Wildfire Patterns in Western Boreal Canada

Healthy Landscapes Research Series Report No. 8

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Abstract

The idea of using natural patterns as guides for policy and practices continues to gain favour in forestland management. Towards this, research activities in the boreal have been focusing on the patterns and processes of wildfires over the last 15-20 years. This project is the fourth phase of a long-term research study looking at natural wildfire patterns across the forested areas of Alberta and central Saskatchewan. This phase involved creating mortality maps from an additional 76 fires across Alberta, generating fire events from wildfires, and completing meta-analyses for all 129 fires.

The event-defining algorithm defined an outer boundary for each *event* representing the general area of influence of each fire. This generated two types of remnants: *island remnants* as mapped by the original mortality maps, and 2) *matrix remnants* created by the algorithm, largely representing corridors between disturbed patches. The algorithm captured the entire area of each wildfire into a single event for all but five wildfires.

Fires shared many key pattern characteristics across the entire data range, although some of the specifics varied. The average proportional area in remnants within each event averaged 41%, and was unrelated to event size. This finding challenges the notion of the boreal as having a so-called stand-replacing fire regime. Less than 10% of the samples in this study technically qualify as *stand-replacing*, and at least as many would be considered *stand-maintaining* based on mortality figures.

One of the previously undocumented patterns noted in this study was the presence of multiple disturbed patches within individual wildfires. About half of all events had multiple disturbed patches, the number of which increased significantly with the log of event size. Furthermore, there tended to be one very large disturbed patch, the size of which was unrelated to event size. The shapes of both events and disturbed patches increased significantly as the log of event area increased, although disturbed patch shapes were more convoluted. Most of the island remnant area was partially disturbed, and small (<1 ha) islands account for 81% of the island numbers, but only 5% of island area. Probability of burning differentials were greater between major land types (such as non-forested versus forested), and less so for differences in forest ages, species, or

densities. The highest probability of burning within an event was for conifer leading forest, and the lowest for non-forested vegetation and areas of wet soil moisture.

More specific burning patterns fell into one of two regimes; one for the Boreal and Shield (B-S) and another for the Foothills and Mountains (F-M). B-S fires had significantly more disturbed patches for a given event size, and thus significantly greater area in matrix remnants relative to the F-M region. The largest disturbed patch in the B-S accounted for an average of 84% of the event area compared to 67% for the F-M region. Fires in the B-S region tended to have significantly greater areas of partially burned remnants (27%) relative to the F-M region (14%).

The findings suggest that boreal wildfires, and by extension boreal landscapes, are far more structurally complex than previously thought. This complexity has significant implications in terms of known, critical boreal ecosystem functions. It also forces us to reconsider what “old” forest is in the boreal, and how it is created.

In terms of management implications, considering the degree to which we have (until 10-15 years ago) relied on the classic stand-replacing model of the boreal as a management guide, most forest management has almost certainly been simplifying the boreal landscape via management. Details aside, finding ways of re-introducing this complexity will be challenging, and require significant levels of collaboration between regulators and managers. However, these challenges create some new opportunities for shifting roles and responsibilities that some believe are necessary if we are to shift to more of an ecosystem-based approach.

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1.0 Introduction

Under the auspices of ecosystem management, the study of natural disturbance patterns towards the development of natural pattern emulation strategies has been growing steadily over the past 20 years (Hansen et al. 1991, Franklin 1993, Landres et al. 1999, Bergeron et al. 2007). There is perhaps no greater opportunity to take advantage of coarse filter disturbance patterns as management guides than in the boreal forests of Canada (Bergeron et al 1999) due to the similarity between the natural and existing cultural disturbance regimes. Forest management plans, park management plans, Provincial management regulations, Provincial and Federal monitoring initiatives, and certification agencies now include indicators and targets that reference natural boreal wildfire patterns (BC MoF and BC Environment 1995, Bergeron et al. 1999, OMNR 2001, FSC 2004).

Adopting natural disturbance patterns as management guides for any form of disturbance activities create some significant opportunities. For example the indicators associated with disturbance patterns are potentially measured with available data at low cost, easily linked to biological thresholds, and correspond directly to management activities such as harvesting and prescribed burning. Terms such as “future forest condition” conveniently re-phrase management objectives in terms of cumulative landscape patterns. The philosophical backdrop of using disturbance patterns as a proxy for ecosystem function is also consistent with the tenets of ecosystem-based management (Grumbine 1994).

Whether we take full advantage of the potential of a natural pattern approach to forest management depends in large part on how well we understand spatial patterns. At fine scales, surprisingly little research is available. Although a significant amount of research has been conducted on fire severity, most of it focuses on fire behaviour parameters (Kobziar et al. 2006, Michaletz and Johnson 2008), biomass consumption (Key and Benson 2006, French et al 2008, Soverel et al. 2010), or overall residual levels (Eberhart and Woodard 1987, Delong and Tanner 1996). To date, no one has taken a holistic approach to describing and understanding wildfires as spatio-temporal events.

The objective of this research is to understand and quantify patterns and (where possible also the) processes of wildfire burning within historical, natural forest fires in Alberta and Saskatchewan towards defining the natural range of variation for a series of intermediate scale indicators for planning and monitoring purposes. This research builds on, and includes all of the data from previous wildfire pattern studies conducted by Andison (2004, 2005, 2006a, 2006b).

2.0 Objectives

The immediate goal the proposed study is to *understand the spatial patterns and the processes of natural wildfire events across Alberta and Saskatchewan.*

The ultimate goal of this project is to *provide to land management partners in Alberta and Saskatchewan the basic knowledge required to develop more robust and sustainable management guidelines / models for managing for historical disturbance patterns at operational scales.*

The objectives of this project towards those goals are:

- To quantify the patterns of survival within historical, natural wildfires across Alberta and Saskatchewan.
- To identify, and predict for the purposes of planning, how wildfire regimes manifest themselves in terms of pattern response at fine to intermediate scales across most of the western Canada boreal landscape.
- To make research results available in a variety of forms to maximize exposure to new knowledge and their potential implications to forest management within Alberta and Saskatchewan.
- To provide links to related management tools that will help partners integrate this new information into operational reality.

3.0 Background

Three phases have already been completed on this topic:

1) Phase 1 - FRI: 24 wildfire mortality maps from west-central Alberta collected in 1997-99, and final reports and manuscript (blue dots in Figures 1 and 2)

2) Phase II - Bandaloop: 10 wildfire mortality maps from western Saskatchewan collected in 1999-2001, and final reports (dark green squares in Figures 1 and 2)
3) Phase III - Bandaloop: 19 wildfire mortality maps in Saskatchewan collected in 2003-05, and final reports (light green squares in Figures 1 and 2)

The data from each of the first three phases was collected using identical methods. Phase I data was analysed and final reports / manuscripts completed on behalf of the Foothills Research Institute (Andison 2003, 2004, and 2012). The data from phases II and III were combined and analysed by Bandaloop on behalf of Mistik Management, Saskatchewan Environment and the Saskatchewan Forest Centre (Andison 2005, 2006a, 2006b).

This project is the fourth phase of this western boreal wildfire pattern research. It expands the study area to include all of the forested area of Alberta, and the analyses include the fire data from all previous phases.

Table 1. Project History Summary

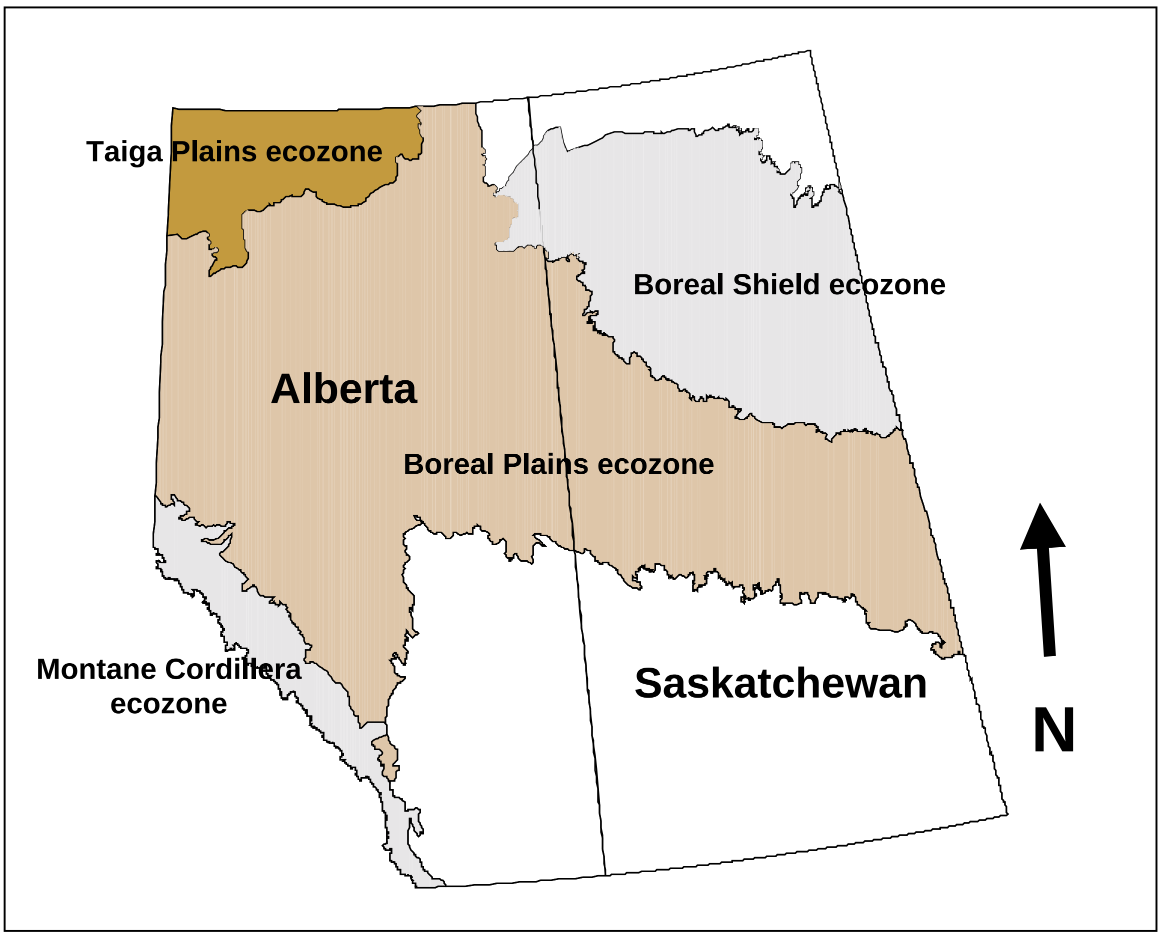
Project Phase	Timeline	Final Fire Sample Size	Collaborators
Phase I	1997-98	24	• Foothills Research institute • Alberta Environment & Sustainable Resource Development • Hinton Wood Products • Jasper National Park • Alberta Newsprint Company • Weyerhaeuser Canada
Phase II	1999-2001	10	• Mistik Management
Phase III	2003-2005	19	• Saskatchewan Forest Centre • Saskatchewan Environment
Phase IV	2006-2012	(see ahead)	• Foothills Research Institute • Forest Resource Improvement Association of Alberta • Alberta Sustainable Resource Development • Alberta Pacific (AlPac) • Daishowa-Marubeni International (DMi) • Tolko International (High Level) • Slave Lake Pulp • Manning Diversified Forest Products

4.0 Methods

4.1 Study Area

The study area for phase IV of this project encompasses the Boreal Plains and Taiga Plains ecozones of Alberta and Saskatchewan (Figure 1). However, given the width of transition zones, some samples technically occur in both the Montane Cordillera and Boreal Shield ecozones (see the shaded grey areas in Figure 1). The final sampling area thus includes at least parts of the Rocky Mountains, Foothills, Boreal Plains, and Boreal Shield ecological zones

Figure 1. The target study area is the Boreal Plain and Taiga Plain ecozones.



according to provincial classifications (Alberta Environmental Protection 1994) and the Montane Cordillera, Boreal Plains, Boreal Shield, Taiga Shield and Taiga Plains according to the Canadian classification system (Wilken 1986).

The study area is in excess of 100 million hectares, which means that the climate, vegetation and topography are highly variable stretching from the mountains to the west, to the forest-grassland interface to the south, and to the sub-arctic taiga to the north. A brief summary of these attributes is given in Table 2 (see Wilken 1986 for details).

