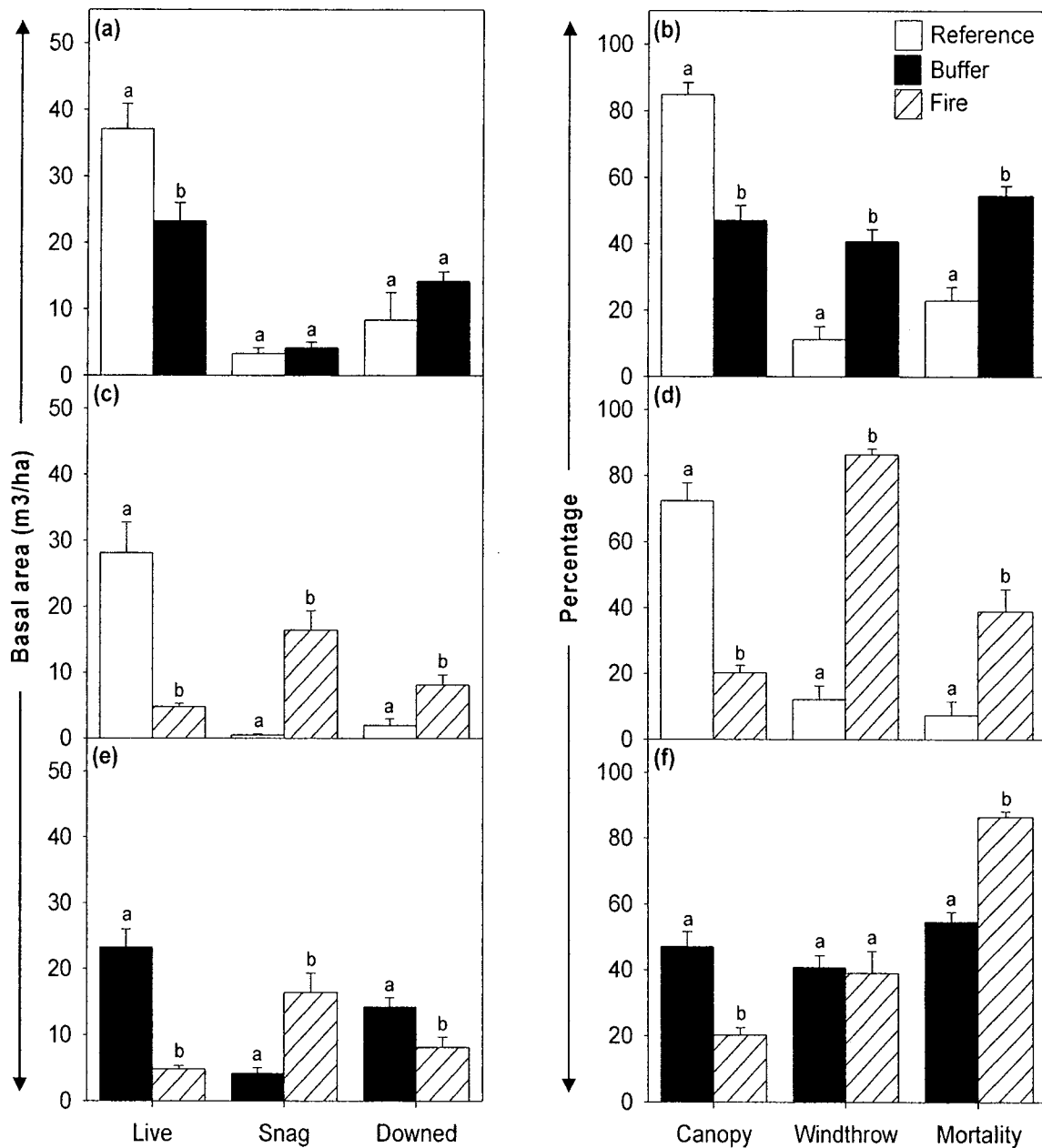
Figure 7. Mean (±SEM.) for basal area (m²/ha), canopy cover (% covered sky), windthrow (%) and mortality (%) at buffer and fire edges, as well as at comparable distances within reference forest. Significant differences (p≤0.05) indentified by Tukey HSD post hoc results are indicated by differing letters (refer to Table 8 for exact values).

Table 8. Mean (± SEM) structural attributes of canopy trees at clearcut and fire edges and corresponding distance from stream of interior reference forest. Significance of edge influence (SEI) is indicates if a response variable at the edge plots is significantly different than its corresponding reference value when p <0.05.

Attribute	Reference* (n = 16)	Buffer (n = 16)	F-ratio	SEI (Prob>F^a)	Reference** (n = 16)	Fire (n = 16)	F-ratio	SEI (Prob.>F^a)
Live SD^b	1737.50 ± 222.65	950.00 ± 90.37	10.7410	**0.0027**	1500.00 ± 208.97	225.00 ± 25.00	36.7030	**>0.0001**
Snag SD^b	225.00 ± 70.42	325.00 ± 47.87	7.2314	**0.0414**	87.50 ± 40.70	825.00 ± 126.33	30.8780	**>0.0001**
Downed SD^b	250.00 ± 90.37	837.50 ± 77.93	24.2390	**>0.0001**	87.50 ± 36.37	675.00 ± 126.33	19.9780	**>0.0001**
Live BA^c	37.10 ± 3.76	23.29 ± 2.71	8.8596	**0.0057**	28.12 ± 4.64	4.78 ± 0.53	24.9090	**>0.0001**
Snag BA^c	3.28 ± 0.91	4.19 ± 0.84	0.5389	0.4686	0.46 ± 0.22	16.46 ± 2.97	28.5900	**>0.0001**
Downed BA^c	8.37 ± 4.15	14.25 ± 1.41	1.7976	0.1901	1.97 ± 1.06	8.11 ± 1.57	10.4870	**0.0029**
Windthrow (%)^d	11.21 ±3.90	40.82 ± 3.53	31.6810	**>0.0001**	12.10 ± 4.24	86.45 ± 1.69	15.9330	**>0.0001**
Mortality (%)^d	23.00 ± 4.09	54.58 ± 2.89	39.6810	**>0.0001**	7.36 ± 4.23	38.97 ± 6.70	264.9900	**0.0004**
Canopy (%)^d	84.89 ± 3.60	47.11 ± 4.48	43.1560	**>0.0001**	72.36 ± 5.51	20.23 ± 2.28	76.4820	**>0.0001**

^{a}ANOVA results. If Prob.>F is less than 0.05, then the difference between edge and reference plots is considered significant at the 95% confidence level.
^{b} SD = stem density (stems/ha). Log transformed for analysis.
^{c} BA = basal area (m^2/ha). ln transformed for analysis.
^{d} Raw data (untransformed) used for analysis.
*Reference mean values at the corresponding distance of the clearcut edge.
**Reference mean values at the corresponding distance of the fire edge.

Buffer versus fire edge structure

Clearcutting and natural fire induced vast structural dissimilarities between the edges. The buffer edges had 33.1% higher live stem densities (Table 9) and consequently a significantly higher basal area (Figure 7e). Conversely, fire edges had much higher density of snags with a 46.7% greater in snag basal area (Figure 7e). Although downed stem basal areas differed at the two edges (Figure 7e), there was no significant difference in downed stem density between fire and clearcut edges (Table 9). There was no detectable difference of windthrow at the two edges. However, there was 1.6 times higher tree mortality and a 26.9% lower canopy cover (Figure 7f) at the fire edges compared to clearcut edges.

Table 9. Mean (± SEM) structural attributes of mature trees at buffer and fire edge. The p-value indicates if the response variables in buffer edge plots are significantly different than the fire edge plots.

Attribute	Buffer (n = 16)	Fire (n = 16)	F-ratio	p-value (Prob.>F[a])
Live stem density(stems/ha)[b]	950.00 ± 90.37	225.00 ± 25.00	59.7870	**0.0011**
Snag stem density (stems/ha)[b]	275.00 ± 47.87	825.00 ± 126.33	16.5750	**0.0003**
Downed stem density (stems/ha)[b]	837.50 ± 77.93	675.00 ± 126.33	1.1986	0.2823
Live basal area (m²/ha)[c]	23.29 ± 2.71	4.78 ± 0.53	44.8060	**0.0023**
Snag basal area (m²/ha)[c]	4.19 ± 0.84	16.46 ± 2.97	15.6440	**0.0004**
Downed basal area (m²/ha)	14.25 ± 1.41	8.11 ± 1.57	8.4286	**0.0069**
Windthrow (%)[d]	40.82 ± 3.53	38.97 ± 6.70	0.0596	0.8087
Mortality (%)[d]	54.58 ± 2.89	86.45 ± 1.69	90.4880	**>0.0001**
Canopy Cover (%)[d]	47.11 ± 4.48	20.23 ± 2.28	28.5560	**>0.0001**

[a] Nested ANOVA results for [b]log, [c]ln, and [d]untransformed data. If Prob.>F is less than 0.05, then the difference between edges is considered significant at the 95% confidence level.

Near ground microenvironment at the edges

Soil temperature was significantly higher at buffer edges compared to the same location in reference forest (Figure 8a). Buffer edges had an average of 7.4 m^3/m^3 less soil moisture compared the reference forest (Figure 8a). Fire edge had significantly higher soil moisture compared to the reference forest and buffer edges (32.2 and 50.6% respectively, Figure 8b & c). The clearcut and fire edges were had a significantly higher vapour pressure deficit than the reference forest (Figure 8a & b), but did not differ from each other (Figure 8c). The increased vapour pressure deficit is presumed to be a direct result of increased exposure from canopy removal. Organic matter depth was significantly lower at the fire edge than the reference forest (Figure 8b). Also organic matter depth was significantly higher (average of 2.65 cm) at buffer edges than fire edges (Figure 8c).
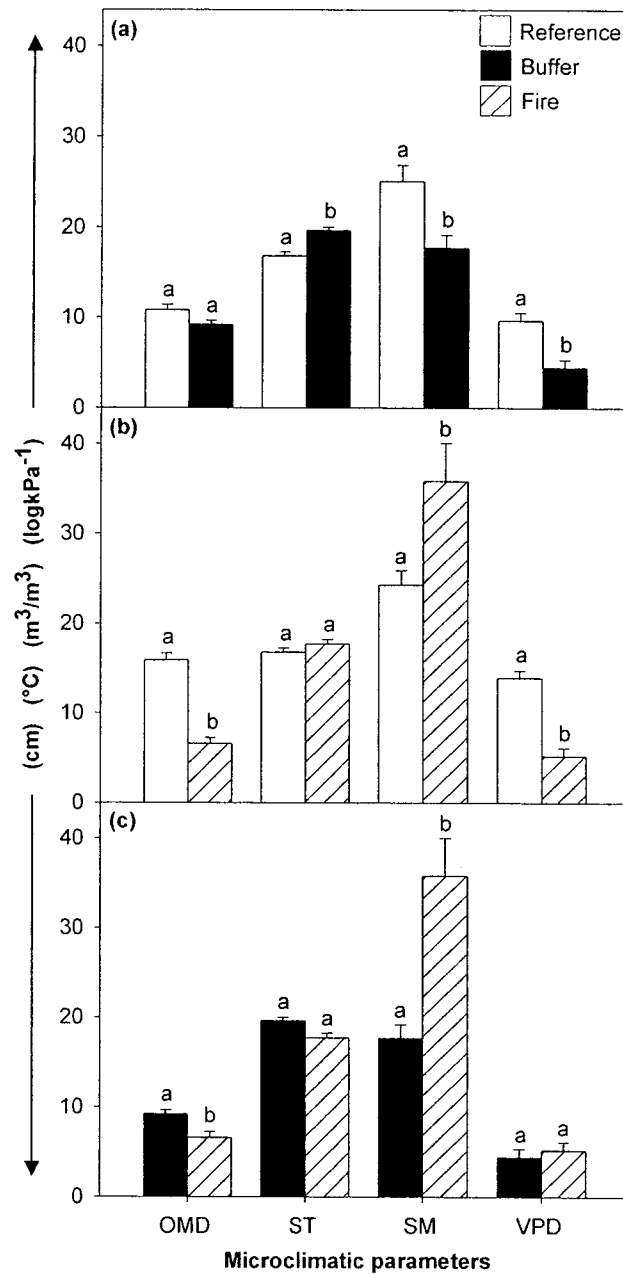
Figure 8. Mean (±SEM) of organic matter depth (OMD, cm), soil temperature (ST, °C), soil moisture (SM, m^3/m^3), and vapour pressure deficit (VPD, kPa) at; (a) clearcut edges versus reference forest, (b) fire edge vs. reference forest, and (c) clearcut versus fire edges. Significant differences ($p \leq 0.05$) are indicated by unlike letters (refer to table 9 for exact values). Note: the inverse log transformation of VPD is displayed.