Forest wildfires are the dominant natural disturbance agent of the study area. In general, very large fires every 50-200 years of moderate to high severity are responsible for the vast majority of the historical landscape patterns (Johnson 1992, Ward and Tithecott 1993, Andison 2000). While many of the details remain unknown, the sheer size and complexity of the study area strongly suggests there may be several distinct fire regimes (di Castri and Hansen 1992, Delong et al. 2003, Boulanger et al. 2012).

Table 2. Summary of the characteristics of the ecozones in the study area.

Ecozone	Climate	Precip (mm)	Mean Temps. (^{o}C)	Vegetation	Topography
Montane Cordillera	Mountain and interior. Long cold winters & short warm summers.	600-1200	Jan -8 to -18 July 13 to 18	Lichen and shrubs, lodgepole pine, alpine fir, Englemann spruce	Steep to very steep with few flat valleys.
Boreal Plains	Humid continental. Cold winters, Moderately warm summers	300 - 500	Jan -18 to -22 July 12 to 18	White spruce, black spruce, lodgepole pine Tamarack, white birch, trembling aspen, balsam poplar.	Flat to gently rolling.
Boreal Shield	Continental. Long cold winters, short warm summers.	400-700	Jan -10 to –20 July 15-18	White and black spruce, jack pine, balsam fir, tamarack, white birch, trembling aspen, balsam poplar.	Rolling
Taiga Plains	Cold, semi-arid. Short cool summers and long cold winters.	300-400	Jan -22 to –35 July 10 to 15	White spruce, black spruce, tamarack, lodgepole pine, white birch, trembling aspen, balsam poplar	Flat to gently rolling.
Taiga Shield	Subartic continental. Short summers, and long very cold winters.	175-200	Jan -18 to –28 July 8 to 18	Open grown black spruce, alder, willow, tamarack, white spruce, balsam fir, trembling aspen.	Rolling.

4.2 Data

The raw date requirements for the data collection phase of the project were identical to those of the first three phases of this research. The ideal raw data for this are accurate, precise, high-resolution images of tree survival immediately following a naturally occurring forest fire. This eliminates the use of existing data sources such as forest inventories, satellite imagery, and ecological mapping because they lack the necessary precision and accuracy, and/or they not likely to capture patterns of natural wildfires immediately following the event. The best possible raw data for studying natural wildfire patterns originate from high-resolution aerial photographs taken within a few years of naturally occurring fires that have burned in naturally vegetated areas (*i.e.*, no pre-burn cultural features, and no post-burn modifications such as salvage logging).

Towards generating these data (for the remainder of the un-sampled forested area of Alberta completed in phase I), a list of candidate fires was generated using the Forest Fire History Maps of Alberta, 1931 to 1983 (Delisle and Hall, 1987), later augmented by various forms of digital and hard copy fire records. All fires were then censored according to the following selection criteria:

1) Post-fire photography should be available at no less than 1:20,000 within five years after the fire.
2) Pre-fire photography should be available at no less than 1:20,000 within ten years prior to the fire.
3) No physical overlap with other fires in the database.
4) No significant known fire fighting activities or post-fire salvage logging before post-fire photos were taken.

Exceptions to rules 1 and 2 were made to include some fires with lower-resolution post-fire photos of exceptional quality, or when combinations of availability and scale of photos were available.

4.3 Spatial Data Interpretation

A set of first-generation post-burn aerial photos (phase I) or negatives (phases II, III, and IV) to cover the area of each of the fires was acquired along with the relevant base maps for geo-referencing. The number and location of aerial photos / negatives from each source, for the area covering each fire (before and after burning as appropriate) were identified and purchased from the respective provincial air photo repository, provincial archives, or the Forestry Canada library as necessary. The pre-fire photo coverage included a greater number of photos per fire to allow for a buffer width of 100 m around the perimeter of each fire area.

The technology used to interpret the fires from phase I differ from those used for fires from phases II, III, and IV. Since digital map products and software were unavailable at the time of phase I, fire mortality maps were generated using stereo photo pairs, a stereoscope, and a magnifier light. To avoid the feature and scale distortion associated with uncorrected aerial photos, only the centre 10cm of each photo was used for interpretation. In most cases, individual trees could be identified. To minimize bias, the same person interpreted all fires.

For the remaining fires, integrated Mapping Technologies Inc. (Vancouver, BC) used a photogrammetric scanner at a resolution of 10um to digitally scan each photo and/or negative. Base map information from provincial 1:50,000 and 1:250,000 NTS base maps and the most recent (1:12,500) inventory photos were used as controls (*e.g.*, spatial references for data within a Geographic Information System). These data were all imported into an ISM DiAP viewer version of Softcopy Systems, a stereo imaging on-screen digitizing software program. A single interpreter completed all three phases of this work.

To reduce the potential for bias, the same person searched for and generated the sample fire list for the all four phases of the project. This individual also manually interpreted the fires in phase I (as above) and trained the person doing the interpretations for phases II-IV. This second interpreter was also a veteran of fire mapping.

The only interpretation required to create the mortality maps was mortality level, based on percentage of tree death. Where trees were absent, the percentage of burnt area was used as a surrogate. Six classes of mortality were recognized in the raw data:

0 = no loss of crown.
1 = 1-25% loss of crown
2 = 26-50% loss of crown
3 = 51-75% loss of crown

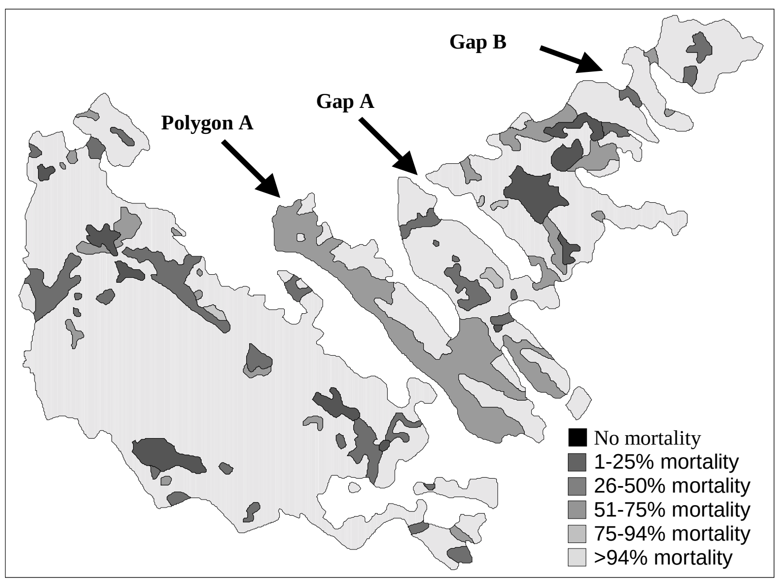
4 = 76-94% loss of crown
5 = >94% loss of crown

The minimum required mapping resolution was 0.02 ha, which represented a clump of about four live trees. Given the high quality of photos used, many polygons much smaller than this were eliminated from the dataset. An example of a fire mortality map is shown in Figure 2.

Mortality map polygons were digitized (as required), corrected for curvature, geo-referenced, and saved as overlays within ArcMap 8.3 (ESRI 2002). The data were then saved as shapefiles, and transferred to ArcView 3.2 Geographic Information System and Spatial Analyst (ESRI 1999).

The outermost boundaries, or "shells", of a subset of these fires were buffered outwards by 100 m to generate the areas requiring pre-fire photo interpretation. Pre-fire vegetation for a subset of fires was then interpreted using either the Alberta Vegetation Inventory (Government of Alberta 2007) or the Saskatchewan Forest Vegetation Inventory standards (McLaughlan 2003). Fortunately, the two provincial forest inventory standards are very similar.

Figure 2. Example of a fire mortality map.



All completed files were delivered as ARCVIEW compatible shapefiles as UTM zones 11, 12, or 13, using NAD83 projection. These data were also converted into comma-delimited ASCII raster data using 50 m pixels.

Where possible, we sought to collect additional data regarding ecological classification data, creek and water layers, digital elevation models (DEM), and fire weather. Unfortunately, the availability of these data, at the time, was inconsistent.

4.4 Creating a Spatial Language

Before proceeding with any analysis, I needed to define a precise and consistent series of spatial definitions. Spatial languages for forest vegetation (Government of Alberta 2007), ecosystem classification (Wilken 1986), soils, geology, agriculture, and wetlands are the foundation of sound management and monitoring practices. The best ones share several key characteristics:

1) Inclusive, but discreet,
2) Hierarchical,
3) Science based,
4) Relevant to the scales of the process (es), and,
5) Simple and easily measured and mapped.

Unfortunately, although several provincial forest management (BC MoF and E 1995, OMNR 2001) and certification agencies (FSC 2004) have developed natural pattern guidelines, none include a formal spatial language for describing disturbance patterns with the qualities listed above. Towards developing such a language, the following three rules were applied to the raw spatial data:

1) Each polygon was classified as either ***remnant*** (<95% mortality) or ***disturbed*** (>94% mortality) (the green boxes in Figs. 3 and 4).
2) The boundaries of adjacent remnant polygons were then dissolved to create spatially contiguous ***island remnants*** (Figs 3 and 4).
3) The boundaries of all spatially contiguous polygons (regardless of mortality level) were dissolved to create ***disturbed patches*** (Figures 3 and 4).

Figure 3. Summary of the spatial language developed for this study.

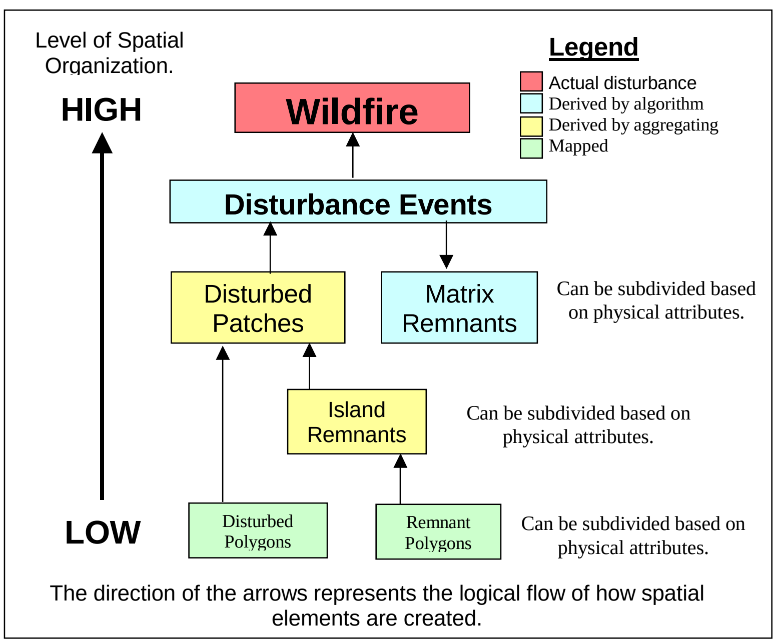
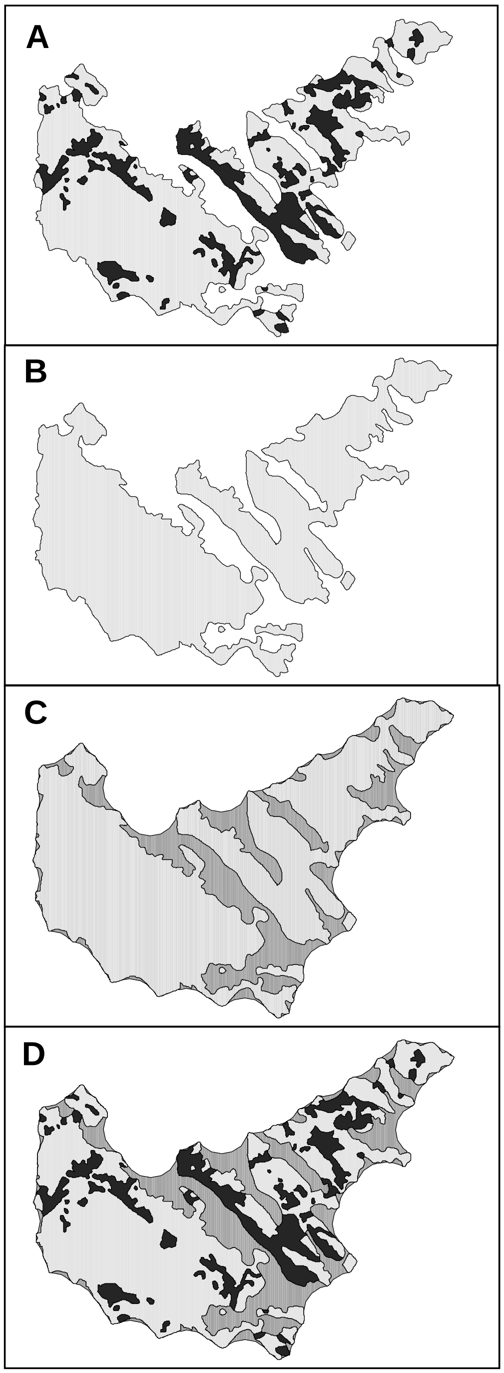


Figure 4. To create a disturbance event: A) start with a mortality map, B) dissolve all disturbed and island polygon boundaries, C) buffer out distance X, and then back in distance X, and D) add back all island polygons.