Species composition at the edges

Buffer and fire edges versus reference forest

Richness of deciduous seedlings, tall shrubs, shade-intolerant herbs, grasses and sedges richness and abundance were significantly higher at the buffer edge compared to the same locations in reference forest (Figure 9a). However, the abundance and richness of conifer seedlings and pleurocarpous mosses was significant lower at buffer edges. Fewer life forms showed a significant difference at the fire edges (Figure 9b). Similar to the buffer edges, sphagnum mosses and pleurocarpous mosses had a significant decrease in abundance and richness at the fire edges. The deciduous saplings and acrocarpous mosses had increased abundance at the fire edges than reference forest (Figure 9b)

Buffer versus fire edges

Compared to fire edges, buffer edges had more than three times higher abundance of deciduous saplings, low shrubs and pleurocarpous mosses (Table 10). But the fire edges had more than three times higher abundance of conifer saplings and apocarpous mosses, as well as 13 times higher abundance of ferns (Table 10).
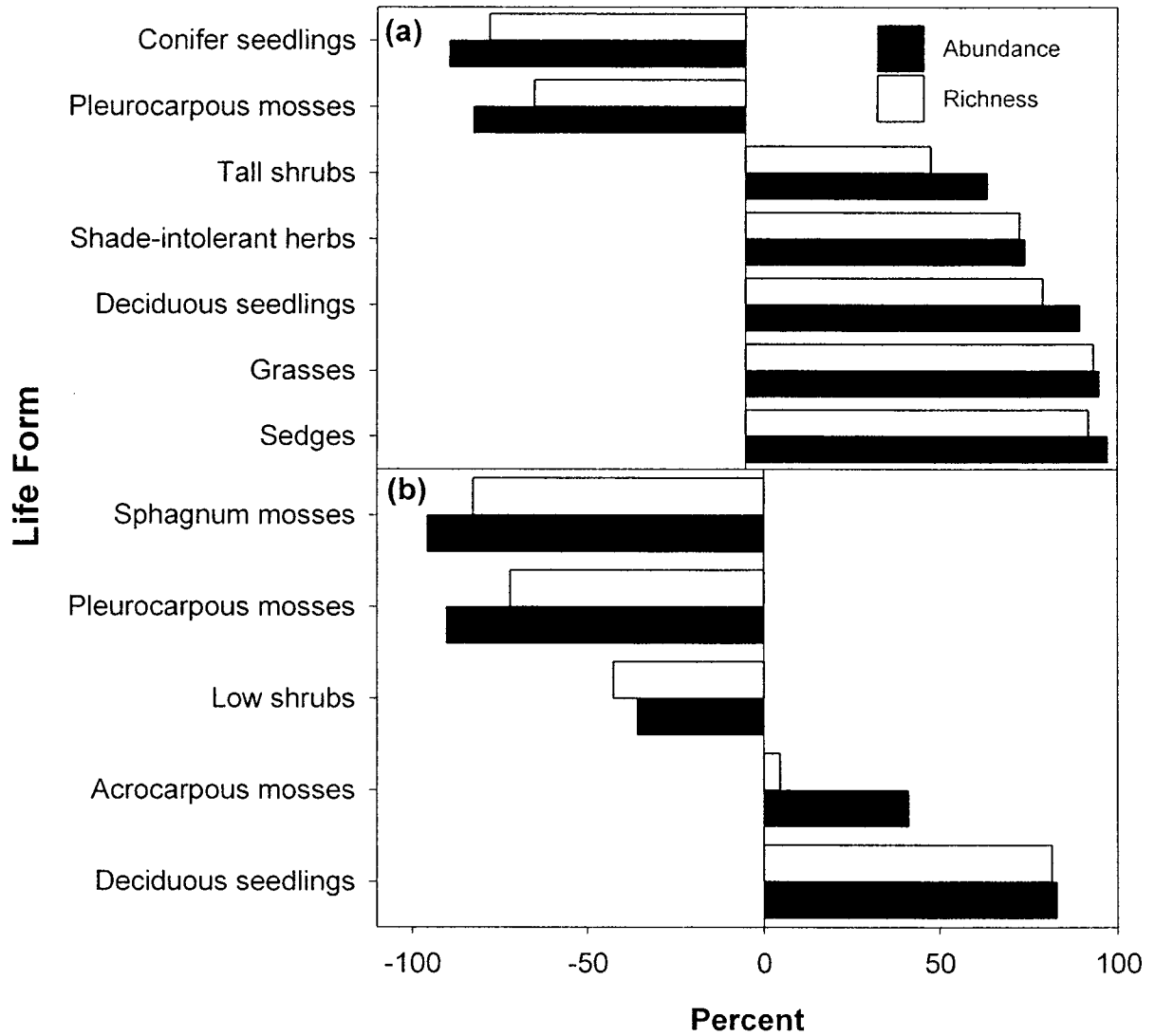
Figure 9. The percent difference in abundance and richness of life forms relative to reference forest averages at (a) clearcut edges, and (b) fire edges. Only life forms determined to be significantly different than interior reference forest are shown. No error bars are displayed because these values were calculated from the absolute mean values at the edges and comparable distances within reference forest.

Table 10. Mean abundance and richness (±SEM) of life forms at clearcut and fire edges. The p-value indicates if the response variable in buffer edge plots are significantly different than the fire edge plots.

Life form	Abundance				Richness			
	Buffer (n = 96)	Fire (n=96)	F-ratio	p-value (Prob.>F[a])	Buffer (n = 96)	Fire (n=96)	F-ratio	p-value (Prob.>F[a])
Conifer saplings	1.51 ± 0.77	4.69 ± 1.18	4.0233	**0.0540**	0.17 ± 0.05	0.54 ± 0.11	9.1124	**0.0054**
Deciduous saplings	14.45 ± 2.74	3.54 ± 1.01	5.6870	**0.0236**	0.96 ± 0.13	0.33 ± 0.08	7.2230	**0.0073**
Tall shrubs	31.31 ± 3.01	25.91 ± 4.10	0.7937	0.3801	2.19 ± 0.16	1.77 ± 0.16	2.1383	0.1541
Low shrubs	47.01 ± 2.89	17.28 ± 1.58	49.2980	**0.0001**	4.35 ± 0.20	2.02 ± 0.11	69.2270	**0.0001**
Shade-tolerant herbs	27.27 ± 2.02	21.34 ± 2.26	10.6729	**0.0125**	3.56 ± 0.19	2.67 ± 0.24	5.7506	**0.0229**
Shade-intolerant herbs	23.39 ± 1.84	24.51 ± 3.29	0.0400	0.8428	2.77 ± 0.25	3.29 ± 0.39	0.5795	0.4525
Grasses	7.35 ± 1.31	5.74 ± 0.70	1.0056	0.3240	0.92 ± 0.13	1.02 ± 0.11	0.3086	0.5826
Sedges	5.13 ± 1.74	7.55 ± 1.76	0.5809	0.4519	0.56 ± 0.14	0.90 ± 0.17	1.5496	0.2228
Horsetails	0.14 ± 0.08	0.00 ± 0.00	3.2710	0.0805	0.06 ± 0.04	0.00 ± 0.00	3.4615	0.0726
Club mosses	5.34 ± 0.84	3.02 ± 0.62	3.8784	0.0582	1.02 ± 0.14	0.44 ± 0.07	6.7820	0.0142
Ferns	0.50 ± 0.21	6.82 ± 1.32	13.1720	**0.0010**	0.13 ± 0.05	0.65 ± 0.11	11.5600	**0.0019**
Acrocarpous mosses	6.28 ± 1.06	22.61 ± 1.81	44.2490	**>0.0001**	1.35 ± 0.18	2.46 ± 0.19	14.4500	**0.0007**
Pleurocarpous mosses	9.60 ± 2.08	3.04 ± 0.98	4.2650	**0.0476**	1.10 ± 0.16	0.69 ± 0.18	5.5974	**0.0175**
Sphagnum mosses	0.00 ± 0.00	0.15 ± 0.08	3.4186	0.0743	0.00 ± 0.00	0.06 ± 0.04	3.4615	0.0726

[a]Results of ANOVA. If the Prob>F is less than 0.05, then the difference between clearcut and fire edges is considered significant at the 95% confidence interval.

Plant communities at buffer and fire edges

MRPP Analysis

The first null hypothesis of no difference in understory species composition between the microhabitats of the site types can be rejected because MRPP showed significant differences for each site type (all p-values ≤ 0.05; Table 11). For the most part, within-group homogeneity was very low (A < 0.1), with the exception of comparisons of the fire sites at and beyond the upland microhabitat (Table 11). This indicates that species composition at the fire sites is fairly uniform at the upland location. Overall, the fire sites showed the largest separation from the reference forest and buffer across the majority of the locations (Figure 10). Also, the highest within-group homogeneity (A = 0.8877) was found between the buffer and fire sites at the clearcut/fire location indicating that species composition in the buffer sites is also fairly uniform at this location. Although the buffer treatment was significantly different than the reference forest locations (p $\leq$ 0.05) the degree of separation was far less than that of the fire and reference forest (Figure 10).
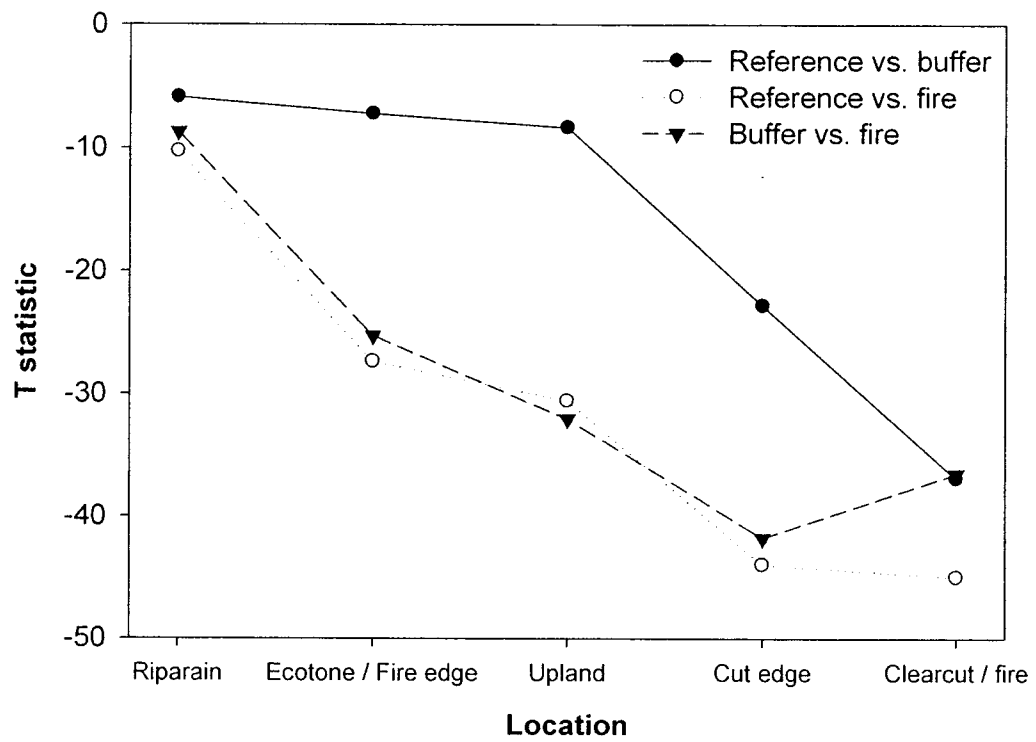
Figure 10. Scatter plot of the site type comparison by T-statistic at the riparian, ecotone/fire edge, upland, clearcut edge and clearcut/burned forest. See Table 3 for description of the calculation procedure.

Table 11. Fifteen separate MRPP analyses show a significant difference in understory species composition at different microhabitats for the site types. The observed delta was determined from the data and the expected delta was obtained from the null distribution. The chance-corrected within group agreement (A) indicates within-group homogeneity compared to random expectation. The p value indicates the probability of having a more extreme observed delta and is significant at < 0.05.

Microhabitat	Comparison	n	# Species	Observed delta	Expected delta	MRPP Statistics T	A	p
Riparian	Reference vs. buffer	64	171	72.9751	74.4686	-5.9066	0.0201	0.0001
	Reference vs. fire	64	162	73.8021	76.3699	-10.3132	0.0336	< 0.0001
	Buffer vs. fire	64	160	68.0289	70.0133	-8.7355	0.0283	< 0.0001
Ecotone / Fire edge	Reference vs. buffer	96	123	73.1977	75.4371	-7.2290	0.0297	0.0001
	Reference vs. fire	96	127	72.5273	81.7742	-27.3832	0.1131	< 0.0001
	Buffer vs. fire	96	117	73.5538	81.7091	-25.2754	0.0998	< 0.0001
Upland	Reference vs. buffer	64	109	69.8123	72.0530	-8.3433	0.0311	< 0.0001
	Reference vs. fire	64	103	67.3573	78.1469	-30.5870	0.1381	< 0.0001
	Buffer vs. fire	64	111	65.1651	76.3461	-32.0759	0.1465	< 0.0001
Cut edge	Reference vs. buffer	96	129	69.4673	73.1770	-22.8281	0.0507	< 0.0001
	Reference vs. fire	96	119	70.0988	80.0553	-43.9838	0.1244	< 0.0001
	Buffer vs. fire	96	121	67.3744	75.7918	-41.7875	0.1111	< 0.0001
Cut over Fire	Reference vs. buffer	64	123	69.7957	77.0752	-36.9370	0.0944	< 0.0001
	Reference vs. fire	64	108	70.2235	80.2515	-44.9923	0.1250	< 0.0001
	Buffer vs. fire	64	116	68.4550	75.1238	-36.5210	0.8877	< 0.0001

The second null hypothesis of no difference in composition of life forms between clearcut and fire edges can also be rejected because the majority of the p-values for life forms were < 0.05 (Table 12). The within-group homogeneity was very low (i.e. A < 0.1) for all life forms expect the pleurocarpous mosses (A = 0.1437. Conifer saplings were the only life form not to significantly differ at the buffer and fire edges.

Table 12. Eleven separate MRPP analyses showed significant difference between buffer and fire edges for all life forms, except conifer saplings. Refer to Table 11 for details regarding MRPP.