At this point the spatial language still had two deficiencies. First, the language was not inclusive because many of the fires have multiple disturbed patches, likely as a result of spotting (Albini et al. 2012). Spot fires, caused by embers flying ahead of the main fire, are common phenomenon in Alberta (Kill et al. 1977) and beyond (Porterie et al. 2007). A robust spatial language for disturbance patterns should gather any multiple disturbed patches associated with an individual fire event into a single spatial entity.

The second shortcoming of the spatial language so far was that it was neither consistent nor objective because it assumed that the only legitimate form of a remnant is that which could be easily mapped. For example, gap A in Figure 2 is only 60 m wide, but the peninsula of undisturbed area behind it is over 800m long. This area would be a 10 ha island remnant if the fire killed just a few trees across this gap. Most would agree that this would be a fire remnant physically and functionally. However, undisturbed gap B in Figure 2 is only 100 m deep and 150 m wide. It is less clear whether this gap, or what portion of this gap, should be considered a remnant (and thus within the fire boundary). Similarly, the large polygon noted at C in Figure 2 is only partially burned. This area might be counted as a *residual* in one study, a *feathered edge* by a second study, and ignored entirely by a third. These seemingly small differences in spatial definitions translate into significant differences in pattern metrics (Andison 2012), potentially defeating the purpose of using NRV as objective management guides.

I resolved both shortfalls by applying a buffering algorithm as follows:

- Buffer the outside of all disturbed patches distance X.
- Buffer the inside of the resulting polygon(s) from step 1 distance X and fill in any interior "holes" that this may generate. The resulting spatial entity is a ***disturbance event***, and the new polygons generated within events, and between disturbed patches are called ***matrix remnants*** (Figures 3 and 4).

This resolved all outstanding spatial language issues. First, it gathered multiple disturbed patches together into a single spatial entity – the event. Second, it objectively generated a second type of remnant that is still physically attached to the surrounding landscape matrix - matrix remnants (Figures 3 and 4). To help identify the buffer width to apply to the algorithm, I calculated for 10 buffer widths from 50 to 500m:

a) The percent of fires with one event (which is ideally 100%) and,
b) The average percent of the area of each fire accounted for by the largest event (which is also ideally 100%).

4.5 Analyses

Two scales of analyses were completed; landscape and event. At the landscape scale, I wanted to test for significant differences in burning patterns associated with major ecological zones (Boulanger et al. 2012). Based on the findings from these landscape-scale tests, I then summarized for each regime event shapes, disturbed patch density, disturbed patch shape, largest disturbed patch index or LPDI (the percentage of the largest disturbed patch relative to the total disturbed area), the proportional area of matrix remnants, the proportional area of island remnants, the proportional area of total remnants, the proportional area of residuals by mortality class, island remnant sizes, and island remnant shapes. For most of these metrics, I tested the relationship to event size (SYSTAT 2009).

For a subset of these data for which pre-burn data were available, I then tested for relationships between the pattern metrics, and the probability of burning based on pre-burn conditions such as the percentage of hardwood and softwood leading forest, soil moisture regime, and forest age-class, and forest density using SYSTAT (SYSTAT 2009). From that, I then generated estimated fire return intervals for each vegetation / land type.

5.0 Results

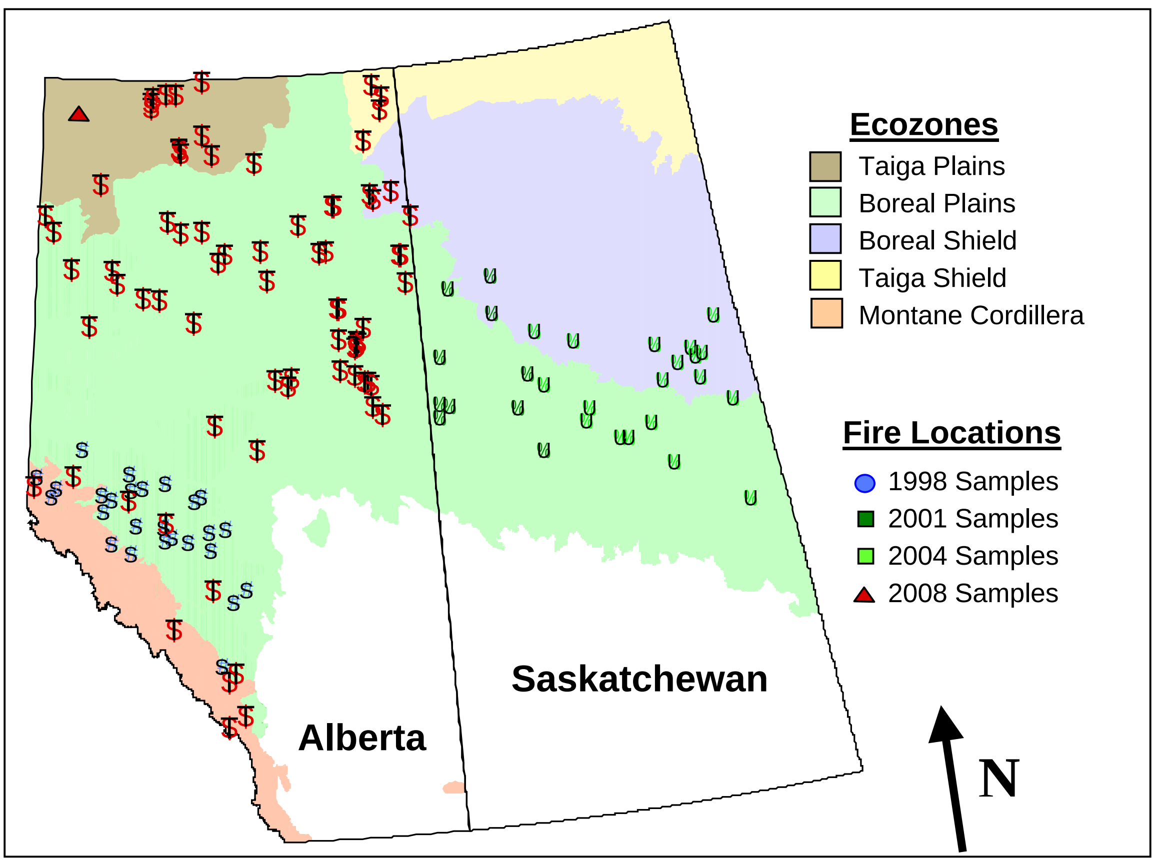
The results section is organized in the same general order as section 4. For each sub-section I have also included an associated practical question to help the reader understand the potential management implications of each section.

5.1 Fire Sampling

Question: ***Do these data, and the results in this report, represent natural burning conditions for my landscape?***

The phase IV sampling resulted in an additional 76 wildfires (Figure 5). These fires represent the only ones that represent historic, "natural" burning patterns. In other words, these are the only 129 fires found during an exhaustive search that

Figure 5. Study area and previous sample locations by Canadian ecozones.

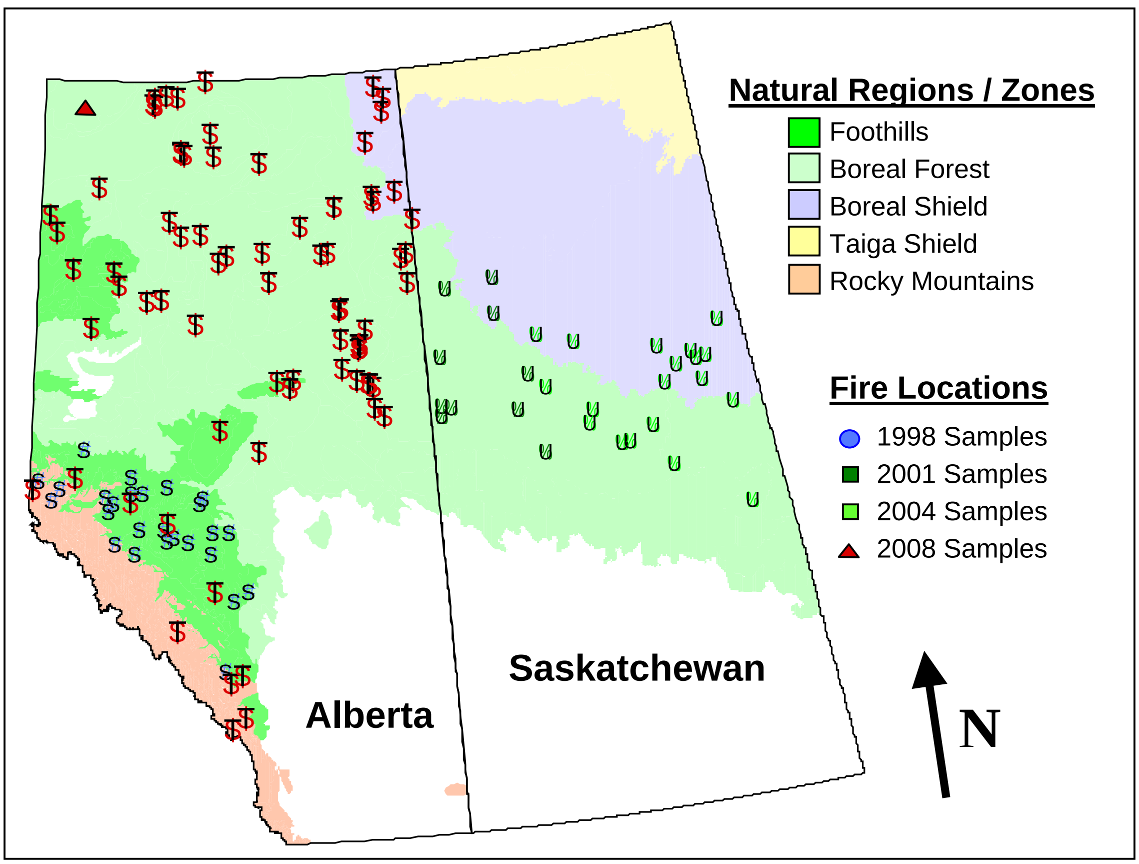


matched the criteria on these landscapes. Any effort to expand this dataset within the current study area will require a compromise on sampling criteria.

The four phases of wildfire sampling identified appropriate post-fire photos for 129 fires ranging in size from two to over 27,000 ha, totalling more than 247,000 ha. Most of the fires were in the Boreal Plains ecozone (89), with 13 in the Boreal Shield, 10 in the Montane Cordillera, 13 in the Taiga Plains, and 4 in the Taiga Shield (Figure 5). Pre-fire photos were collected for 83 of these fires. Of the 46 fires without pre-fire photos, 13 had no photos available (Tables 3-5).

According to provincial ecological zones, there were 29 fires in the Foothills natural region, ten in the Rocky Mountains, 57 in the Boreal Forest, and four in the Canadian Shield (shown as the Boreal Shield in Figure 6). For Saskatchewan, there were 17 fires in the Boreal Plains zone (shown as Boreal Forest in Figure 6), and 12 in the Boreal Shield zone. To standardize the scales of the two provincial systems, Figure 6 shows the fire samples within the ecological zones of Saskatchewan, and the natural regions of Alberta.

Figure 6. Study area and sample locations by Province and Natural Regions.



Note that the samples taken for this fourth phase of the study (the red triangles in Figures 5 and 6) were all within Alberta, including some within, and south of, the original phase I 1998 samples. Phase IV sampling was designed to create a sample that was representative of the entire forested areas of Alberta, and the 'working forest' of Saskatchewan (*i.e.*, those areas that are now being actively managed). See Tables 3-5 for details.

Table 3. Summary of Alberta Phase I Fire Samples.