Life form	n	# species	MRPP Statistic				
			Observed delta	Expected delta	T	A	p
Conifer sapling	25	5	0.7196	0.7389	-1.0172	0.0261	0.1407
Hardwood sapling	37	4	0.5822	0.7390	-13.2246	0.0122	< 0.0001
Tall shrubs	84	15	0.7938	0.8294	-10.1438	0.0428	< 0.0001
Low shrubs	94	15	0.6164	0.6932	-28.9012	0.1108	< 0.0001
Shade-tolerant herbs	93	12	0.6326	0.6841	-19.0662	0.0753	< 0.0001
Shade-intolerant herbs	89	24	0.7078	0.7696	-19.1605	0.0802	< 0.0001
Grasses	59	7	0.6913	0.7557	-9.9985	0.0853	< 0.0001
Sedges	28	10	0.6807	0.7459	-6.9834	0.0874	0.0001
Ferns & allies	64	12	0.8179	0.8588	-7.9068	0.0476	< 0.0001
Acrocarpous mosses	75	19	0.7289	0.8086	-20.9091	0.0985	< 0.0001
Pleurocarpous mosses	62	13	0.6880	0.8034	-21.8636	0.1437	< 0.0001

Indicator Species Analysis

Within each site type at least one species was identified as an indicator of a microhabitat (Table 13). The clearcut/fire matrix (microhabitat 5) of the buffer sites had a relatively high number of weedy shade-intolerant herbs (*Aster ciliolatus*, *Hieracium* spp. and *Solidago canadensis*), as well as deciduous saplings (*Betula papyrifera*, *Populus tremuloides and Populus balsamifera*). Grasses (*Poa compressa* and *P. pratensis*) were the most common indicators at the clearcut edges (microhabitat 4) of the buffers. The upland location (microhabitat 3) of the buffers

Table 13. Indicator species at different microhabitats for each treatment. Relative frequency and abundance of each species within each treatment is summarized by the observed indicator value (IV). Complete listing of indicator species and significance levels can be found in Appendix V.

Microhabitat	Reference			Buffer			Fire		
	Indicator Species	Life form[a]	IV[b]	Indicator Species	Life form[a]	IV[b]	Indicator Species	Life form[a]	IV[b]
Riparian (1)	*Rhizomnium pseudopunctatum*	AM	53.6	*Thuidium delicatulum*	PM	74.1	*Thuidium delicatulum*	PM	89.2
	Lycopus uniflorus	SIH	52.3	*Alnus incana*	TS	72.5	*Thalictrum dayscarpum*	SIH	74.5
	Mnium marginatum	AM	52.2	*Aulocomnium palustre*	AM	71.9	*Viola septentrionalis*	STH	62.5
	Fissidens spp.	PM	50.0	*Rhizomnium punctatum*	AM	71.8	*Scorpidium scorpioides*	AM	59.4
	Thalictrum dayscarpum	SIH	49.6	*Mnium marginatum*	AM	68.7	*Athyrium felix-femina*	F	58.2
Ecotone/ Fire edge (2)	*Pleurozium schreberi*	PM	24.1	*Lycopodium annotinum*	CM	CM	*Trientalis borealis*	STH	17.1
	Dicranum ontariense	AM	19.3	*Trientalis borealis*	STH	22.8	*Polytrichum juniperinum*	AM	16.2
	Galium triflorum	SIH	18.2	*Maianthemum canadense*	STH	22.5	*Carex bebbii*	SD	12.2
	Fragaria virginiana	SIH	12.8	*Coptis trifolia*	STH	21.0	*Corylus cornuta*	TS	10.2
	Epilobium angustifolium	SIH	9.2	*Petasites frigidus*	SIH	18.1	*Dierbilla lonicera*	LS	12.3
Upland (3)	*Dicranum fuscescens*	AM	9.4	*Pleurozium schreberi*	PM	35.0	*Vaccinium angustifolium*	LS	30.3
				Ptilium crista-castrensis	PM	31.0	*Ceratodon purpureus*	AM	27.7
				Cornus canadensis	STH	24.1	*Polytrichum commune*	AM	26.0
				Aralia nudicaulis	STH	21.8	*Conyza canadensis*	SIH	19.6
							Corydalis sempravirens	SIH	18.4
Cut edge (4)	*Diervilla lonicera*	LS	17.0	*Diervilla lonicera*	LS	29.6	*Epilobium angustifolium*	SIH	30.1
				Corylus cornuta	TS	20.6	*Rubus idaeus*	LS	18.8
				Poa compressa	G	13.4	*Lycopodium dendroideum*	CM	16.7
				Poa pratensis	G	7.9	*Picea mariana*	CS	14.6
				Scirpus atrovirens	SD	6.2			
Clearcut/fire matrix (5)	*Populus tremuloides*	HWS	11.6	*Populus tremuloides*	HWS	34.6	*Aster macrophyllus*	SIH	20.3
				Rubus idaeus	LS	32.3	*Rosa acicularis*	LS	11.2
				Vaccinium angustifolium	LS	29.7	*Polygonum cilinode*	SIH	10.6
				Epilobium angustifolium	SIH	28.7	*Populus tremuloides*	HWS	10.2
				Aster macrophyllus	SIH	26.6	*Hieracium caespitosum*	SIH	9.8

[a] Species were divided based on life form; TS = tall shrub, LS = low shrub, STH = shade intolerant herb, SIH = shade tolerant herb, G = grass, SD = sedge, F = fem, CM = club-moss, AM = acrocarpous moss, PM = pleurocarpous moss. [b] IV = indicator value.

had indicators of shade-tolerant understory herbs (*Aralia nudicaulis* and *Cornus canadensis*) as well as the pleurocarpous mosses (*Pleurozium schreberi* and *Ptilium crista-castrensis*). The ecotonal/fire edge location (microhabitat 2) of the buffers were primarily composed species indicative of high moisture (*Anemone canadensis*, *Petasites frigidus* and *Trientalis borealis*).

The clearcut and upland locations (microhabitats 3, 4 &5) of the fire sites had species indicative of fire (*Ceratondon purpureus*, *Epilobium angustifolium*, *Polytrichum commune*, and *Vaccinium myrtilloides*) and shade-intolerant species (*Carex* spp., *Polygonum cilinode, Prunus virginiana*). The ecotonal/fire edge (microhabitat 2) of the fire sites had species indicative of disturbance (*Conyza canadensis*, *Polytrichum juniperinum*, and *Taraxacum borealis*), as well as indicators of high moisture (*Aster puniceus*, and *Carex bebbii*).

The largest number of indicator species among all three site types occurred within the riparian zone (microhabitat 1) (Appendix VI). The high occurrence of similar species within the riparian zone can be attributed to the high species evenness of riparian zones. *Picea mariana* was the only species shared across all three site types within the clearcut/fire matrix location (microhabitat 5). Also, within the clearcut/fire matrix (microhabitat 5) the buffer and fire sitess shared many species, namely those adapted to disturbance (*Aster macrophyllus*, *Epilobium angustifolium, Hieracium aurantiacum* and *Rubus idaeus*). The edge locations (i.e. region 2 within fire treatment and region 4 within buffer treatment) shared two species; *Corylus cornuta* and *Diervilla lonicera*.

Depth of edge influence (DEI)

Structural DEI

At the clearcut edges, live and snag basal areas had a DEI of 15 m (Figure 11a & b), followed by 10 m for downed tree basal area and mortality (Figure 11c & f). Canopy cover and windthrow had a DEI of 5 m (Figure 11d & e), which was the lowest of all the structural parameters at the clearcut edges. All structural parameters were found to significantly differ across the entire area sampled for fire disturbance (Figure 11a – f).

Microclimatic DEI

Soil moisture at the clearcut edges was significantly lower than the reference forest up to 20 m. Conversely, at the fire edge the soil moisture was significantly higher for 2 m (Figure 12a). Within the buffer soil temperature was significantly higher for 8 m, while no significant DEI was found at the fire edge (Figure 12b). Organic matter depth did not differ at any locations within the buffer, but was significantly lower for 6 m from the fire edge (Figure 12c). Vapour pressure deficit (VPD) was significantly higher for both disturbance types. VPD had DEI of 24 and 2 m for the clearcut and fire edge respectively (Figure 12d).
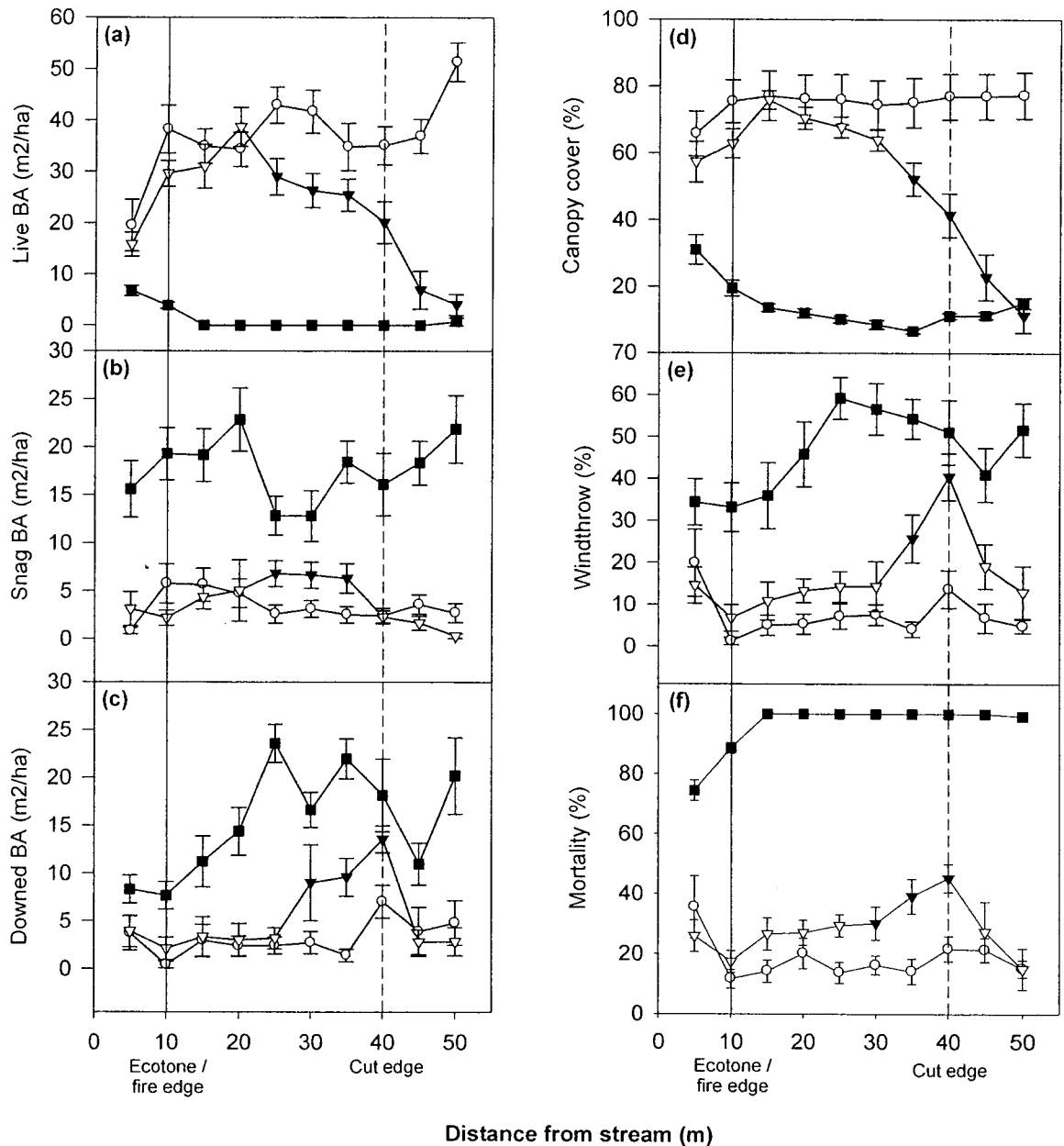
Figure 11. Mean (± SEM) of structural variables along gradients starting at stream edge, (a) live tree basal area (m²/ha), (b) snag basal area (m²/ha), (c) downed basal area (m²/ha), (d) canopy cover (%), (e) windthrow (%), and (f) mortality (%). Symbols represent; reference forest (circles), buffer (triangles), and fire (squares). Filled symbols indicate values that are significantly different from reference forest (see Methods). Solid line at 10 m represents the average burn edge/ecotone location, and the dashed line at 40 m represents the average cut edge location.
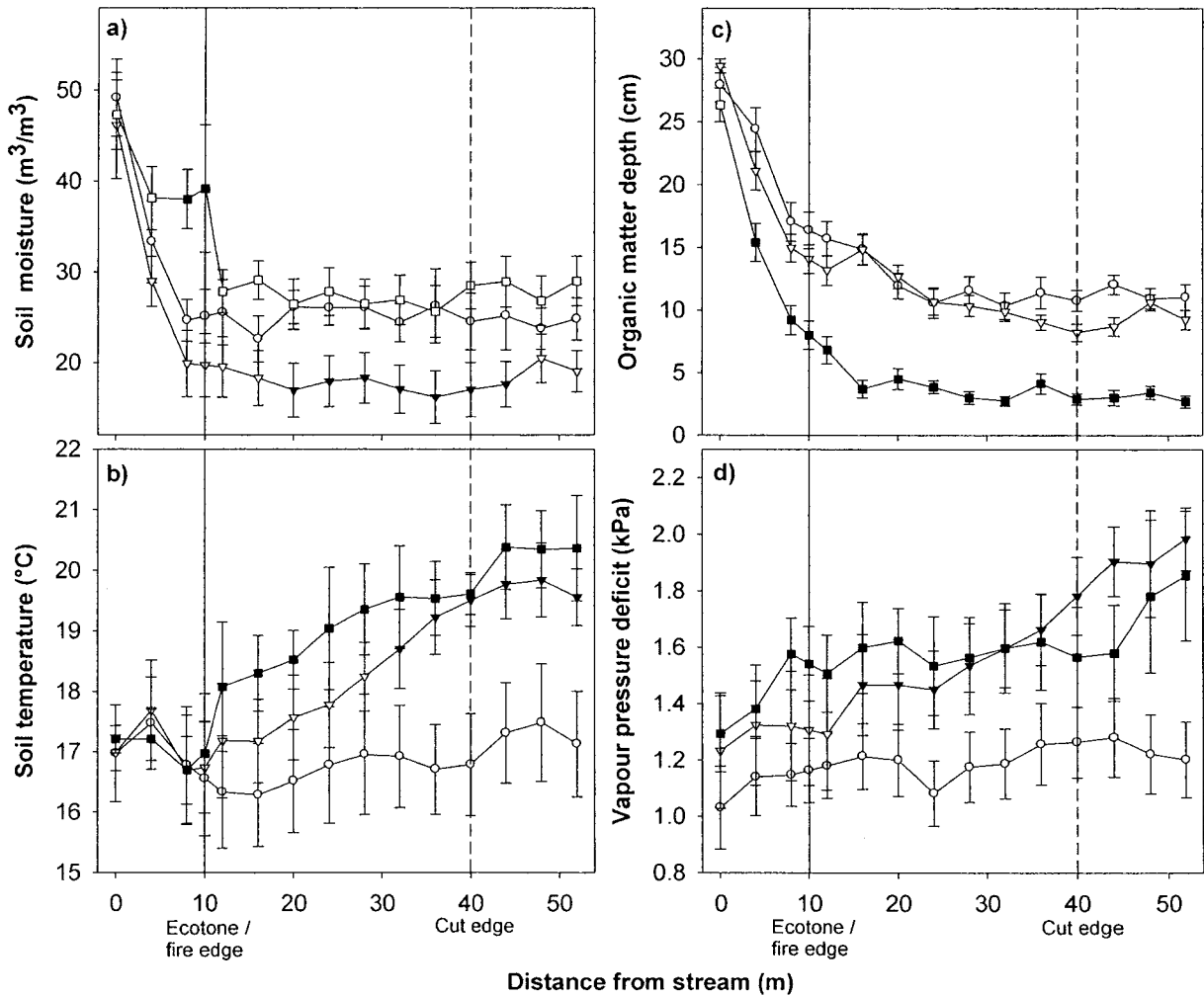
Figure 12. Mean values (± SEM) of microclimatic variables along gradients starting at steam edge, (a) soil moisture (m³/m³), (b) soil temperature (°C), (c) organic matter depth (cm), and (d) vapour pressure deficit (kPa). Symbols represent reference forest (circles), buffer (triangles), and fire (squares). Filled symbols indicate values that are significantly different than reference forest condition (see Methods). Solid lines at 10 m represent the average burn edge/ecotone location and the dashed lines at 40 m represent the average cut edge location.