Name	Fire Size (ha)	Year (season)	Ecozone	Natural Region	Post-Fire Photos		Pre-Fire Photos	
					Year(s)	Scale(s)	Year(s)	Scale(s)
March Head Creek	28	1970 (s)	BP	F	1970	15,840	1950	15,840
Little Smoky	37	1950 (s)	MC	RM	1951	15,840	1949	40,000
Open Creek A	40	1949 (sp)	BP	F	1951	15,840	U	U
Cutbank River	59	1956 (s)	BP	F	1957	15,840	1950	15,840
Little Sundance Creek	162	1956 (sp)	BP	F	1961	15,840	1950	15,840
Berland River North	194	1956 (s)	BP	F	1957	15,840	1950	15,840
Pembina River	197	1961 (sp)	BP	F	1961	15,840	1950	15,840
Open Creek	203	1958 (sp)	BP	F	1958, 1962	15,840, 31,680	1951	15,840
Wolf River	215	1961 (sp)	BP	F	1961	15,840	1957	15,840
Erith River	222	1956 (s)	BP	F	1957	15,840	A	A
Bigoray River	278	1958 (sp)	BP	F	1961	15,840	1950, 1957	40,000, 15,840
Athabasca	365	1974 (s)	BP	F	1978	15,000	1970	15,840
Prairie Creek	409	1961 (sp)	MC	RM	1967, 1970	31,680, 18,000	1958, 1960	15,840
Sheep Creek	439	1961 (s)	MC	RM	1974, 1993	21,120, 40,000	1950, 1952	40,000, 15,840
Brule	634	1946 (s)	BP	F	1951	15,840	U	U
Tony Creek	680	1950 (s)	BP	F	1950	15,840	U	U
Rat Creek	697	1956 (sp)	BP	F	1957	15,840	1952	15,840
Timber Creek	1,164	1970 (f)	MC	RM	1970	15,840	1958	15,840
Horse creek	1,165	1956 (sp)	BP	F	1958	15,840	1951, 1952	15,840
Lick Creek	1,225	1982 (s)	MC	RM	1982, 1983	15,840, 60,000	1975, 1980	21,120, 60,000
McLeod River	3,635	1956 (sp)	BP	F	1958	15,840	1949, 1950, 1951	40,000, 15,840
Gregg River	8,887	1956 (s)	BP	F	1963	31,680	A	A
Smith Creek	9,117	1956 (s)	BP	F	1963	31,680	A	A
Moose Creek	15,909	1956 (sp)	BP	F	1957	15,840	A	A

Year (season): sp=spring, s=summer, f=fall.
Ecozones: BP = Boreal Plains, MC = Montane Cordillera.
Provincial Natural Region: F=Foothills, RM= Rocky Mountains.
Pre-Fire Photos: A=available but not collected, U=unavailable.

Table 4. Summary of Saskatchewan Phases II and III Fire Samples (bold italic fires are phase II Fires).

Name	Fire Size (ha)	Year (season)	Ecozone	Natural Region	Post-Fire Photos		Pre-Fire Photos	
					Year(s)	Scale(s)	Year(s)	Scale(s)
Ridge	24	1981 (sp)	BP	MBU	1981	12,500	1980	12,500
McArther	84	1984 (s)	BS	CRU	1866	12,500	1971	15,840
Rail	87	1984 (s)	BP	MBU	1984	12,500	1979	12,500
Overflow	88	1984 (s)	BP	MBU	1988	12,500	1979	12,500
Rainbow	113	1986 (sp)	BP	MBU	1986	12,500	1979	12,500
Kerr	139	1982 (s)	BP	MBL	1982	12,500	1970	15,840
Two Forks	219	1981	BP	MBU	1986, 1987	12,500	1973, 1975	15,840, 12,500
McDonald lake	336	1966	BS	CRU	1968, 1971	15,840, 31,680	U	U
Preston Lake	373	1967	BS	CRU	1972	15,840	1947, 1951	31,680
Bear river	413	1974	BP	MBU	1974, 1976	12,500	1962, 1963	15,840
Sixty	470	1981 (s)	BP	MBU	1981	12,500	1970	15,840
Dillon Lake	551	<1978	BP	MBU	1978 / 1981	12,500	1959, 1960	15,840
Brett	552	1977	BP	MBU	1977	12,500	1970	15,840
Bow River	592	1970	BP	MBU	1974, 1975	12,500	1963	15,840
Sandfly	651	1968	BS	CRU	1968	15,840	U	U
Thunder Mountain	667	1970	BP	MBU	1974, 1975	12,500	1964	15,840
Kidd Lake	778	1967	BS	CRU	1967	15,840	1946, 1947	15,840, 25,000
Montreal Lake	829	1970	BP	MBU	1974, 1975	12,500	1961, 1963	15,840
Tower Island	976	1981 (sp)	BS	CRU	1982	12,500	1970	15,840
Contest #2	1,282	1981 (sp)	BS	CRU	1982, 1983	12,500	1972	15,840
Alfred	1,428	1981 (s)	BS	CRU	1981	12,500	1971	15,840
Northwood Lake	1,920	1980 (sp)	BS	CRU	1982	12,500	1972	15,840
Pitching Lake	3,573	1956	BS	CRU	1958	15,840	1947	15,840
Darieu Lake	4,087	1980 (sp)	BS	CRU	1982, 1983	12,500	1972	15,840
Guilloux Lake	5,796	1964	BS	CRU	1968, 1972	15,840	1947, 1951	31,680
Falling Horse	5,820	1979	BP	MBL	1979	12,500	1974, 1975	12,500
Elk	6,680	1983 (sp)	BP	MBU	1986	12,500	1965, 1971	15,840
Harry Lake	10,163	1980 (s)	BP	MBU	1980	12,500	1968	15,840
Carlton	27,287	1980 (sp)	BP	MBU	1982	12,500	1970	15,840

Year(season): sp=spring, s=summer.

Ecozones: BS= Boreal Shield, BP = Boreal Plains.

Provincial Ecoregion: CRU = Churchill River Upland, MBL = Mid-Boreal Lowland, MBU = Mid-Boreal Upland

Pre-Fire Photos: U=unavailable.

Table 5a. Summary of Alberta Phase IV Fire Samples.

Name	Fire Size (ha)	Year (season)	Ecozone	Natural Region	Post-Fire Photos		Pre-Fire Photos	
					Year(s)	Scale(s)	Year(s)	Scale(s)
Chelsea	2	1971 (sp)	BP	B	1971	15,840	A	A
Algarita	9	1971 (sp)	BP	B	1971	15,840	A	A
Fawcett	22	1971 (sp)	BP	F	1971	15,840	1970	15,840
Pantom Crag	22	1971 (s)	MC	RM	1971	15,840	1957	15,840
Little Boiler	23	1971 (sp)	BP	B	1971	15,840	A	A
James	23	1970 (s)	TP	B	1970	21,120	1964	31,680
Glover	24	1974 (sp)	BP	B	1974	15,840	1970	24,000
Trail	25	1956	BP	F	1958	15,840	1950	15,840
Barber	25	1979 (s)	BP	B	1979	15,000	A	A
Leggo	29	2004	TS	CS	2004	20,000	A	A
Flett	29	2004	TS	CS	2004	20,000	A	A
Arrow	31	1956	BP	F	1957	16,840	1952	15,840
Hilltop	31	1971 (sp)	BP	F	1971	5,400	1971	21,120
Jackpine	38	1978 (sp)	BP	B	1978	15,000	U	U
Brewster	44	1971 (sp)	BP	F	1971	15,840	1969, 1957	31,680, 15,840
Christina	52	1971 (sp)	BP	B	1971	15,840	A	A
Tanghe	55	2002	BP	B	2002	20,000	A	A
Muskeg	59	1971 (sp)	BP	B	1971	15,840	A	A
Turtle	60	1970 (sp)	TS	CS	1970	15,840	A	A
Filion	63	1982 (sp)	BP	B	1982	15,000	A	A
Clayton	70	1979 (s)	BP	B	1979	15,000	1966	13,680
Abraham	70	2001	MC	RM	2001	15,000	A	A
Little Sun	71	1974 (sp)	BP	F	1974	15,840	A	A
Hangingstone	87	1971 (sp)	BP	B	1971	16,080	1967, 1951	31,680, 15,840
Steephill	88	1990 (sp)	BP	B	1991	20,000	1989	20,000
Letter Y	92	<1953	TP	B		31,680	U	U
Logan	100	1981 (s)	BP	B	1983	10,000	1978	15,000
Steepbank	111	1970 (sp)	BP	B	1970	15,840	A	A
Wallah	116	1949	BP	B	1951	15,840	U	U
Halverson	148	194?	BP	F	1952	15,840	U	U
Francis Peak	173	1956	MC	RM	1956	15,840	1952	15,840
Algar	175	1971 (sp)	BP	B	1971	15,840	1951	15,840
Stone Creek	195	1958	BP	B	1958	15,840	A	A
Liege	205	2002	BP	B	2002	20,000	1996	15,000
Sputina	243	1982 (sp)	BP	B	1983	15,000	A	A
McIvor	247	1969 (s)	BP	B	1969	15,840	1967	31,680
Indefatigible	249	1967 (s)	MC	RM	1972	21120	1957	15,840
Livock	249	1982 (sp)	BP	B	2004	20,000	1978	15,000
Nihani	253	1954	MC	RM	1967	31,680	1944	19,000

Year (season): sp=spring, s=summer.

Ecozones: BP = Boreal Plains, MC = Montane Cordillera, TP = Taiga Plains, TS = Taiga Shield.

Provincial Natural Region: F=Foothills, RM= Rocky Mountains, B= Boreal, CS = Canadian Shield

Pre-Fire Photos: A=available but not collected, U=unavailable.

Table 5b (con't). Summary of Alberta Phase IV Fire Samples.

Name	Fire Size (ha)	Year (season)	Ecozone	Natural Region	Post-Fire Photos		Pre-Fire Photos	
					Year(s)	Scale(s)	Year(s)	Scale(s)
Perry	301	2004	TP	B	1979	15,000	1991	20,000
Airstrip	309	1967 (s)	TP	B	1979	20,000	1954	40,000
Richardson	331	1979 (s)	BP	B	2003	20,000	A	A
Sand	347	1978 (s)	BP	B	1970	15,840	1977	15,000
Cone	380	2003	BP	B	1971	15,840	1994	20,000
Ells	491	1971 (sp)	BP	B	1971	15,840	1952	15,840
Wabasca	511	1971 (sp)	BP	B	1971	15,840	1962, 1952	31,680, 15,840
Pistol	551	1971 (sp)	BP	B	1970	15,840	1966, 1952	31,680, 15,840
Firebag	597	1971 (sp)	BP	B	1970	11,200	A	A
High Hill	628	1970 (sp)	BP	B	1970	15,840	1951	15,840
Tepee	738	1970 (s)	TP	B	1951	15,840	1964	31,680
Woodman	745	1970 (sp)	TS	CS	1978	15,000	1950	40,000
Joker	751	1949	BP	B	1950, 1949	15,840, 40,000	U	U
House	796	1978 (sp)	BP	B	1978	15,000	1973, 1978	15,000, 50,000
Tepee 49	852	1949	BP	B	1943	15,840	U	U
May	872	1978 (sp)	BP	B	1971	15,840	A	A
Pakwanutik	1,102	1950	BP	B	1951	15,840	U	U
Boiler	1,121	1971 (sp)	BP	B	1970	18,000	1967, 1951	31,680, 15,840
Meikle	1,164	1952/53	BP	F	1985	10,000, 20,000	A	A
Dizzy	1,177	1970 (s)	TP	B	1971	15,840	A	A
Pelican	1,597	1982 (sp)	BP	B	1971	15,840	1970, 1978	15,000, 25,000
Yates	1,847	1971 (s)	TP	B	1970	21,120	A	A
Baseline	1,885	1971 (sp)	BP	B	1970	21,120	1951	15,840
Eva	2,169	1970 (s)	TP	B	2076	15,000	1964	31,680
Ponton	2,201	1970 (s)	TP	B	1969	15,840	1953	15,840
Little Rapids	2,473	1975 (s)	TP	B	1970	15,840	1964	31,680
Margeurite	2,794	1969 (s)	BP	B	1976	15,000	A	A
Robert	3,752	1970 (sp)	BP	B	1962	15,840	A	A
Buffalo Head	4,226	1972 (sp)	BP	B	1944	19,000	1962	31,680
Sousa	4,356	1955	TP	B	2006	20,000	1953	15,840
Waiparous	5,039	1939	BP	F	2004	20,000	A	A
Levellers	5,624	2006	BP	B	1958	15,840	A	A
Keane	6,960	2004	BS	B	1971	15,840	1983	25,000
Sloan	8,917	1956 (sp)	BP	F	1952	15,840	A	A
Caribou	9,934	1971 (sp)	TP	B	1970	21,400	A	A
Ferris Flats	12,005	1948	BP	B	1952	15,840	U	U
Tourangeau	27,052	1970 (s)	TP	B	1970	21,400	A	A

Year (season): *sp=spring, s=summer.*

Ecozones: BP = Boreal Plains, MC = Montane Cordillera, TP = Taiga Plains, TS = Taiga Shield.

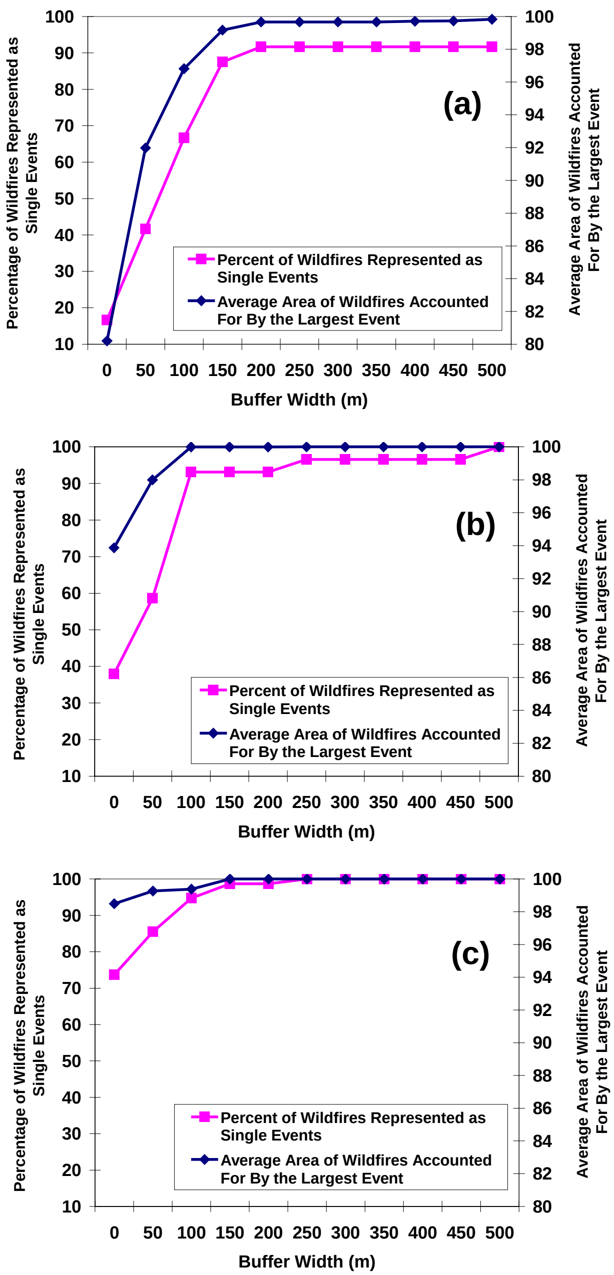
Provincial Natural Region: F=Foothills, RM= Rocky Mountains, B= Boreal, CS = Canadian Shield

Pre-Fire Photos: A=available but not collected, U=unavailable

5.2 Wildfires vs. Events

Question: ***What buffer width is used in this study to create disturbance 'events' as described in section 4.4?***

Figure 7. Disturbance event buffer width summary for (a) Alberta1, (b) Saskatchewan, and (c) Alberta2 datasets.



Even seemingly subtle changes to the spatial language of natural disturbance events can have significant impacts on pattern outcomes, potentially resulting in confusion or disagreement over NRV objectives, or even outright rejection of NRV principles (Andison 2012). Under ideal conditions, one would hope to capture all parts of every wildfire using the buffering algorithm described in Section 4.4. Unfortunately, that was not realistic. The data suggested that buffer widths of no less than 1,300 m were required to gather each fire into a single spatial event. However, the buffer test results did reveal some useful thresholds.

The best buffer width to apply for the wildfire event definitions (as described in Section 4.4) differed between the three major sampling areas. For the original phase I sampling, there was no significant improvement of either the proportion of fires with a single event or the average area accounted for by the largest event beyond 200 m (Figure 7a).

For the Saskatchewan samples, at 100 m, the average proportional area accounted for by the largest event was nearly 100%, although there were still several wildfires with multiple (very small) events (Figure 7b). Similarly, the phase IV Alberta sampling suggested that a 150 m buffer captured all but a small number of very small events (Figure 7c).

Given the need for a single buffer width to describe all 129 events, I chose to use a 200 m buffer. Beyond 200 m, there was very little to be gained in terms of representing wildfires. Below 200 m, some data integrity was lost. Thus, *for the analyses to follow, a 200 m buffer width will be used to define disturbance events*. A 200 m buffer width also aligned with local observations and expertise suggesting that spotting activity of 2-300 m is fairly common (Kill *et al*. 1977, Alexander 2009), although spotting up to several km is possible. A 200 m buffer captured (the dynamics of more common) spotting distances of up to 400m.

The event-forming algorithm generated 152 events, of which 11 were smaller than five ha, which were not included in the analyses to follow.

5.3 Wildfire Regimes

Previous studies suggest that major ecological zones are the most likely to capture fire regime differences (Delong et al. 2003, Boulanger et al. 2012). The simplest way of assessing whether there were multiple disturbance regimes present in the sampling area (based meso-scale burning patterns) was to compare several key pattern metrics for significant differences between different pre-defined geographic zones such as major ecological boundaries. I chose to include five such metrics in this test; four representing different types of unburned residuals, and one representing the relative patchiness of a fire. The four residual metrics tested included; 1) percentage of event area in matrix remnants, 2) percentage of event area in island remnants, 3) percentage of event area partially burned, and 4) percentage of event area in all forms of residuals. The metric chosen to represent patchiness was the Largest Disturbed Patch Index (LDPI) based on the total disturbed area of an event.

5.3.1 Significant Interactions

Question: ***Are there any significant interactions between fire size and fire patterns that might complicate the differentiation of regional fire patterns?***

Before proceeding, it was necessary to test for any significant interactions with other event metrics that may bias the results. More specifically, previous studies have found significant relationships between the level of residuals and event size (Eberhart and Woodard 1987, Delong and Tanner 1996). Such knowledge is otherwise valuable for practitioners. To test for such relationships, I applied simple linear regression using each of the five pattern metrics described here as the independent variables against the log of event area as the independent variable. In each case, there was no significant relationship to event size (Figures 8-9). R-squared values did not exceed 0.11, and standard errors were all very high, suggesting no significant relationship.

Figure 8. The percentage of wildfire event area in matrix remnants (a), island remnants (b), partially burned areas (c) and all remnants (d) for all Alberta and Saskatchewan wildfire events.

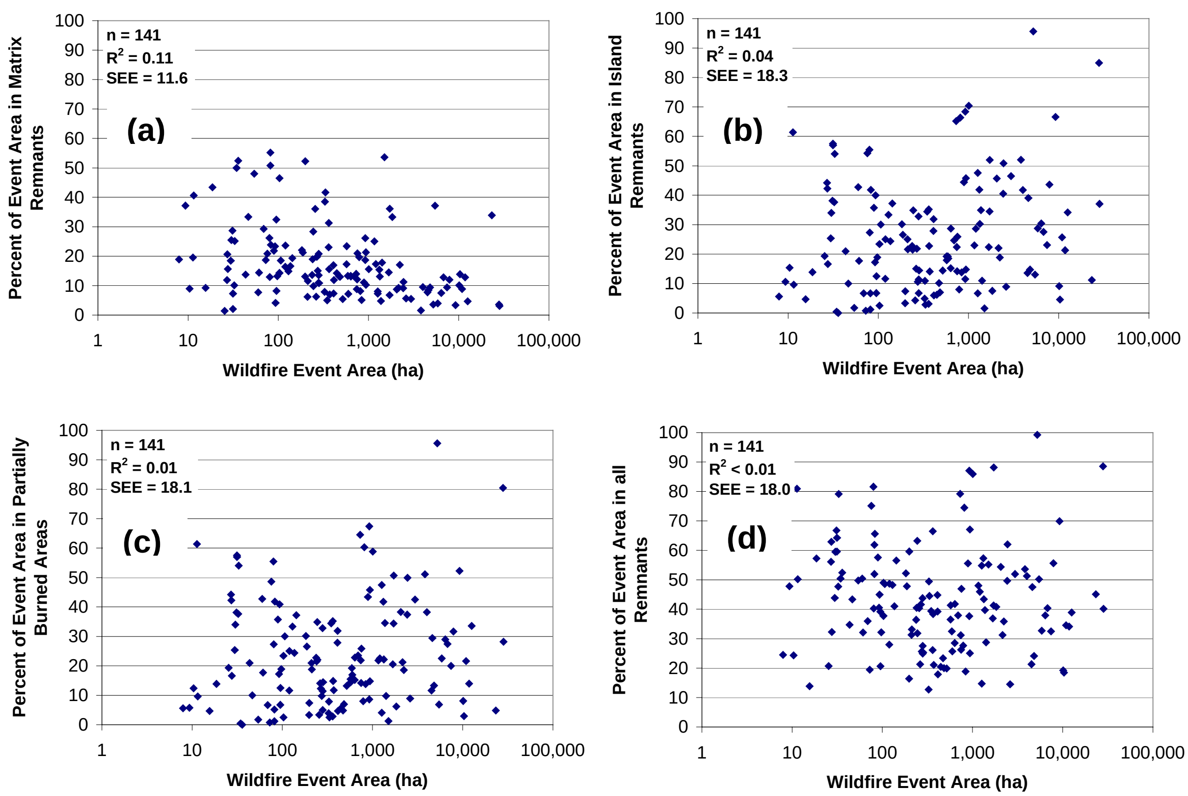
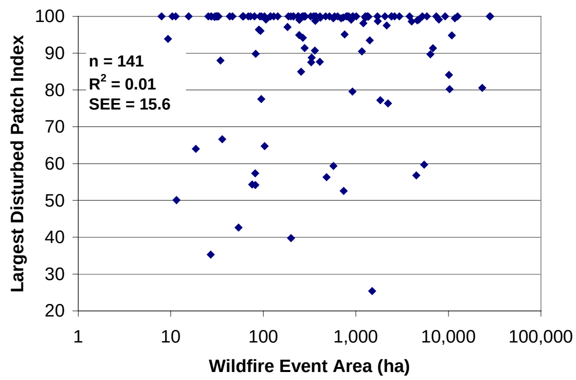


Figure 9. The largest disturbed patch index (LDPI) for all Alberta and Saskatchewan wildfire events.



5.3.2 Wildfire Pattern Differentiation by Region

Question: ***Are there significant regional differences in burning patterns across the study area?***

Having determined that event shape is unrelated to the five metrics, I tested for the presence of different wildfire regimes based on within fire burning patterns. There were two mapping systems available for classifying wildfire burning patterns in the study area 1) the Canadian ecozones, and 2) the Provincial ecological regions (in Alberta) and zones (in Saskatchewan). I tested for significant differences based on these two spatial classification systems using ANOVA tests using the five fire pattern metrics described in section 5.3.1.

The provincial classification systems created some distinctive burning patterns. In particular, those fires within the Foothills were significantly different than those in the Boreal Forest and Canadian Shield for the proportional area in matrix remnants, island remnants, and partially burned areas (Table 6). The LDPI for the Foothills was also significantly different than that of the Boreal Forest. Fire patterns within the Rocky Mountain region were strongly associated with those from the Foothills, but differed significantly from wildfire patterns in both the Boreal Forest and Canadian Shield in terms of the proportional area in island remnants and partially burned areas. Wildfire patterns of the Boreal Forest and Canadian Shield were not significantly different from each other (Table 6).

Table 6. ANOVA p-levels for five key wildfire pattern metrics by Provincial ecological regions (Alberta) and zones (Saskatchewan).