Understory species cover and richness DEI

Clearcut edge understory species cover and richness was significantly higher than reference forest (Figure 13). At fire edges understory species richness did not differ significantly from either reference forest or the clear cut edges. However, clearcut edges had a significantly higher understory species cover than fire edges. Also, higher species cover was found in the clearcut area compared to similar locations in the burned area.

No significant DEI was found in understory cover and richness for fire edges. All three site types followed the same general trend with a marked increase of understory cover and richness from ecotonal edge locations towards the stream. Conversely, the clearcut edges had significantly higher understory cover and richness for 8 m past the edge location.

Species functional groups DEI

The depth of edge influence was generally greater at clearcut edges compared to fire edges (Figure 14). Tall and low shrub abundances were significantly higher for approximately 15 m into the buffers from clearcut edges. However, only the low shrubs showed a difference at fire edges with lower abundance and richness for 7 m past the edge. Shade intolerant herbs had abundance and richness for 11 and 15 m into the buffers respectively. Similarly, shade-intolerant herbs had higher abundance and richness at fire edges, however the changes were only detected at 3 to 4 m. The graminoids had higher abundance and richness for 5 to 8 m past buffer edges. Graminoid abundance was higher for 2 to 3 m past fire edges and richness remained unchanged. Pleurocarpous mosses showed significantly lower richness for about 5 m into the buffers while changes in species richness were detected for 10 m (Figure 14).
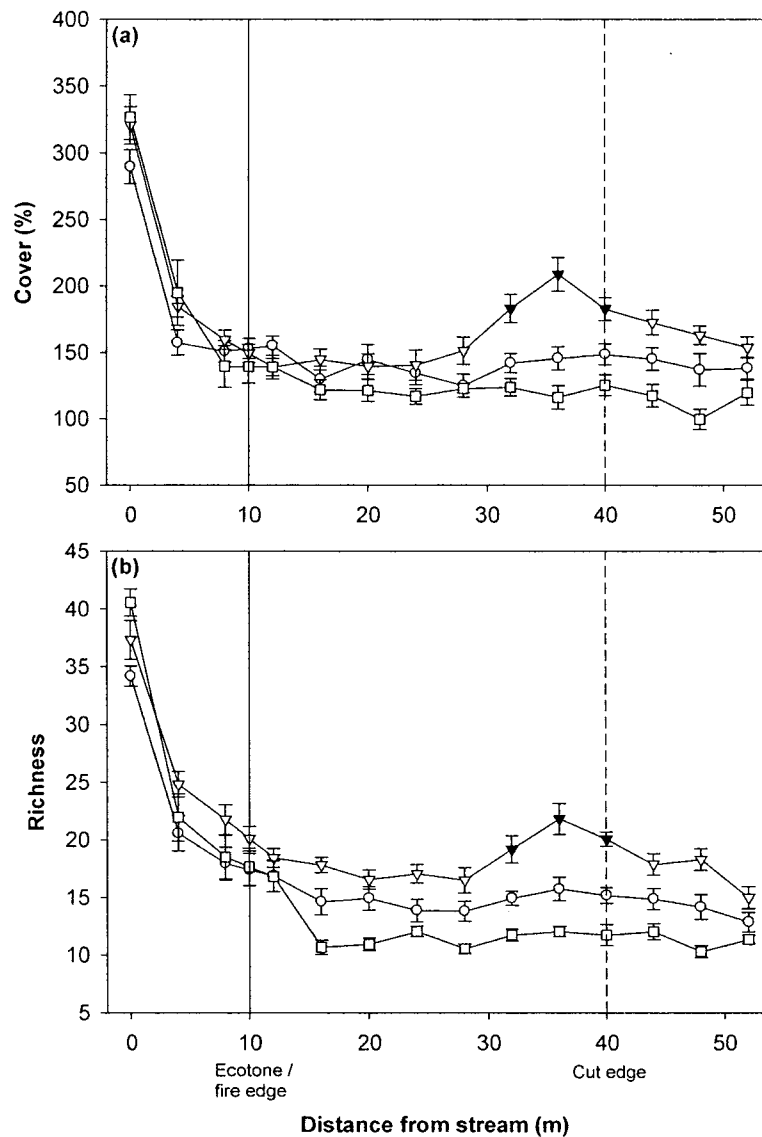
Figure 13. Mean (±SEM) understory species (a) cover, and (b) richness along gradients starting at stream edge. Symbols represent reference forest (circles), buffer (triangles); and fire (squares). Filled symbols indicate values were significantly different than those of reference fore condition (see Methods). The soil line at 10 m represents the average ecotonal / fire edge distance and the dashed line at 40 m represents the average clearcut edge location.
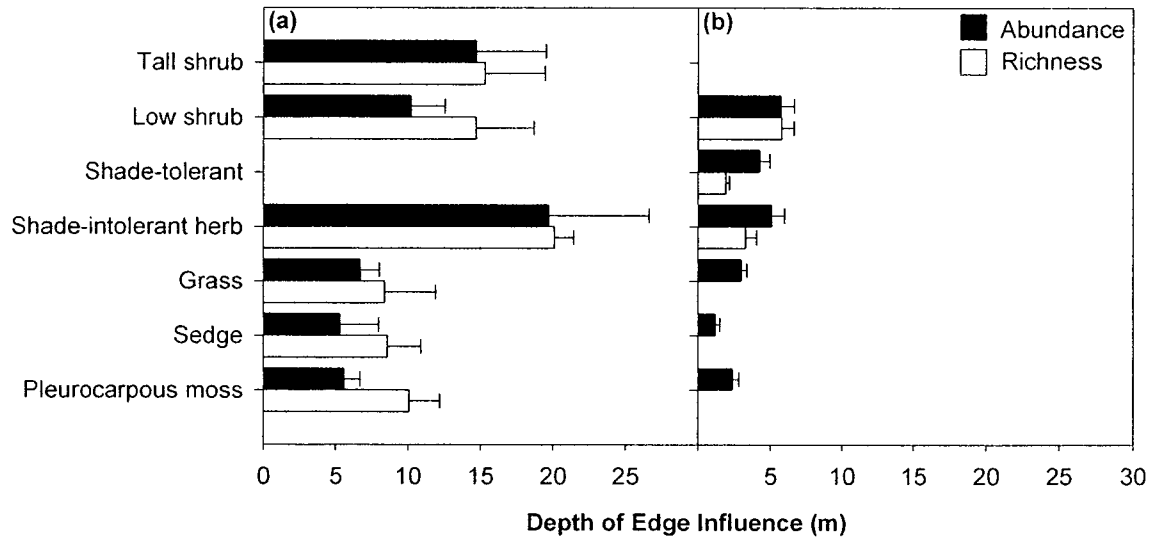
Figure 14. Mean (± SEM) depth of edge influence of plant life forms at (a) buffer edges, and (b) fire edges. Depth of edge influence was considered the point of two consecutive plots showing significant difference (p<0.05) from reference forests.

Magnitude of edge influence

For overstory structure, the absolute value of MEI was approximately two times greater (stronger MEI) at fire edges than buffer edges (Figure 15a). Soil moisture was negatively affected at clearcut edges and had an absolute MEI value more than two times greater than the fire edge. Overall, the absolute MEI values for the microclimatic variables were low and therefore, showed the least deviation from reference forest conditions (Figure 15b). The absolute MEI values for understory species richness and abundance were similar between the clearcut and fire edges (Figure 15c & d).
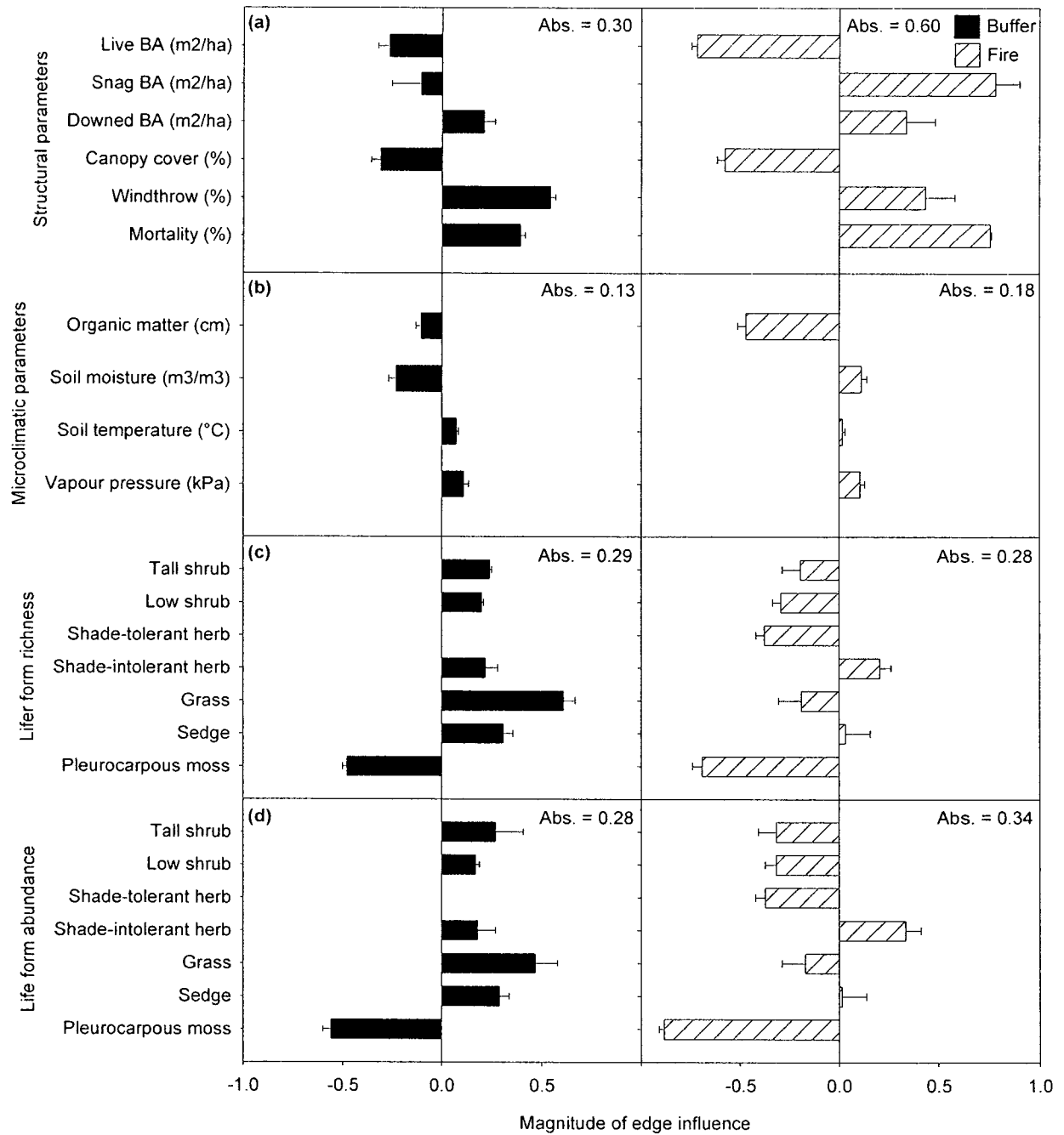
Figure 15. Mean (±SEM) magnitude of edge influence (see methods for calculation) of (a) structural parameters, (b) microclimatic parameters, (c) plant life form richness, and (d) plant life form abundance for the buffered and wild-fire sites. Abs. represents the overall absolute value for the response parameter group at the given edge type.