MATRIX REMNANTS

Provincial Region / Zone	Boreal Forest	Cdn. Shield	Foothills	Rocky Mtns.
Boreal Forest	X	0.18	<0.01**	0.25
Cdn. Shield		X	<0.01**	0.15
Foothills			X	0.25
Rocky Mtns.				X

ISLAND REMNANTS

Provincial Region / Zone	Boreal Forest	Cdn. Shield	Foothills	Rocky Mtns.
Boreal Forest	X	0.56	<0.01**	0.02**
Cdn. Shield		X	<0.01**	0.01**
Foothills			X	0.93
Rocky Mtns.				X

PARTIALLY BURNED

Provincial Region / Zone	Boreal Forest	Cdn. Shield	Foothills	Rocky Mtns.
Boreal Forest	X	0.71	<0.01**	0.02**
Cdn. Shield		X	<0.01**	0.02**
Foothills			X	0.87
Rocky Mtns.				X

ALL REMNANTS

Provincial Region / Zone	Boreal Forest	Cdn. Shield	Foothills	Rocky Mtns.
Boreal Forest	X	0.87	0.81	0.33
Cdn. Shield		X	0.74	0.32
Foothills			X	0.39
Rocky Mtns.				X

LARGEST DISTURBED PATCH INDEX (LDPI)

Provincial Region / Zone	Boreal Forest	Cdn. Shield	Foothills	Rocky Mtns.
Boreal Forest	X	0.42	<0.01**	0.37
Cdn. Shield		X	0.11	0.71
Foothills			X	0.48
Rocky Mtns.				X

** Significant at the 0.10 level.*
*** Significant at the 0.05 level*

The ecozones of Canada created a different interpretation of wildfire pattern regimes. Wildfire patterns within the Taiga Shield differed from those from all of the other ecozones for the proportional area in island remnants, all remnants, and partially burned areas (Figure 7). The only other pattern of note was that the proportional area in matrix remnants within fires in the Boreal Plains differed significantly than those from both the Taiga Plains or Boreal Shield.

Overall, the evidence suggests that Foothills wildfire patterns were unique, although strongly related to those of the Rocky Mountains. The fires within the Boreal Forest and the Canadian Shield were not significantly different, and were grouped together. This division also made sense geographically.

The distinctiveness of the wildfire patterns in the Taiga Shield was diminished by the fact that there were only four samples, none of which were larger than 750 ha (Tables 3-5). Furthermore, the four wildfires were clustered in one corner of what is a very large ecological zone. Thus, for the purposes of this study, I did not isolate the Taiga Shield fires from the rest of the sample.

Table 7. ANOVA p-levels for five key wildfire pattern metrics by Canadian Ecozones.

MATRIX REMNANTS

Canadian Ecozone	Boreal Plains	Boreal Shield	Montane Cord.	Taiga Plains	Taiga Shield
Boreal Plains	X	<0.01*	0.16	<0.01**	0.08*
Boreal Shield		X	0.13	0.82	0.99
Montane Cord.			X	0.22	0.34
Taiga Plains				X	0.91
Taiga Shield					X

ISLAND REMNANTS

Canadian Ecozone	Boreal Plains	Boreal Shield	Montane Cord.	Taiga Plains	Taiga Shield
Boreal Plains	X	0.54	0.16	0.31	0.01**
Boreal Shield		X	0.1	0.53	0.01**
Montane Cord.			X	0.08	<0.01**
Taiga Plains				X	0.03*
Taiga Shield					X

PARTIAL BURN

Canadian Ecozone	Boreal Plains	Boreal Shield	Montane Cord.	Taiga Plains	Taiga Shield
Boreal Plains	X	0.64	0.2	0.31	0.01**
Boreal Shield		X	0.15	0.47	0.01**
Montane Cord.			X	0.09	<0.01**
Taiga Plains				X	0.02**
Taiga Shield					X

ALL REMNANTS

Canadian Ecozone	Boreal Plains	Boreal Shield	Montane Cord.	Taiga Plains	Taiga Shield
Boreal Plains	X	0.15	0.09*	0.91	0.05**
Boreal Shield		X	0.44	0.48	0.01**
Montane Cord.			X	0.23	0.01**
Taiga Plains				X	0.03**
Taiga Shield					X

LARGEST DISTURBED PATCH INDEX (LDPI)

Canadian Ecozone	Boreal Plains	Boreal Shield	Montane Cord.	Taiga Plains	Taiga Shield
Boreal Plains	X	0.35	0.4	0.01**	0.83
Boreal Shield		X	0.92	0.49	0.62
Montane Cord.			X	0.67	0.61
Taiga Plains				X	0.49
Taiga Shield					X

** Significant at the 0.10 level.*

*** Significant at the 0.05 level*

5.4 Wildfire Event Burning Patterns

5.4.1 Event Shape

Question: ***(How) are disturbance event shapes related to event size?***

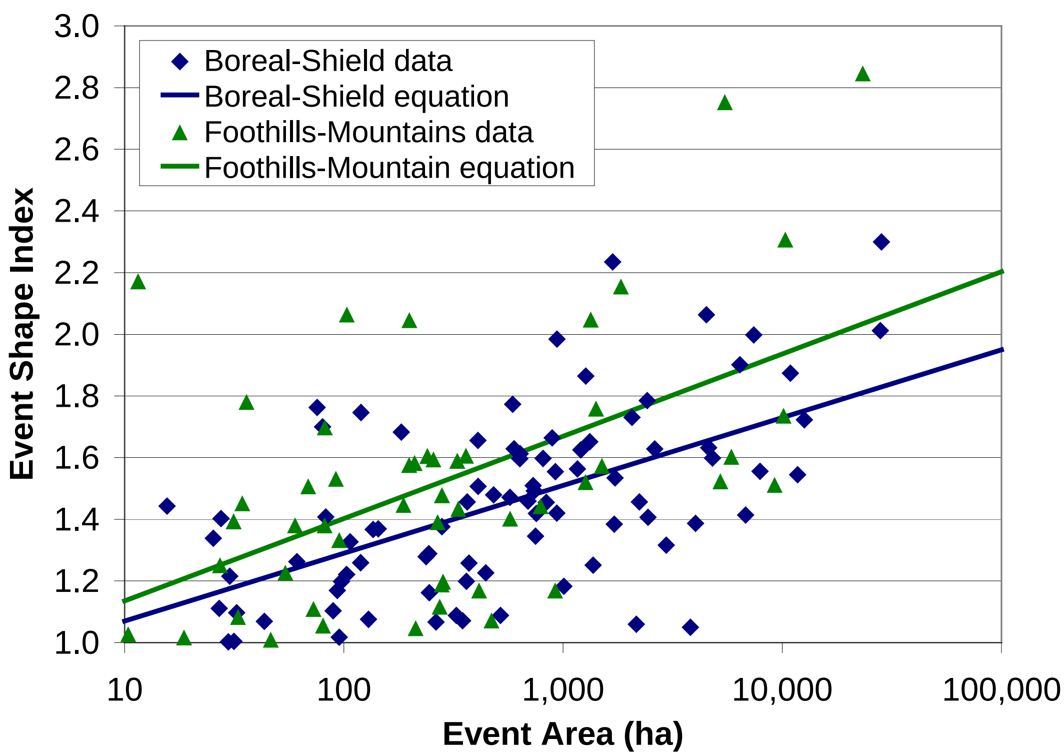
Although highly variable, wildfire event shape increased significantly with event size (Figure 10). I used regression to test / define the relationship between shape and event size for both the Boreal-Shield (B-S), and the Foothills-Mountains (F-M) regions.

$$ShapeBS = 0.85 + 0.22 \times Log(Earea)$$ n = 87, R^2 = 0.34, SEE = 0.24, F<0.01

$$ShapeFM = 0.87 + 0.27 \times Log(Earea)$$ n = 54, R^2 = 0.30, SEE = 0.35, F<0.01

Where *ShapeBS* = the shape of events in the Boreal-Shield, *ShapeFM* = the shape of events in the Foothills-Mountains, and *Earea* = event area.

Figure 10. Event shape index for Boreal-Shield and Foothills-Mountain fires.



In fact, the shape equations for B-S and F-M wildfires were not significantly different than each other. I provided separate equations for consistency with the separation of the sample in Section 5.3, but a single equation for all of the data would be justified:

$$Shape = 0.89 + 0.22 \times Log(Earea)$$ n = 141, R^2 = 0.28, SEE = 0.29, F<0.01

5.4.2 Disturbed Patch Density

Question: ***How many disturbed patches are there in an event?***

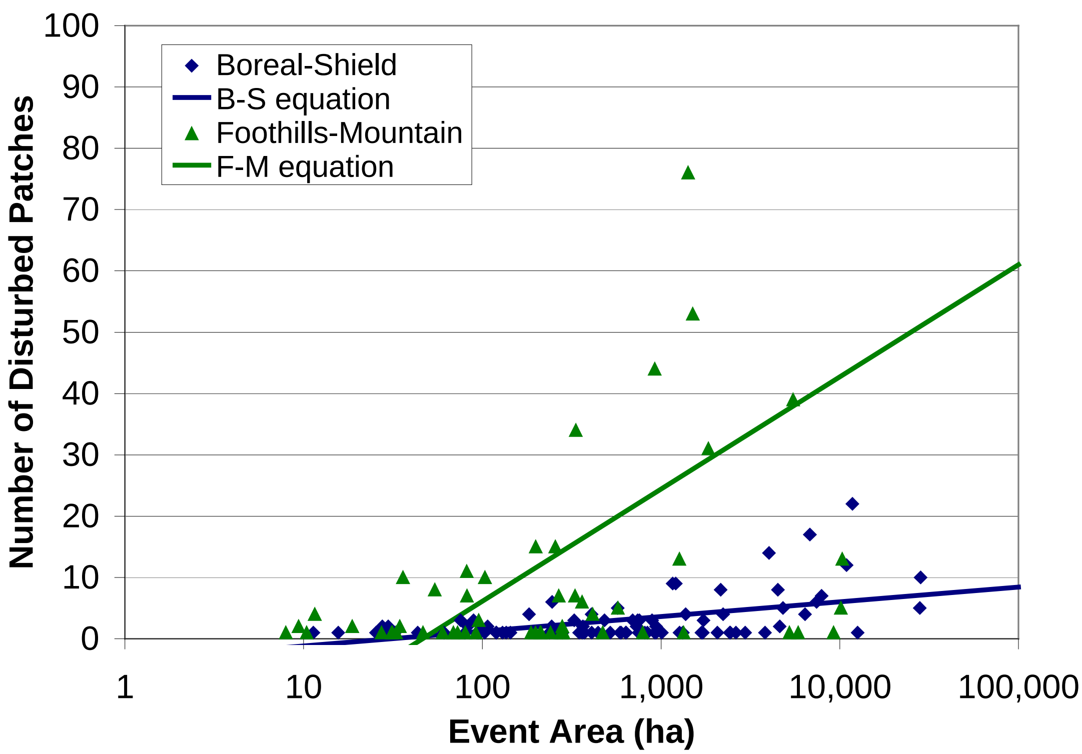
Fifty-six percent of the Boreal-Shield wildfires, and 48% of Foothills-Mountain fires consisted of only a single disturbed patch. The number of disturbed patches in each case was significantly related to event size (Figure 11).

$DPbs = -3.6 + 2.4 \times Log(Earea)$ n = 87, R^2 = 0.20, SEE = 31.6, F<0.01

$DPfm = -30.5 + 18.3 \times Log(Earea)$ n = 54, R^2 = 0.26, SEE = 3.2, F<0.01

Where *DPbs* = the number of disturbed patches per event in the Boreal-Shield, *DPfm* = the number of disturbed patches per event in the Foothills-Mountains, and *Earea* = event area.

Figure 11. Disturbed patch density for Boreal-Shield and Foothills-Mountain fires.



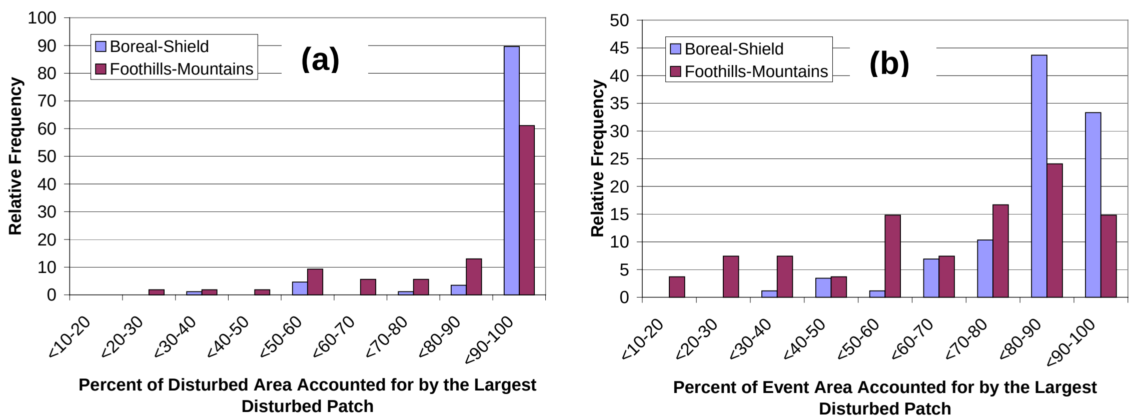
Relative to event size, the patchiness of wildfires in the two regions was significantly different. On average, a 1,000 ha fire in the F-M had 24 disturbed patches, compared to only four disturbed patches in the F-M. The average 10,000 ha fire in the F-M had 43 disturbed patches, while the same 10,000 ha fire in the R-M area only had six (Figure 11).