Discussion

The results of this study do not support the hypothesis that the understory species composition at fire edges would have a stronger MEI compared to reference forest than buffer edges. The species composition of fire edges showed a similar magnitude of deviation from reference forest conditions compared to buffer edges. However, differences between the plant communities with respect to edge type were found. Buffer edges had higher cover and richness of weedy graminoids, shade-intolerant tall shrubs and herbs with lower pleurocarpous moss cover. Whereas, the fire edges showed significantly lower abundance of low shrubs and pleurocarpous mosses with higher acrocarpous moss cover. The second hypothesis that edge influences would be weaker and less extensive at fire edges was supported by my results; the number of variables with stronger MEI was as great at fire edges as at buffer edges. Despite the greater structural damage encountered at riparian fire edges, the DEI was much lower for the near ground microclimatic variables. The close proximity of fire edges to the riparian zone appears to maintain the microclimatic conditions that are more similar to those of reference forest, thus affecting fewer understory species.

Edge location

In this study I found differences in the location (distance from stream) of clearcut and fire edges. The location of the edges is a product of how they are created. In the boreal forest, fires typically burn to the edge of the riparian zone (the ecotone or transition zone) and then either stop or jump the wetter barrier (Lamb et al. 2003). In the mid-Boreal region of Alberta, Macdonald et al. (2004) tested the hypothesis that canopy trees surrounding riparian areas were older which would indicate that fire stops further from the stream edge leaving a "natural buffer" around riparian zones. They found no significant difference in tree age directly adjacent to

riparian area compared with upland trees. This suggests that fire historically consumes most trees at the riparian ecotone and also tends to burn up to the riparian ecotone.

Edge orientation

According to Harper et al. (2005) edges facing prevailing winds are expected to have a relatively more pronounced edge influence. This assumption did not agree with my findings for the structural attributes of the edges. Over 260 000 observations between 1971 to 2000 at a weather station near the study area showed that the most frequent wind direction and maximum gust direction is from the west (Environment Canada, 2009). Therefore, although westerly winds tend to dominate, there was very little difference in the structural attributes of east and west-facing edges. Mascarúa-López et al. (2006) reported similar results when they compared the structural attributes of large forest patches, cutblock separators and riparian buffer strips in the boreal forest of northwestern Quebec. In their study, structural damage was either absent or did not extend into riparian buffers. They expected that older riparian buffers would have more structural damage after a longer exposure to increased wind than the more recently created cutblock separators. The difference was attributed to the fact that the trees within the buffers had already adapted to the edge environment near the stream and have likely become windfirm. Furthermore, Harper et al. (2002) suggested that in black spruce forests, trees are more likely windfirm since they are accustomed to growing in open conditions and are therefore less likely to be affected by increased wind at the edge. Although, fire left few live trees, these trees are assumed to be strongly windfirm because of their close proximity to the stream with increased wind exposure.

Edge effects on microclimate and vegetation in the northern hemisphere are typically more pronounced on south-facing edges than on north-facing edges, and east and west-facing

edges lie somewhere in between (Palik and Murphy, 1990; Fraver, 1994). Similarly, Hylander (2005) found that the difference in light exposure between north and south-facing edges modified the magnitude of the growth of two bryophyte species only when occurring close to the edge. I found no significant difference in microclimatic or vegetation compositional difference between east and west facing edges. Gysel (1951) and Bruner (1977) found similar results with no difference in edge effects between northern and southern orientations.

Buffer edge structure

The structural damage that I observed at the buffer edges is a common response in the boreal forests (Essen and Renhorn, 1998; Burton, 2002; Harper and Macdonald, 2002; Rheault et al., 2003; Stewart, 2004; Harper et al., 2004; Gignac and Dale, 2005; Mascarúa-López et al., 2006). After edge creation, increased tree mortality leads to decreased tree density and canopy cover and greater downed stem abundances (Harper and Macdonald, 2002). In black spruce forest, greater tree mortality near clearcut edges is most likely the result of windthrow rather than exposure to environmental factors (Harper et al., 2004). The significant reduction in canopy cover, and also increased densities of snag, and downed stems of buffer edges in this study were consistent with those reported for buffer edges of the black spruce forest of northwestern Quebec (Mascarúa-López et al., 2006) and northwestern Ontario (Stewart, 2004). However, my study shows approximately 16% increase in tree mortality and almost three times more windthrow than values reported by Mascarúa-López et al. (2006). The stand composition and age were similar in both studies and therefore makes explanations for the discrepancy difficult to interpret. However, the overall changes in live tree density, primarily attributed to windthrow, are consistent with those reported in the sub-boreal forest of British Columbia (Burton, 2002).

Fire edge structure

Very few studies have characterized the structural attributes of fire edges adjacent to riparian zones. Harper et al. (2004) preformed a study on clearcut and fire edges of upland the black spruce boreal forest of northwestern Quebec and found that fire edges had less than half the canopy cover and much higher densities snags and downed stems than clearcut edges. I found values that were comparable to those reported at 15 m away from the fire edge. The differences can be attributed to identification of the edge location. Harper et al. (2004) considered the fire edge the point at which continuous forest canopy began. Conversely, the fire edge of my study was considered either the riparian ecotone (edge of the riparian area where fire damage to foliage stopped) or the point of live canopy trees. Therefore, it could be expected to have significant structural differences when comparing upland fire edges to riparian fire edges.

Buffer compared to fire edge structure

The majority of structural differences observed between clearcut and fire edges are comparable to findings of Stewart (2004) and Harper et al. (2004). However, Harper et al. (2004) found an abrupt change in canopy cover only at clearcut edges whereas the response was much more gradual at fire edges. The immediate change in canopy cover was noted for both edges in this study. At the fire edge the alteration in canopy cover was the result of a few trees surviving the fire. Again, the method of fire edge location used in this study produced lower canopy cover, and higher mortality than studies performed on fires in continuous upland conditions (Stewart, 2004; Harper, et al. 2004).

Structural depth of edge influence

The DEI on forest structure extends 5-15 m from the clearcut edge into the riparian buffer. These findings are fairly consistent with other boreal studies (Appendix IV). Although

58

within a similar region as this study, Stewart (2004) found that the structural damage was restricted to edge plots, with no DEI. Alternatively, Mascarúa-López et al. (2006) found higher mortality for 5 m and lower live stem density for 10 m from buffer edge which agrees with my findings (±5 m). However, their research showed lower in canopy cover and higher windthrow for 15 m, in comparison to 5 m of my study. This disagreement may be function of time since disturbance. Their work was conducted on edges 2 to 5 years older than my study which would allow for more canopy damage from wind exposure. Nonetheless, my estimates of DEI in black spruce forest provide concurrent verification that edge influence is less extensive in boreal forests (Harper and Macdonald, 2002; Harper et al. 2005) compared to other ecosystems (Chen et al., 1992; Laurence et al., 1998), because boreal forests have shorter trees, are less productive, and are adapted to frequent disturbance (Harper and Macdonald, 2002).

Buffer compared to fire edge microclimate

Clearcut and fire edges had extensive microclimatic differences. In all but the most severe cases, fire leaves standing dead trees, which produces partial shade ameliorating the forest floor microclimate during hot, dry summer conditions (Carelton and MacLellan, 1994). Snags can intercept relatively high percentages of full light (Dwire and Kuaffman, 2003). Therefore, the combination of high shrub abundance at the riparian ecotone (Stewart and Mallik, 2006) and abundance of snags resulted in soil temperatures at fire edges that did not significantly differ from the reference forest. Furthermore, riparian plant communities exert considerable influence on local microclimate (Chen et al., 1999), with usually dense, closed canopies reducing evaporation and maintaining high relative humidity which in turn keeps fuel moisture levels high (Pettit and Naiman, 2007). Therefore, the close proximity of the fire edge to stream with higher water table and reduction of live stem density results in less water uptake and significantly

increased the soil moisture than the reference forest. Although organic matter depths at the fire edge were lowest of the three treatments, my study found no detectable difference in organic matter depth in plots directly at stream edge indicating that the severity of fire within the riparian zone was minimal. Also, no detectable difference in organic matter depth was noted between buffer and reference forest

Depth of edge influence on microclimate

Reference forest sites showed a slight increase in VPD with distance from stream. However, the fire and buffered streams showed significantly higher VPD with increasing distance from the stream. VPD had a significant DEI of 2 m past fire edge, while the DEI for the buffer sites was significantly higher for 30 m past the cut edge. Studying the microclimatic gradients of 1 to 2 year old buffers in northwestern Ontario, Stewart and Mallik (2006) had similar trends for VPD. Soil moisture also had a strong riparian-upland gradient at reference sites with moisture content decreasing with distance from stream edge. Soil moisture levels for the buffer followed a similar overall trend with increasing distance from stream, but significantly differed from reference condition for 20 m past the cut edge. Stewart and Mallik (2006) found less dramatic DEI for soil moisture at buffer edges. This difference may be related to the much larger sampling intervals used by Stewart and Mallik (2006) than my study (10 vs. 4 m). Murcia (1995) explained that edge effects are not necessarily monotonic and cautioned that a fine enough scale of sampling must be selected to give precise estimates and to identify significant fluctuations.

Overall, fire edges had a 2.5 m influence on the microclimatic parameters sampled in this study, while the clearcut edges averaged 8 m. These findings are much lower than those found at clearcut edges of upland boreal regions, which generally range from 10 to 50 m (Matlack, 1993;

Burke and Nol, 1998; Gignac and Dale, 2005). A critical difference between riparian buffers and upland forest fragments is that riparian buffers lie adjacent to a stream and are greatly influenced by the natural ecotone (i.e. stream-forest edge (Rykken et al., 2007). In a small stream setting, the edge effects of the clearcut and ecotonal edge are spatially compressed and overlapping (Sheridan & Olson, 2003; Stewart and Mallik, 2006). It is generally assumed that the cool/humid influence exerted by the stream on riparian and upslope microclimate mitigates the opposing warm/dry conditions contributed by the harvest-influenced edge effects (Pohlman et al., 2009). In this study, only partial burning of the riparian zone was observed with little scorching of the upper stems of shrubs. Therefore, the relatively intact riparian zone recovering from fire was minimally affected and was still able to have continued cooling and increased moisture influence on the surrounding area, as well as keeping consistent levels of VPD to that of undisturbed conditions.

My results are comparable to the 10 to 14 m zone of greatest microclimatic change reported for managed riparian stands in western Oregon (Chan et al., 2004; Rykken et al., 2006) , and also similar to a 10 m gradient for relative humidity measured by Danehy and Kirpes (2000) in eastern Oregon and Washington. While the structural changes and damage of boreal riparian buffers has been found to have lower DEI values than other forest types, my results suggest that microclimatic changes at buffer edges may be comparable to other geographical regions.

Plant community characteristics at the edges

The first set of MRPP analysis showed, that all site types reference, buffer and fire had low and alike T values for the riparian zone. Among all of the microhabitats, riparian, ecotone, upland, clearcut edge and clearcut/fire, the plant communities of the riparian zone appears to be most similar. Lamb et al. (2003) found similar results with no significant differences in the

overall abundance and distribution of riparian species adjacent to streams recovering from clearcutting and fire in northwestern Ontraio. The individual pair-wise comparison of reference to buffer sites and reference to fire sites revealed a similar trend of increasing dissimilarity with distance from stream edge. These findings suggest that as the distance from stream increases the effects of the disturbance on understory composition also increases. However, the second set of MRPP analyses showed that plant life form compositions were quite different between the fire and clearcut edges and the only form that showed similar response to the differing edge creations was the composition of conifer saplings.

Understory species responses

Forest edges generally have high plant species richness (Ranney et al., 1981; Lovejoy et al., 1986) and woody species stem density (Ranney et al., 1981; Matlack, 1994). In forests of eastern and central United States, forest edges were found to have higher species richness (Burner 1977; Palik & Murphy 1990). The buffer edges of this study agreed with these results. Lower total cover and richness with increased distance from the buffer edges may be due to the exclusion of weedy shade-intolerant herbs as light diminishes with greater canopy closer towards the interior (Stewart, 2004).

I found no significant change understory species richness at the fire edges. Comparable boreal studies on fire have generally found higher richness from 5 to 10 m past the edge (Harper et al., 2002; Stewart, 2004). Stewart (2004) explained that higher richness at fire edges is most likely due to the response of plant functional groups to the area of high moisture observed at fire edges. However, locality of the fire edge (i.e. near the streams) of this study becomes a more deterministic factor for the response of the plant communities. The riparian species present at the fire edge ecotone exhibit a range of adaptations to disturbance that contribute to rapid recovery

of streamside habitats following fire (Dwire and Kauffman, 2003). These include adaptations that facilitate the survival of plants on site, such as sprouting and thick bark, and those that contribute to re-colonization of burned sites, including wind and water dispersal, reproductive responses, and the capacity to establish in post-fire environments (Pettit & Naiman, 2007).

I found that the deciduous saplings (*Betula papyrifera* and *Populus* spp.) were significantly higer in cover and richness in the clearcut, but did not penetrate past buffer edge plots. Hardwood species often become more abundant on post-logging sites after the conifer component is removed (Davidson et al., 1988; Carleton & MacLellan, 1994). Logged mixed wood stands place fast-growing species at a competitive advantage over conifers causing a wholesale conversion to hardwood stands (Carleton & MacLellan, 1994).

Shrub and herb species richness tend to peak in clearcuts two to five years after logging, which is accompanied by a peak in richness of invading herbs (Schoonmaker & McKee, 1988). My study found that tall shrub cover and richness was significantly higer at buffer edges and to an average depth of 15 m. My results support the conclusions of Matlack (1994) for eastern deciduous forests, and Fraver (1994) and Ranney et al (1981) for mixed hardwood forests, who found that shrub richness was greatest along the edges. Also, my results support the findings of increase shrub cover for 20 to 25 m past old edges in the boreal mixed-wood forest of northern Alberta. However, these results are contrary to the findings of decreased shrub richness at 5 to 16 year old *Populus*-dominated forests of boreal forest in Alberta (Harper and Macdonald, 2002).

In this study the majority of indicator species of the clearcut and burn areas were grasses, sedges, weedy shade-intolerant herbs and shrubs. Laurance (1991) found that edges promote a high abundance of weedy and alien species. Brumelis and Carleton (1989) also found peak

abundances of *Calamagrosits canadensis* and *Carex* spp. in conifer dominated clearcuts. In my study graminoid species cover was higher in both the clearcut and up to 5 m past the buffer edge. However, fire edges only showed increases in graminoid cover to edge depths of 3 m. Arnup et al. (1995) explained that after fire sedge and grass species are rapid colonizers of burned sites due to fire stimulated flowering and rapid spread of rhizomes.

Several studies have found high abundance of shade-intolerant herbs and shrubs in the clearcut at the edges five years after fire (Schoomake & McKee, 1988; Harper and Macdonald 2002). This study found higher shade-intolerant herb cover and richness extending greater than 15 and 4 m past buffer and fire edges respectively. The majority of shade-intolerant herbs that showed higher cover and richness compared to reference forest are heavily reliant on seed dispersal by wind (*Aster ciliotaus, Epilobium angustifolium, Hieracium* spp. and *Taraxacum* spp.). Janzen (1983) found that regenerating vegetation and patch edges often experience a 'seed rain' of weeds and that are frequently better adapted to exposed and disturbed environments. These findings are consistent with the results of a study preformed in the same and adjacent area as mine, showing that buffers act as windbreaks where higher concentrations of wind dispersed species occur (Biswas, 2008).

The response of bryophytes to edge creation has been the focus of many studies in the boreal forest (Harper and Macdonald, 2002; Hylander et al., 2002; Hylander, 2005; Stewart, 2004; Gignac and Dale, 2005; Stewart and Mallik, 2006), particularly because these species can act as phytometers of environmental conditions within riparian buffer strips. In most boreal landscapes well-developed moss communities are common in conifer stands (Vitt, 1990). Bryophytes are poikilohydric, meaning they lack effective mechanisms for regulating uptake and loss of water (Proctor, 1990), and therefore time in hydrated condition is crucial for growth.

Many mosses cease to photosynthesize in hot dry conditions often encountered in clearcuts (Callaghan et al., 1978). My study showed comparably lower pleurocarpous moss cover and richness in the clearcut area and 5 to 10 m into the buffer respectively. However, the fire edge only showed a significantly lower pleurocarpous moss cover at the edge. The fire edge and burnt area supported higher abundances of acrocarpous mosses such as *Ceratodon purpureus* and *Polytrichum* spp. classified as "invaders", which are short-lived, easily dispersed pioneer species (Nguyen-Xuan et al., 2000). Other studies have reported the rapid increase in acrocarpous mosses following fire and a similar successional pattern of ground vegetation (Foster, 1985; Per-Anders et al., 1992). Fire edges were also characterized by significantly lower *Sphagnum* spp. abundance and richness, which is believed to be destroyed by the fire.

Two species *Corylus cornuta* and *Diervilla lonicera* were identified as indicators of both clearcut and fire edges. Both species are shade-intolerant shrubs present in variable moisture regimes, with high fire and drought tolerance and share similar regeneration strategies following disturbance (USDA NRCS., 2008). Analysis of disturbed sites in Manitoba revealed that *C. cornuta* was the most abundant shrub on burned, logged, or spruce budworm-defoliated sites (Kemball et al., 2005). The recovery of preharvest *C. cornuta* on logged sites is generally rapid (Ahlgren, 1974). Also following fire, *C. cornuta* sprouts from crown and/or rhizomes after top-kill (Buckman, 1964; Carleton and MacLellan, 1994). The rhizomatous growth habit greatly increases the ability of *C. cornuta* to rapidly regain or exceed prefire cover (Kemball et al., 2005). Similar to *C. cornuta*, *D. lonicera* rapidly regenerates vegetatively following fire and its abundances in post-fire communities are comparable to pre-fire densities (Ahlgren, 1974; Krefting et al., 1974). Although both species were identified as important components of the clearcut and fire edges in this study, no significant DEI was apparent for either species at both

edge types. Harper et al. (2002) found similar results of increased cover of *Lonicera dioica* (a member of the same family, Caprifiaceae as *D. lonicera*) at clearcut edges, which indicates that species of this family are highly adapted to gaining dominance after clearcutting.

Limitations

The following limitations of this study were identified: (1) Issues of serial statistics were not dealt with. As a consequence of running large numbers of ANOVA tests, there is an increased likelihood of creating Type I error (i.e. rejecting the null hypothesis when it is true). Therefore, as a 0.05 alpha level for the p-value (significance level) was selected, it is probable that one in every 20 ANOVA tests resulted in false results. Furthermore, the response variables used in this study cannot be considered independent. I should have run preliminary nested MANOVAs to determine which variables were contributing the most deviation between the treatments prior to the use of individual ANOVA tests. (2) Only east and west-facing edges were studied. Similar edge research performed in the northern hemisphere has shown a detectable difference between north and south-facing edges. Therefore, edge effects may be different along other edge orientations across the landscape. (3) DEI for microclimate and understory composition may have been more precise had contiguous quadrats been used, instead of 4 m intervals.

Conclusions

The following conclusions were made for this study: (1) The structural characteristics of the overstory and species compositions of the understory at fire edges was significantly different than buffer edges. I concluded that the main reason for observed difference is that fire edges

were located closer to the streams while buffer edges were on average 40 m from the stream and this had a greater overlap with the riparian ecotone. (2) Buffer edges have large DEI than fire edges. This is mainly due to: (i) the more gradual structural changes from snags and downed trees at fire edges than the abrupt edge structure at buffer edges; and (ii) the location of the fire edge is closer to the stream which has a moderating influence on the microclimate and species compositions. (3) The magnitude of edge influence on species composition between the buffer and fire edges were the same. This implies that the dominant species at the edges are resistant to both types of disturbance. (4) East and west-facing edges were similar in structure, near ground microclimate, and species composition. Wind firming of trees along riparian zones because of naturally increased wind within riparian corridors and comparable sun angles of east and west facing edges are atonable to this similarity.

Management implications

This study demonstrated that under current management buffer edges are structurally and compositionally different than fire edges along streams. Many studies have recommended that wider buffers be implemented to ensure sufficient continuous forested habitat to maintain and preserve forest interior species (Brososke et al. 1997; Pearson & Manuwal 2001; Hannon et al. 2002; Hylander et al. 2002; Dignan & Bren 2003; Hylander et al. 2004; Potvin & Bertrand 2004). Although, the preservation of interior species is of great importance, current hypotheses have been proposed to maintain biodiversity in managed forests using natural disturbance regimes to guide forest harvesting operations. In Ontario, guidelines have been designed to change the pattern of forest harvesting at the landscape and stand level to create forest harvesting operations that better simulate the way fire disturbs the forest. However, these guidelines rely on

the assumption that following the size and shape of wildfires alone will create conditions with similar ecological functions. My results suggest that the sharp edges of buffers create much a different microclimate and as a result much different understory composition than the gradual structural features of fire edges. Therefore, the ecological processes occurring at buffer and fire edges are much different and should be considered with attempts of emulation. Furthermore, conventional management of streams using buffers is not consistent with efforts to implement harvesting techniques that approximate natural disturbance patterns because such disturbances occur in riparian areas at rates similar to the upland forest.

Within the managed forests of northwestern Ontario, some measures must be adopted to reduce contrast between buffers and the clearcut area. The findings that the DEI at buffer edges diminishes between 5 to 20 m suggest that current buffer prescriptions are effective for the protection of riparian zones. However, I recommend reducing the contrast between the harvested area and the edge by maintaining the structural features of natural forest edges through harvesting boundaries that are 'feathered' or selectively harvested to create wider edges that mimic the effects of burning. Harvesting closer to stream is not a viable option because other studies have shown severe negative effects on water quality, aquatic communities and the soil structural stability of stream banks. Therefore, managing riparian buffers that better emulate the structure and composition of fire edges may be accomplished with of a mixture of selective harvesting techniques with possibly prescribed burning, although much research is needed before such techniques could be implemented.

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Appendix I – Site descriptions

Buffer Sites	UTM Coordinates[a]	Flow Direction	Stream Width[b]	Slope (°) East	Slope (°) West	Ecotonal Edge[b]	Dominant Riparian Vegetation[c]	Harvest Year[d]	Cut Edge(m) East	Cut Edge(m) West
BEA4	350690 5399382	N to S	0.47	5	5	8	*A. incana L. uniflorus*	2002	44 32	32 36
RRD1	351257 5395312	S to N	0.89	16	5	9	*A. incana T. dayscarpum*	2004	52 52	32 28
RRD2	351509 5395659	N to S	0.94	7	8	9	*A. incana Poaceae spp.*	2001	48 36	28 36
MBT2	360317 5395285	N to S	0.67	11	9	9	*A. incana Poaceae spp.*	2003	28 32	44 40
RC1	349902 5393928	S to N	0.98	14	18	9	*A. incana T. dayscarpum*	2004	44 32	40 32
HB11	349515 5397546	N to S	0.67	16	12	11	*A. incana L. uniflorus*	2002	40 48	48 48
MBT3	349515 5397546	N to S	0.85	15	11	8	*C. stolonifera Poaceae spp.*	2002	36 44	36 36
EEL1	361774 5399324	S to N	1.44	12	11	8	*A. incana Poaceae spp.*	2003	40 44	40 40

Fire Sites	UTM Coordinates[a]	Flow Direction	Stream Width[b]	Slope (°) East	Slope (°) West	Ecotonal edge[b]	Dominant Riparian Vegetation	Fire Year[e]	Fire Index[e]	Fire Cause[e]
NIP52	390521 5435531	S to N	1.01	10	14	12	*A. incana A. spicatum*	2005	15	Lightning
NIP101	373268 5462009	N to S	1.63	10	5	6	*A. incana T. dayscarpum*	2000	15	Industrial
NIP102	374084 5462075	N to S	0.96	17	15	6	*A. incana T. dayscarpum*	2000	6	Industrial
NIP103	375112 5462002	N to S	1.11	13	11	10	*A. incana Poaceae spp.*	2000	6	Industrial
THU16S	327285 5496915	N to S	0.78	5	6	10	*A. spicatum C. stolonifera*	2002	2	Lightning
THU16N	328330 5498250	N to S	1.21	8	14	8	*A. incana Poaceae spp.*	2002	2	Lightning
THU576	296206 5474626	S to N	1.34	5	9	11	*A. incana A. felix-femina*	2005	12	Lightning
THU577	286942 5473816	N to S	0.61	11	10	10	*A. incana A. spicatum*	2005	12	Lightning

Reference Sites	UTM Coordinates[a]	Flow Direction	Stream Width[b]	Slope (°) East	Slope (°) West	Ecotonal Edge[b]	Dominant Riparian Vegetation[b]			
LFP21	353828 5397791	S to N	0.77	4	3	8	*A. incana Poaceae spp.*			
WTT23	349144 5400527	N to S	1.26	6	8	10	*A. incana C. stolonifera*			
RSW24	351809 5399986	S to N	1.66	8	12	11	*A. incana C. stolonifera*			
AM4	373475 5391721	N to S	1.77	11	13	6	*A. incana Poaceae spp.*			
RN5	373357 5391007	N to S	1.04	9	5	5	*C. stolonifera Poaceae spp.*			
LIL3	349454 5404111	N to S	0.43	10	14	8	*A. incana Poaceae spp.*			
MAC4	346581 5406791	S to N	1.96	10	13	9	*A. incana Poaceae spp.*			
LIL2	351063 5401825	N to S	0.57	7	11	6	*C. stolonifera*			

[a] UTM coordinates are within the 16 U zone (i.e. 90 to 84° W longitude).

[b] Meters from stream.

[c] Based on visual estimates.

[d] Harvest dates were provided by Ontario Ministry of Natural Resources (McCormick 2008).

[e] Fire data by the Thunder Bay Fire Management Headquarters – Ontario Ministry of Natural Resources (Raman 2008).

Appendix II – Stand composition by site type

Mean (± S.E.M.) for species composition (%); calculated by the ratio of number of stems/species : total stem number.

Species	Reference (n = 160)	Buffer (n = 128*)	Fire (n = 160)	Prob.>F**
Abies balsamea	5.0 ± 1.8[a]	8.3 ± 3.4[a]	3.0 ± 0.6[a]	0.251
Betula papyrifera	3.8 ± 1.2[a]	5.7 ± 1.7[a]	17.4 ± 1.5[b]	0.000
Fraxinus nigra	0.3 ± 0.2[ab]	0.0 ± 0.0[a]	0.7 ± 0.2[b]	0.010
Picea glauca	0.2 ± 0.1[a]	2.8 ± 1.2[b]	0.0 ± 0.0[a]	0.009
Picea mariana	60.4 ± 4.3[a]	63.8 ± 2.7[a]	65.2 ± 1.5[a]	0.526
Pinus banksiana	21.7 ± 4.8[b]	5.3 ± 2.9[a]	11.5 ± 2.1[ab]	0.006
Pinus resinosa	0.1 ± 0.1[a]	0.0 ± 0.0[a]	1.0 ± 0.6[a]	0.021
Populus balsamifera	0.4 ± 0.3[a]	0.9 ± 0.6[a]	0.0 ± 0.0[a]	0.324
Populus tremuloides	4.5 ± 1.9[a]	13.2 ± 3.3[b]	0.5 ± 0.4[a]	0.001
Thuja occidentalis	3.7 ± 1.7[b]	0.0 ± 0.0[a]	0.7 ± 0.3[ab]	0.033

*Plots adjacent to cut edge were excluded from analysis.
**Results of analysis of variance on square root-transformed data using SPSS v. 16 (SPSS 2008). If Prob.>F is greater than 0.05, then difference between treatments is considered insignificant at the 95% confidence level.