5.4.3 Largest Disturbed Patch

Question: ***What proportion of each event does the largest disturbed patch represent?***

As determined in section 5.3.1, there was no significant relationship between LPDI and event size. Beyond that, in general, wildfires in both regions tended to have one very large disturbed patch. Largest disturbed patch index (LDPI) can be expressed two ways; a) as a percentage of the disturbed area within an event (DA-LDPI), and b) as a percentage of the event area (E-LPDI). The average DA-LDPI for the BS was 95.9% (11.9% s.d.), which was significantly different than the 86.6% average (19.2% s.d.) for the FM area (Figure 11a). The average E-LDPI for the BS was 83.5% (11.9% s.d.), which was also significant different than the 67.0% (23.8% s.d.) for the FM (Figure 11b).

Figure 11. Two versions of the largest disturbed patch index (LDPI) as a percentage of the total disturbed area within an event (a), and the total event area (b).



The number of single-patch events did not bias the tendency towards having a single large disturbed patch. When events with single disturbed patches were removed from the DA-LDPI calculation (since they would all have an LDPI of 100%), the average was still 75.4% for the F-M, and 91.3% for the B-S.

When the data from Figure 7 is combined with that from both 5.4.2 and 5.4.3, it provides another way of considering the spatial relationships between disturbed patches. Keeping in mind that the 0 m buffer width data represents raw disturbed patch data, the dramatic increase in the number of wildfires represented by single events between the 0 m and 50 m in each case suggests that most of the disturbed patches were within 100 m of each other (which is the equivalent of a 50m buffer). Furthermore, the fact that the average proportion of the wildfire area accounted for by the largest event was over 97% confirms that very large disturbed patches were universal phenomenon within fire events. That both the Alberta2 and Saskatchewan data were almost entirely resolved with a 100 m buffer suggests that a) almost all of the disturbed patches in the B-S zones were within 200 m of each other, and b) those disturbed patches that were not within 200 m were extremely small. However, similar metrics were not observed from the Alberta1 (*i.e.*, Foothills) data until 200-250 m. This suggests that not only were natural wildfires in the Foothills-Mountains more patchy than those in the Boreal-Shield, but also that their disturbed patches tended to be more distant.

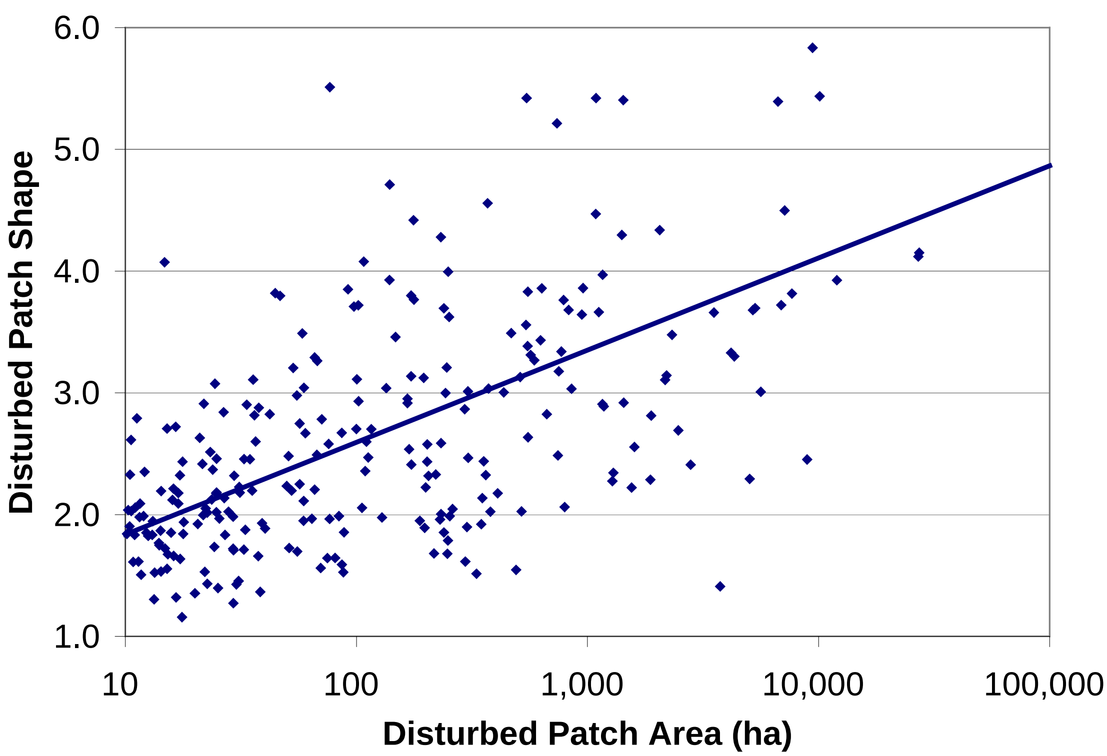
5.4.4 Disturbed Patch Shape

Question: ***What shape are disturbed patches?***

Disturbed patch shapes increased significantly as disturbed patch size increased (Figure 12).

$DPshape = 1.1 + 0.76 \times Log(DParea)$ n = 256, R^2 = 0.31, SEE = 0.92, F<0.01
Where *DPshape* = disturbed patch shape, and *DParea* = disturbed patch area (ha)

Figure 12. Disturbed patch shape relative to event area, for the study area.



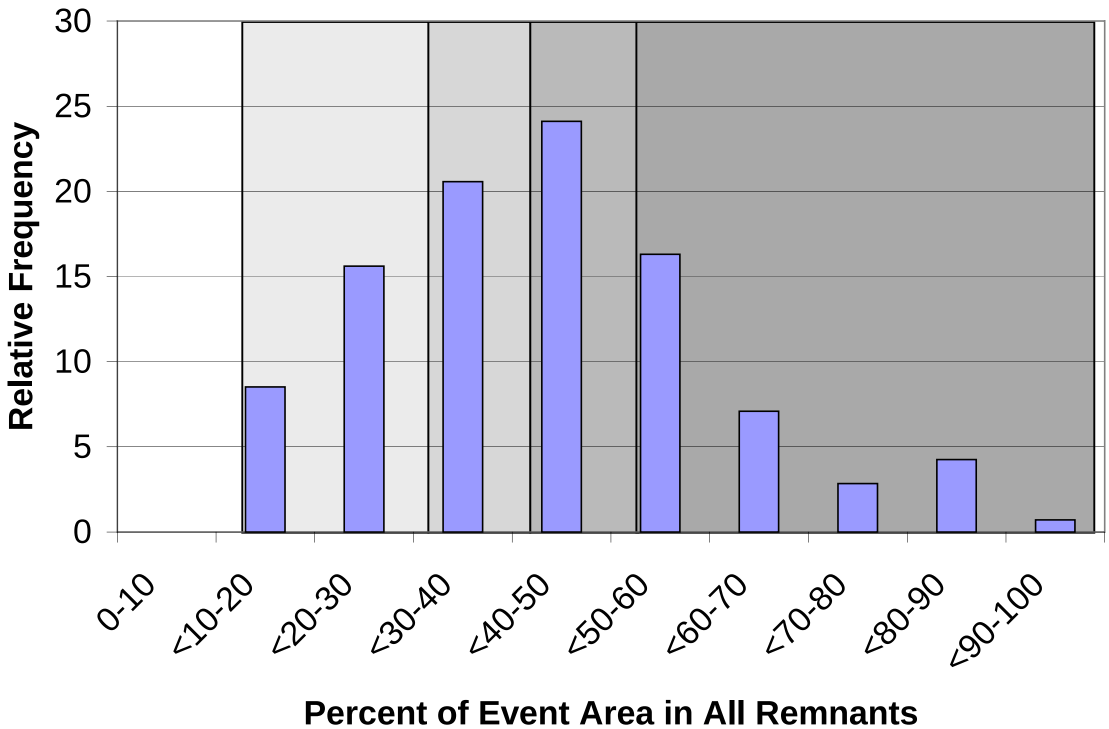
There was no significant difference between the shapes of disturbed patches within B-S fires and F-M fires. However, the shapes of disturbed patches were 2-3 times more spatially complex than those of disturbance events. For example, on average, the shape of a 1,000 ha event was 1.3, compared to 3.4 for a 1,000 ha event. The shape of a 10,000 ha event was 1.9, while the shape of a disturbed patch of the same size is 4.1.

5.4.5 Total Remnant Levels

Question: ***What areal proportion of disturbance events survives, in some form?***

As determined in sections 5.3.1 and 5.3.2 the total proportional area in unburned remnants in natural wildfires was related to neither the ecological region, nor event size. Thus, the frequency distribution shown in Figure 13 is truly the natural range of remnant levels of wildfires for all forested areas of Alberta and the southern boreal forest areas of Saskatchewan.

Figure 13. Total proportional area in unburned remnants for the study area. Quartiles are shown in progressively shaded grey boxes.



The average area that survived within all wildfires in the sample was 40.7% (s.d. = 17.9%). No fire had remnant levels less than 12%, and none more than 99%.

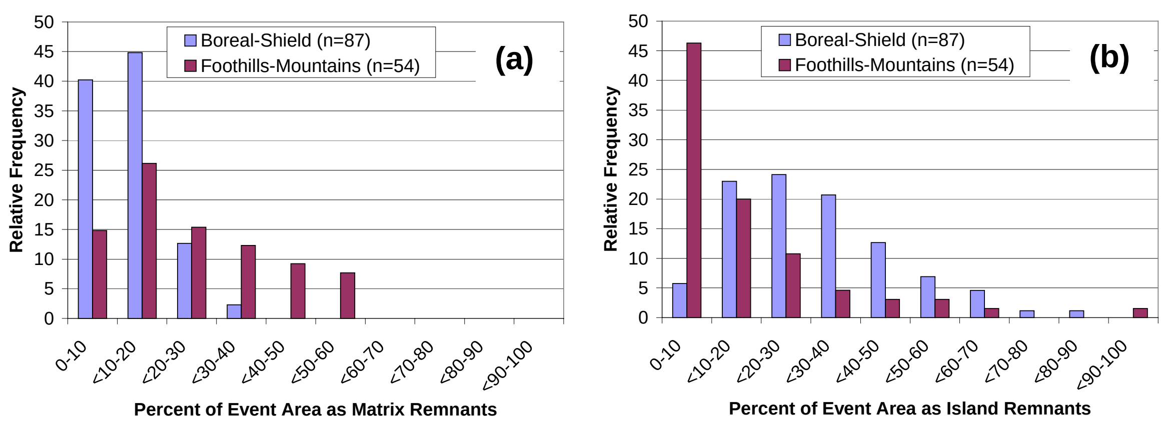
The frequency distribution of the remnant area in wildfire events was normally distributed. At one extreme, only 8% of the fires qualified as *stand-replacing* according to the 20% survival threshold identified by Brown (2000). At the other extreme, the same proportion of the fires had greater than 70% survival, which would easily qualify as *stand-maintaining* fires. The remaining 50% of the wildfire samples (e.g., the 2nd and 3rd quartiles) had 31-52% of their area in remnants.

5.4.6 Matrix and Island Remnant Levels

Question: ***What areal proportion of disturbance events survived as matrix and island remnants?***

Islands and matrix represent the two different physical manifestations of a remnant defined in this study; *islands* tended to be within disturbed patches, and *matrix* between them. Recall that the levels of matrix and island remnants differed significantly between the B-S and F-M regions. The average level of matrix remnants in the Boreal-Shield was 13.0% (n=87, s.d =6.8%), compared to 25.0% (n=54, s.d. = 15.3%) for the Foothills-Mountains (Figure 14a). Island remnants accounted for 31.0% (16.5% s.d.) of Boreal-Shield wildfires and only 17.1% (18.7% s.d.) of wildfires in the Foothills-Mountains (Figure 14b).

Figure 14. Proportional area in matrix remnants (a) and island remnants (b) by Boreal-Shield and Foothills-Mountains.