Appendix III – Stand stem density by site type

Mean (± S.E.M.) for total stem density (stems/ha) and basal area (m^2/ha) for all plots (within each treatment).

Stand Attribute	Reference (n = 160)	Buffer (n = 128*)	Fire (n = 160)	Sig. Prob.>F**
Total stem density	1916.25 ± 66.43[a]	1854.69 ± 82.49[a]	1827.50 ± 40.57[a]	0.5761
Total basal area	19.41 ± 1.53[a]	19.64 ± 1.74[a]	13.44 ± 1.21[b]	**0.0021**
Average tree basal area	2.31 ± 0.09[a]	2.11 ± 0.08[a]	1.93 ± 0.06[b]	**0.0382**

*Plots adjacent to cut edge were excluded from analysis.

**Results of analysis of variance on log transformed data using SPSS v.16 (SPSS 2008). If the Prob.>F is less than 0.05, then the difference between treatments is considered significant at the 95% confidence level.

Appendix IV – Literature review of boreal / temperate forest edge influence

To synthesize the literature on edge influence on forest structure, microclimate and composition at clearcut and fire edges, I compiled the following database of published studies.[a]

Category[b]	Edge Source[c]	Edge type[d]	Edge Age	Response[e]	Mean MEI	Mean DEI	DEI range	Study
Tree mortality								
Boreal	A	LFP	2 to 16	H	0.85	n/a		Esseen & Renhorn, 1998
Boreal	A	LFP	9 to 21	H	0.36	58	54 to 62	Burton ,2002
Boreal	A	LFP	5 to 16	H & L	0.11	22	0 to 60	Harper & Macdonald, 2002
Boreal	A	LFP	5 to 11	H			0 to 60	Mascrúa López et al., 2006
Boreal	A	CBS	5 to 11	H			0 to 20	Mascrúa López et al., 2006
Boreal	A	RB	5 to 11	H			0 to 15	Mascrúa López et al., 2006
Windthrow								
Boreal	A	LFP	5 to 11	H			0 to 25	Mascrúa López et al. 2006
Boreal	A	CBS	5 to 11	H			0 to 30*	Mascrúa López et al. 2006
Boreal	A	RB	5 to 11	H			0 to 15	Mascrúa López et al. 2006
Canopy tree abundance								
Boreal	A	LFP	9 to 21	L	-0.11	67		Burton ,2002
Boreal	A	LFP	16	L	-0.42	2.5	0 to 5	Harper & Macdonald, 2002
Boreal	A	LFP	1 to3 / 16 to 23	L	-0.06	n/a		Rheault et al., 2003
Boreal	F	F	3 to 4	L	-0.11	5		Harper et al., 2004
Boreal	A	CBS	5 to 11	L			0 to 25	Mascrúa López et al., 2006
Boreal	A	RB	5 to 11	L			0 to 10	Mascrúa López et al., 2006
Canopy Cover								
Boreal	A	LFP	9 to 21	L	-0.11	67		Burton ,2002
Boreal	A	LFP	5 to 16	L	0		5 to 10	Harper & Macdonald, 2002
Boreal	A	LFP	1 to3 / 16 to 23	L	-0.14	5		Rheault et al,. 2003
Boreal	A	LFP	3 to 7	L		0**		Stewart, 2004
Boreal	A	LFP	5 to 11	L			0 to 25	Mascrúa López et al., 2006
Boreal	A	CBS	5 to 11	L			0 to 30*	Mascrúa López et al., 2006
Boreal	A	RB	5 to 11	L			0 to 15	Mascrúa López et al., 2006
Snag and downed abundance								
Boreal	A	LFP	2 to 16	H	-0.2	n/a		Esseen & Renhorn, 1998
Boreal	A	LFP	9 to 21	H	-0.11	n/a		Burton, 2002
Boreal	A	LFP	5 to 16	H	0.02	17	10 to 20	Harper & Macdonald, 2002
Boreal	A	LFP	1 to3 / 16 to 23	H	0.37	5		Rheault et al., 2003
Boreal	F	F	3 to 4	H	0.81	40		Harper et al., 2004

84

Appendix IV continued

Category[b]	Edge Source[c]	Edge type[d]	Edge Age	Response[e]	Mean MEI	Mean DEI	DEI range	Study
Boreal	A	LFP	5 to 11	H			0 to 10	Mascrúa López et al., 2006
Boreal	A	CBS	5 to 11	H			0 to 10	Mascrúa López et al., 2006
Boreal	A	RB	5 to 11			ns		Mascrúa López et al., 2006
Species Richness								
Temperate	A	LFP	n/a	H		15		Burner ,1977
Temperate	A	LFP	5 to 10	H		25	5 to 45	Palik & Murphy, 1990
Temperate	A	LFP	n/a	H	n/a	7	2 to 11	Gysel, 1951
Understory tree abundance								
Boreal	A	LFP	n/a	H	n/a	20	10 to 30	Ranney et al., 1981
Boreal	A	LFP	0 to 325	H	n/a	60		Harper & Macdonald, 2002
Shrub cover								
Temperate	A	LFP	0 to 5	H & L	0.37	40		Matlack, 1993
Boreal	A	LFP	5 to 16	L	n/a	3.75	0 to 5	Harper & Macdonald, 2002
Boreal	F	F	3 to 4	L		5		Harper et al., 2004
Boreal	A	LFP	3 to 7	L		40		Stewart ,2004
Boreal	A	LFP	70+	H			20 to 25	Gignac and Dale, 2007
Herb cover								
Boreal	A	LFP	5 to 16	H & L	n/a	1.25	0 to 5	Harper & Macdonald, 2002
Moss cover								
Boreal	F	F	5 to 16	L	-0.21		40	Harper & Macdonald ,2002
Boreal	A	RB	n/a	L			10 to 15**	Hylander et al., 2002
Boreal	A	LFP	2 to 16	L			15 to 20	Gignac and Dale, 2005
Boreal	A	RB	3 to 7	L			10 to 30**	Stewart and Mallik, 2006
Lichen cover								
Boreal	A	LFP	n/a	L		50		Moen and Jonsson ,2002
Boreal	A	LFP	1 to 3 / 16 to 23	L		50		Rheault et al., 2003
Boreal	A	RB / CBS	5 to 11	L			15 to 30	Boudralt et al., 2008
Individual species abundance								
Temperate	A	LFP	8 to 12	H & L	n/a	9	0 to 30	Euskirchen et al., 2001
Boreal	A	LFP	2 to 16	H	-0.18	50		Esseen & Renhorn, 1998
Boreal	A	LFP	5 to 16	H	n/a	25	0 to 60	Harper & Macdonald, 2002
Temperate	A	LFP	50+	H		40	20 to 60	Fraver, 1994
Air Temperature								
Temperate	A	LFP	0 to 5	H			13 to 24	Matlack, 1993
Boreal	A	LFP	70+	H		15		Gignac and Dale, 2005
Boreal	A	RB	1 to 2	H		10		Stewart and Mallik, 2006

Appendix IV continued

Category[b]	Edge Source[c]	Edge type[d]	Edge Age	Response[e]	Mean MEI	Mean DEI	DEI range	Study
Light intensity								
Temperate	A	LFP	0 to 5	H			10 to 35	Matlack 1993
Boreal	A	LFP	9 to 21	H			65 to 70	Burton 2002
Boreal	A	LFP	70+	H		15		Gignac and Dale 2005
Boreal	A	RB	1 to 2	H		10		Stewart and Mallik 2006
Soil Moisture								
Temperate	A	LFP	n/a	H		20		Burke and Nol 1998
Relative humidity								
Temperate	A	LFP	0 to 5	L		50		Matlack 1993
Boreal	A	LFP	1 to 2	L		10		Stewart and Mallik 2006
Vapour pressure deficit								
Boreal	A	LFP	1 to 2	H		10		Stewart and Mallik 2006

[a] Studies were found in previous data compilations (Harper et al. 2005) and from a literature review of common ecology journals.

[b] Results were divided into two categories of region: boreal forest and temperate eastern North American forest.

[c] Anthropogenic (A) or fire (F) edge source.

[d] Edges were divided into type: LFP, large forest patch; CBS, cutblock separator; RB, riparian buffer.

[e] High (H) or low (L) in edges relative to interior forest.

*Edge influence extended across entire cutblock separator / riparian buffer.

** No significant DEI. Only edge plots were found to significantly differ.

No information available (n/a).

Edge influence is non-significant (ns).

Appendix V – Indicator species analysis

Indicator species at different microhabitats (1 = riparian zone, 2 = ecotonal edge/fire edge, 3 = upland, 4 = clearcut edge, 5 = clearcut/burn) for each treatment. Relative frequency and abundance of each species within each treatment is summarized by the observed indictor value. The mean randomized indicator value is determined through 5000 Monte Carlo runs with randomized data. The p-value is the proportion of the randomized trials with an indicator value equal or exceeding the observed indicator value.

Treatment	Indicator Species	Max region	Observed indicator value (IV)	Mean randomized indicator value	p - value (sig.≤ 0.05)
Reference	*Agrostis gigantean*	1	9.4	2.5	0.0120
	Alnus incana	1	48.0	8.7	0.0002
	Aquilegia brevistyla	1	5.5	2.8	0.0488
	Aster nemoralis	1	20.8	3.6	0.0002
	Athyrium felix-femina	1	20.2	5.5	0.0002
	Aulocomnium palustre	1	40.8	6.2	0.0002
	Brachythecium relutinum	1	9.4	2.6	0.0134
	Brachythecium rivulare	1	44.3	5.0	0.0002
	Brachythecium velutinum	1	7.2	3.3	0.0344
	Bromus ciliatus	1	23.6	3.9	0.0002
	Calamagrostis canadensis	1	43.7	4.9	0.0002
	Campanula aparinoides	1	12.5	2.8	0.0018
	Carex intumescens	1	12.5	3.0	0.0018
	Carex lasiocarpa	1	15.6	3.0	0.0004
	Carex spp.	1	39.1	5.1	0.0002
	Chamaedaphne calyculata	1	28.1	3.9	0.0002
	Climacium dendroides	1	48.5	5.1	0.0002
	Cornus stolonifera	1	30.7	6.5	0.0002
	Dicranum spp	1	8.0	3.1	0.0124
	Equisetum sylvaticum	1	13.9	6.6	0.0096
	Eupatorium maculatum	1	9.4	2.6	0.0138
	Fissidens spp.	1	50.0	5.0	0.0002
	Galium asperellum	1	27.7	3.9	0.0002
	Graminoid spp.	1	46.3	8.0	0.0002
	Lycopus uniflorus	1	52.3	5.2	0.0002
	Mentha arvensis	1	18.7	3.2	0.0002
	Mertensia paniculata	1	15.9	4.1	0.0004
	Mitella nuda	1	12.3	5.9	0.0112
	Mnium marginatum	1	52.2	5.3	0.0002
	Mnium spp.	1	38.3	5.4	0.0002
	Physocarpus opulifolius	1	12.5	2.7	0.0016
	Plagiomnium spp.	1	28.3	6.1	0.0002
	Rhizomnium pseudopunctatum	1	53.6	5.6	0.0002

Treatment	Indicator Species	Max region	Observed indicator value (IV)	Mean randomized indicator value	p - value (sig.≤ 0.05)
Reference	*Rhizomnium punctatum*	1	37.3	6.9	0.0002
	Rhodobryum roseum	1	11.7	4.6	0.0098
	Rubus acaulis	1	38.3	6.9	0.0002
	Rubus idaeus	1	21.6	6.2	0.0004
	Rubus pubescens	1	21.0	7.0	0.0006
	Salix spp.	1	15.4	5.5	0.0030
	Scirpus atrovirens	1	9.4	2.5	0.0142
	Scirpus cyperinus	1	9.4	2.6	0.0138
	Scorpidium scorpioides	1	39.9	5.6	0.0002
	Scutellaria galericulata	1	21.9	3.7	0.0002
	Solidago rugosa	1	18.7	3.2	0.0002
	Solidago uliginosa	1	12.5	2.8	0.0026
	Sphagnum girgensohnii	1	10.6	4.5	0.0140
	Thalictrum dayscarpum	1	49.6	5.3	0.0002
	Thelypteris palustris	1	9.4	2.6	0.0122
	Thuidium delicatulum	1	39.5	6.6	0.0002
	Viburnum edule	1	12.5	2.8	0.0020
	Viburnum trilobum	1	12.5	2.9	0.0026
	Viola blanda	1	15.8	5.2	0.0026
	Viola renifolia	1	16.2	8.2	0.0068
	Viola septentrionalis	1	43.7	5.8	0.0002
	Dicranum ontariense	2	19.3	8.6	0.0028
	Epilobium angustifolium	2	9.2	4.1	0.0204
	Fragaria virginiana	2	12.8	5.3	0.0076
	Galium triflorum	2	18.2	4.8	0.0004
	Pleurozium schreberi	2	24.1	18.8	0.0206
	Dicranum fuscescens	3	9.4	5.1	0.0426
	Diervilla lonicera	4	17.0	10.2	0.0144
	Picea mariana	5	11.6	4.6	0.0078
Buffer	*Alnus crispa*	1	17.5	8.3	0.0068
	Alnus incana	1	72.5	9.4	0.0002
	Aquilegia brevistyla	1	12.5	2.8	0.0020
	Athyrium felix-femina	1	62.6	9.4	0.0002
	Aulocomnium palustre	1	71.9	6.0	0.0002
	Brachythecium rivulare	1	56.2	5.2	0.0002
	Brachythecium velutinum	1	12.0	3.1	0.0030
	Bromus ciliatus	1	8.4	3.7	0.0270
	Calamagrostis canadensis	1	22.4	13.6	0.0112
	Caltha palustris	1	15.6	3.2	0.0010
	Campanula aparinoides	1	9.4	2.5	0.0126

Treatment	Indicator Species	Max region	Observed indicator value (IV)	Mean randomized indicator value	p - value (sig.≤ 0.05)
Buffer	*Carex bebbii*	1	11.5	2.9	0.0016
	Carex disperma	1	12.5	2.9	0.0020
	Carex intumescens	1	17.2	4.1	0.0002
	Carex lasiocarpa	1	17.0	5.5	0.0008
	Carex spp.	1	37.2	5.0	0.0002
	Carex trisperma	1	16.1	3.6	0.0004
	Climacium dendroides	1	62.5	6.1	0.0002
	Cornus stolonifera	1	48.6	6.5	0.0002
	Dicranum spp.	1	8.2	2.9	0.0280
	Equisetum fluviatile	1	20.0	3.6	0.0002
	Equisetum sylvaticum	1	16.3	7.5	0.0092
	Euryhynchium pulchellum	1	37.5	4.3	0.0002
	Fissidens spp.	1	53.1	5.3	0.0002
	Galium asperellum	1	63.2	6.0	0.0002
	Galium triflorum	1	13.6	5.7	0.0064
	Gymnocarpium dryopteris	1	15.5	5.4	0.0014
	Impatiens capensis	1	9.4	2.6	0.0126
	Lactuca spp.	1	12.5	2.8	0.0014
	Lycopus uniflorus	1	60.5	6.0	0.0002
	Mertensia paniculata	1	25.9	4.2	0.0002
	Mitella nuda	1	61.3	5.7	0.0002
	Mnium marginatum	1	68.7	5.9	0.0002
	Mnium spp.	1	62.5	5.7	0.0002
	Onoclea sensibilis	1	9.4	2.5	0.0110
	Phegopteris connectilis	1	9.5	4.0	0.0206
	Plagiomnium spp.	1	60.8	6.3	0.0002
	Pteridum aquilinum	1	9.4	2.5	0.0126
	Rhizomnium pseudopunctatum	1	58.6	6.0	0.0002
	Rhizomnium punctatum	1	71.8	6.8	0.0002
	Rhodobryum roseum	1	40.9	5.9	0.0002
	Rhytidiadelphus triquet	1	29.5	8.4	0.0002
	Rubus acaulis	1	52.0	11.3	0.0002
	Rubus pubescens	1	16.3	8.5	0.0086
	Salix spp.	1	9.2	3.2	0.0130
	Scorpidium scorpioides	1	43.8	6.1	0.0002
	Sphagnum angustifolium	1	13.6	4.9	0.0016
	Thalictrum dayscarpum	1	59.4	5.6	0.0002
	Thelypteris palustris	1	12.5	2.8	0.0024
	Thuidium delicatulum	1	74.1	6.4	0.0002
	Viburnum edule	1	9.4	3.0	0.0150

Treatment	Indicator Species	Max region	Observed indicator value (IV)	Mean randomized indicator value	p - value (sig.≤ 0.05)
Buffer	*Viburnum trilobum*	1	7.1	3.4	0.0496
	Viola blanda	1	21.5	8.1	0.0004
	Viola renifolia	1	48.8	10.9	0.0002
	Viola septentrionalis	1	27.4	6.1	0.0002
	Anemone canadensis	2	7.5	3.2	0.0336
	Carex michauxiana	2	8.8	2.8	0.0156
	Coptis trifolia	2	21.0	13.5	0.0138
	Lycopodium annotinum	2	23.8	9.2	0.0002
	Mainthemum canadense	2	22.5	16.4	0.0166
	Petasites frigidus	2	18.1	4.9	0.0006
	Trientalis borealis	2	22.8	10.2	0.0012
	Aralia nudicaulis	3	21.8	15.2	0.0112
	Cornus canadensis	3	24.1	19.7	0.0190
	Pleurozium schreberi	3	35.0	15.8	0.0002
	Ptilium crista-castrensis	3	31.0	10.1	0.0002
	Corylus cornuta	4	20.6	14.3	0.0184
	Diervilla lonicera	4	29.6	18.0	0.0004
	Poa compressa	4	13.4	3.8	0.0010
	Poa pratensis	4	7.9	3.4	0.0296
	Scirpus cyperinus	4	6.2	2.6	0.0470
	Aster ciliolatus	5	12.6	7.8	0.0398
	Aster macrophyllus	5	26.6	15.0	0.0004
	Betula papyrifera	5	10.0	5.7	0.0368
	Carex eburnean	5	14.6	8.9	0.0384
	Ceratodon purpureus	5	12.3	5.2	0.0120
	Corydalis sempravirens	5	10.6	5.6	0.0306
	Dicranum polysetum	5	7.7	3.8	0.0342
	Epilobium angustifolium	5	28.7	9.5	0.0002
	Epilobium spp.	5	13.2	4.3	0.0032
	Hieracium caespitosum	5	6.2	2.6	0.0434
	Hieracium aurantiacum	5	6.2	2.5	0.0414
	Lycopodium digitatum	5	13.0	7.0	0.0172
	Picea mariana	5	12.0	4.2	0.0014
	Polytrichum juniperinum	5	14.6	3.4	0.0006
	Populus balsamifera	5	23.1	6.2	0.0002
	Populus tremuloides	5	34.6	10.3	0.0002
	Prunus pensylvanica	5	11.4	6.8	0.0404
	Rubus idaeus	5	32.3	13.8	0.0002
	Solidago canadensis	5	12.5	3.1	0.0012
	Vaccinium angustifolium	5	29.7	17.4	0.0002

Treatment	Indicator Species	Max region	Observed indicator value (IV)	Mean randomized indicator value	p - value (sig.≤ 0.05)
Fire	Acer spicatum	1	49.7	12.4	0.0002
	Agrostis gigantean	1	9.4	2.6	0.0148
	Alnus crispa	1	18.3	5.1	0.0006
	Aralia nudicaulis	1	25.5	7.0	0.0002
	Athyrium felix-femina	1	58.2	6.5	0.0002
	Atrichum sp.	1	23.3	3.8	0.0002
	Aulocomnium palustre	1	54.3	5.5	0.0002
	Brachythecium rivulare	1	46.9	4.8	0.0002
	Bromus ciliatus	1	41.4	5.2	0.0002
	Calamagrostis canadensis	1	40.7	7.8	0.0002
	Carex deweyana	1	18.7	3.2	0.0004
	Carex disperma	1	28.8	4.2	0.0002
	Carex intumescens	1	19.2	5.3	0.0002
	Carex trisperma	1	15.6	3.0	0.0006
	Chamaedaphne calyculata	1	9.4	2.5	0.0104
	Climacium dendroides	1	53.1	5.3	0.0002
	Clintonia borealis	1	15.0	10.4	0.0486
	Coptis trifolia	1	17.6	4.0	0.0004
	Cornus stolonifera	1	53.1	5.0	0.0002
	Eupatorium maculatum	1	11.3	4.0	0.0064
	Euryhynchium pulchellum	1	37.3	5.6	0.0002
	Fissidens sp.	1	50.0	5.1	0.0002
	Fraxinus nigra	1	22.9	4.3	0.0002
	Galium asperellum	1	46.9	4.7	0.0002
	Galium trifidum	1	28.1	3.8	0.0002
	Galium triflorum	1	41.3	5.5	0.0002
	Glyceria spp	1	43.2	5.8	0.0002
	Graminoid spp	1	15.4	8.0	0.0118
	Hypnum spp.	1	18.7	3.2	0.0002
	Lycopus americanus	1	20.5	4.1	0.0002
	Lycopus uniflorus	1	39.0	5.0	0.0002
	Mentha arvensis	1	18.5	5.0	0.0004
	Mertensia paniculata	1	29.7	6.0	0.0002
	Mitella nuda	1	40.8	6.2	0.0002
	Mnium marginatum	1	53.1	5.2	0.0002
	Mnium spp.	1	43.7	4.7	0.0002
	Onoclea sensibilis	1	25.0	3.6	0.0002
	Osmunda claytoniana	1	15.6	3.3	0.0002
	Phegopteris connectilis	1	35.6	5.7	0.0002
	Plagiomnium spp.	1	56.2	5.2	0.0002

Treatment	Indicator Species	Max region	Observed indicator value (IV)	Mean randomized indicator value	p - value (sig.≤ 0.05)
Fire	*Poa compressa*	1	9.4	2.5	0.0128
	Ptilium crista-castrensis	1	34.4	4.3	0.0002
	Rhizomnium pseudopunctatum	1	43.6	5.2	0.0002
	Rhizomnium punctatum	1	56.2	5.8	0.0002
	Rhodobryum roseum	1	31.2	4.0	0.0002
	Rhytidiadelphus triquet	1	48.0	5.4	0.0002
	Rosa blanda	1	18.7	3.3	0.0004
	Rubus acaulis	1	57.7	7.5	0.0002
	Rubus pubescens	1	22.4	5.3	0.0002
	Sanionia uncinata	1	40.6	4.4	0.0002
	Scorpidium scorpioides	1	59.4	5.4	0.0002
	Scutellaria galericulata	1	10.6	3.9	0.0072
	Sphagnum squarrosum	1	12.5	2.8	0.0016
	Thalictrum dayscarpum	1	74.5	6.4	0.0002
	Thelypteris palustris	1	9.4	2.6	0.0110
	Thuidium delicatulum	1	89.2	7.2	0.0002
	Viburnum trilobum	1	16.8	3.4	0.0004
	Viola blanda	1	22.4	5.4	0.0002
	Viola renifolia	1	43.4	5.1	0.0002
	Viola septentrionalis	1	62.5	5.5	0.0002
	Aster puniceus	2	8.6	3.1	0.0118
	Carex bebbii	2	12.2	3.5	0.0018
	Conyza canadensis	2	19.6	13.7	0.0266
	Corylus cornuta	2	10.2	4.3	0.0111
	Diervilla lonicera	2	12.3	7.2	0.0215
	Polytrichum juniperinum	2	16.2	10.1	0.0238
	Taraxacum officinale	2	9.4	2.5	0.0126
	Trientalis borealis	2	17.1	6.0	0.0018
	Carex eburnean	3	11.6	6.4	0.0292
	Ceratodon purpureus	3	27.7	20.2	0.0012
	Corydalis sempravirens	3	18.4	8.0	0.0008
	Polytrichum commune	3	26.0	12.7	0.0004
	Prunus virginiana	3	6.8	2.8	0.0434
	Vaccinium angustifolium	3	30.3	17.2	0.0002
	Lycopodium dendroideum	4	16.7	8.5	0.0048
	Picea mariana	4	14.6	9.6	0.0476
	Rosa acicularis	5	11.2	4.2	0.0076
	Aster macrophyllus	5	20.3	10.0	0.0024
	Carex canescens	5	8.2	3.9	0.0386
	Epilobium angustifolium	5	30.1	20.3	0.0014

Treatment	Indicator Species	Max region	Observed indicator value (IV)	Mean randomized indicator value	p - value (sig.≤ 0.05)
Fire	*Hieracium caespitosum*	5	9.8	5.3	0.0362
	Hieracium aurantiacum	5	9.6	5.0	0.0322
	Polygonum cilinode	5	10.6	6.3	0.0480
	Populus tremuloides	5	10.2	5.9	0.0500
	Rubus idaeus	5	18.8	14.1	0.0390

Appendix VI – Common indicator species by microhabitat

Indicator species occurring at more than one site type within the same microhabitat (1 = riparian, 2 = ecotone, 3 = upland, 4 = clearcut edge, 5 = clearcut/burn matrix).

Indicator species present in more than one treatment	Indicator Species	Region (Max group)
All treatments	*Athyrium felix-femina*	1
	Aulocomnium palustre	1
	Brachythecium rivulare	1
	Bromus ciliatus	1
	Calamagrostis canadensis	1
	Carex intumescens	1
	Climacium dendroides	1
	Fissidens spp.	1
	Galium asperellum	1
	Lycopus uniflorus	1
	Mertensia paniculata	1
	Mitella nuda	1
	Mnium marginatum	1
	Mnium spp.	1
	Plagiomnium	1
	Rhizomnium pseudopunctatum	1
	Rhizomnium punctatum	1
	Rhodobryum roseum	1
	Rubus acaulis	1
	Rubus pubescens	1
	Scorpidium scorpioides	1
	Thalictrum dayscarpum	1
	Thelypteris palustris	1
	Thuidium delicatulum	1
	Viburnum trilobum	1
	Viola blanda	1
	Viola renifolia	1
	Viola septentrionalis	1
	Picea mariana	5
Reference and Buffer	*Alnus incana*	1
	Aquilegia brevistyla	1
	Brachythecium velutinum	1
	Campanula aparinoides	1
	Carex lasiocarpa	1
	Carex spp.	1
	Cornus stolonifera	1
	Dicranum spp	1
	Equisetum sylvaticum	1
	Salix spp.	1
	Viburnum edule	1
	Diervilla lonicera	4
Reference and Fire	*Agrostis gigantean*	1

Indicator species present in more than one treatment	Indicator Species	Region (Max group)
Reference and Fire	*Chamaedaphne calyculata*	1
	Graminoid spp.	1
	Mentha arvensis	1
	Scutellaria galericulata	1
Buffer and Fire	*Alnus crispa*	1
	Carex trisperma	1
	Euryhynchium pulchellum	1
	Galium triflorum	1
	Onoclea sensibilis	1
	Phegopteris connectilis	1
	Rhytidiadelphus triquet	1
	Aster macrophyllus	5
	Epilobium angustifolium	5
	Hieracium aurantiacum	5
	Lycopodium digitatum	5
	Rubus idaeus	5