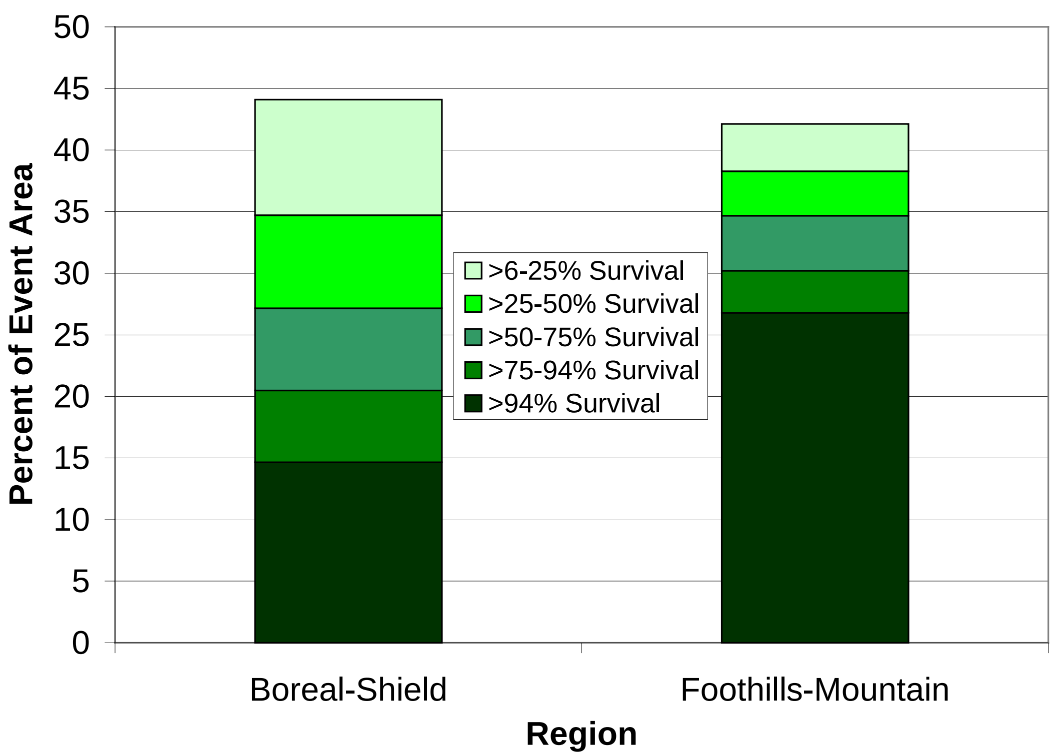
Extremely high levels of matrix remnants were rare, but did occur occasionally for island remnants. Fourteen percent of B-S fires had greater than 50% island remnant area, compared to 7% of F-M fires (Figure 14b).

5.4.7 Remnant Survival Levels

Question: ***(How) does the survival levels of remnants vary?***

Remnants can also be classified according to survival levels. Recall that matrix remnants by definition survived wildfires intact (100% burned), but island remnants were classified into one of five survival classes. Figure 15 shows what the average wildfire in each region looked like when all of the remnant data are merged.

Figure 15. Proportional area of remnants by survival levels for the B-S and F-M regions.



B-S wildfires had almost double the area in unburned remnants, largely due to the influence of matirx remnants. Island remnants that survived with no mortality accounted for less than 2% of the area of a wildfire event in both regions.
It is also interesting to note the high levels of low-survival remnants in the Boreal-Shield. Remnants with less than 50% survival accounted for almost 18% of the area of B-S fires, compared to only 7.5% for F-M fires (Figure 15). In fact, if remnants with less than 50% survival were ignored, the overall remnant levels of B-S fires would be signifcantly higher than those of F-M fires.

5.4.8 Island Remnant Shapes

Question: ***What are the shapes of island remnants?***

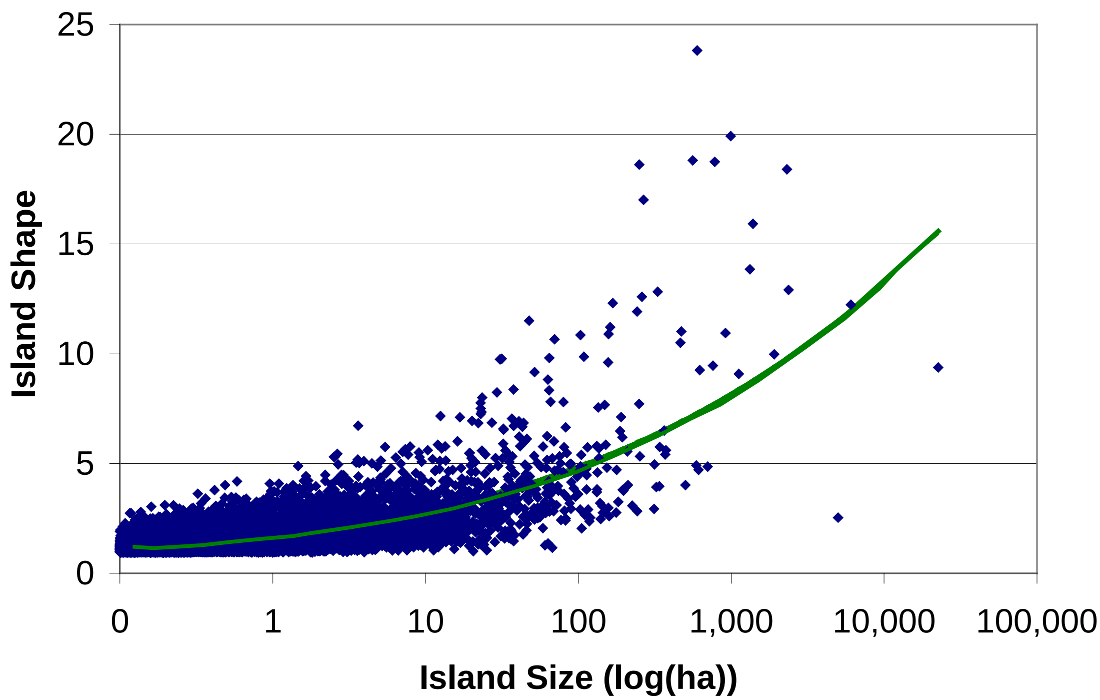
Island shapes became signficantly more complex as they increaed in size. Furthermore, all islands had highly convoluted shapes (Figure 16). For example, the average shape of islands less than 1 ha in size was 1.6, and the shape of islands 10-100 ha in size averaged 3.2. The shape of islands between 100-1,000 ha in size averaged 6.8. This means that larger islands (*i.e*, those between 100-1,000 ha) had more than double the perimter of disturbed patches of the same size, and were 5-10 times more convoluted than wildfire events of the same size.

The best-fit regressiojn equation for island shape was as follows:

$Ishape = 0.96 + 0.003 \times Iarea$ n = 18,008, R^2 = 0.47, SEE = 0.61, F<0.01

Where *Ishape* = island remnant patch shape, and *Iarea* = island remnant area (ha).

Figure 16. Island remnant sizes for islands >0.1 hectares.

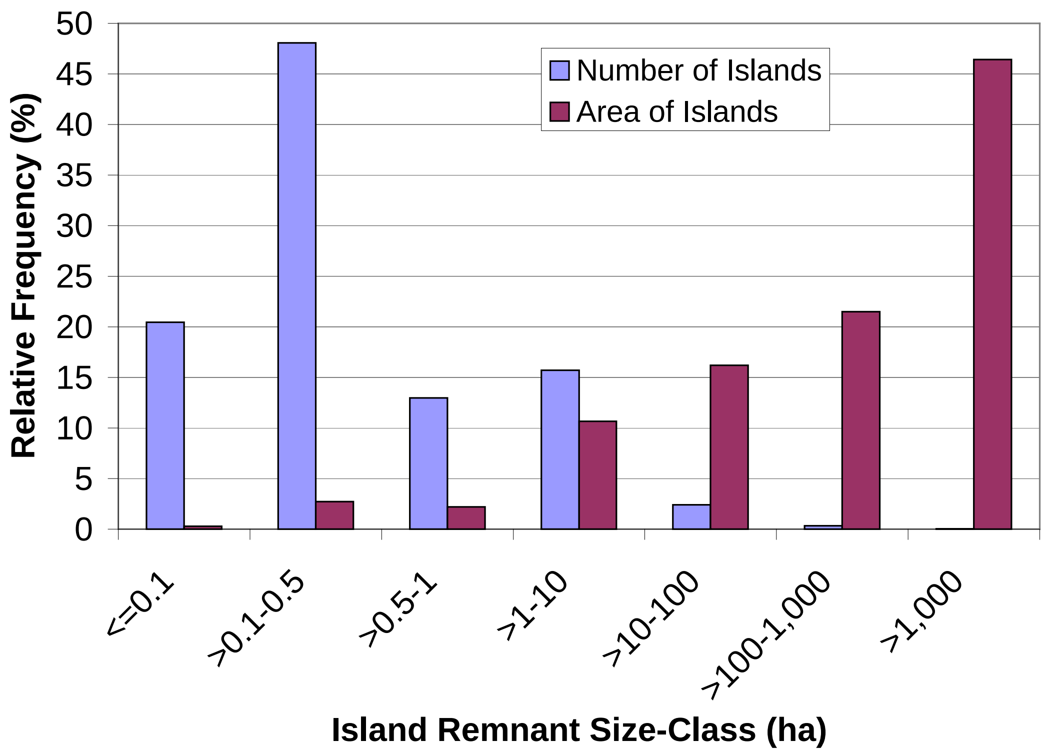


5.4.9 Island Remnant Sizes

Question: ***How big are island remnants?***

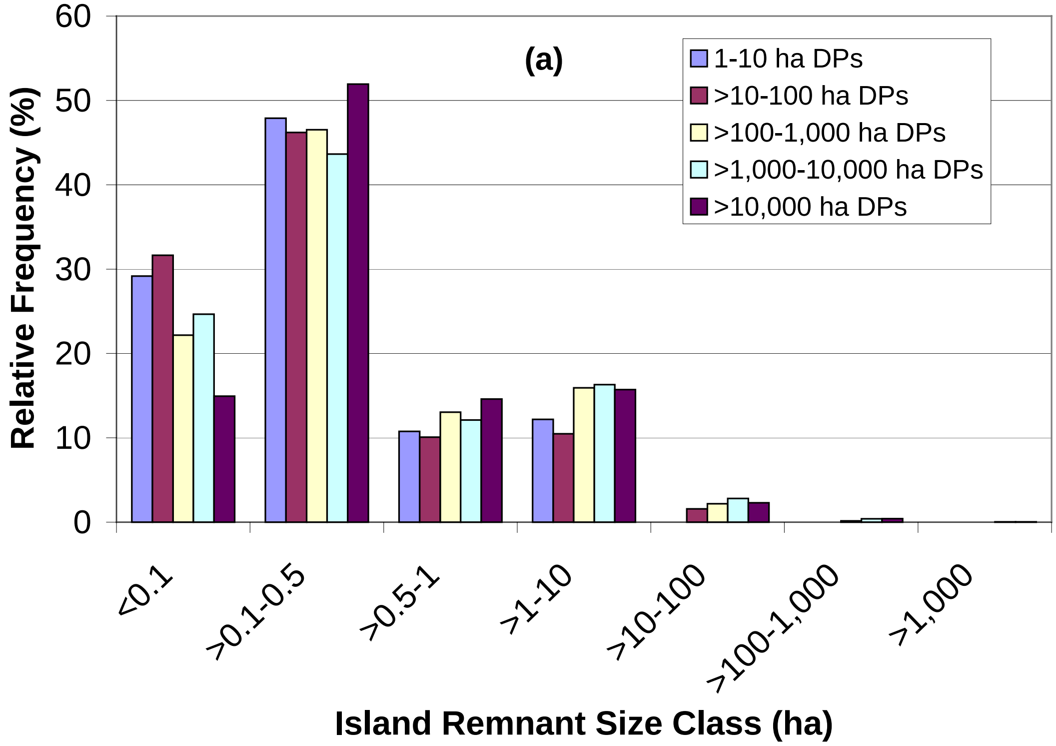
The vast majority of island remnants were very small. In fact, over 81% of the more than 23,000 islands in this study were less than 1 ha in size. However, those islands only accounted for 5.2% of the area in island remnants. On the other hand, islands larger than 10 ha accounted for almost 68% of the island remnant area, but only 0.4% of the numbers of islands (Figure 16).

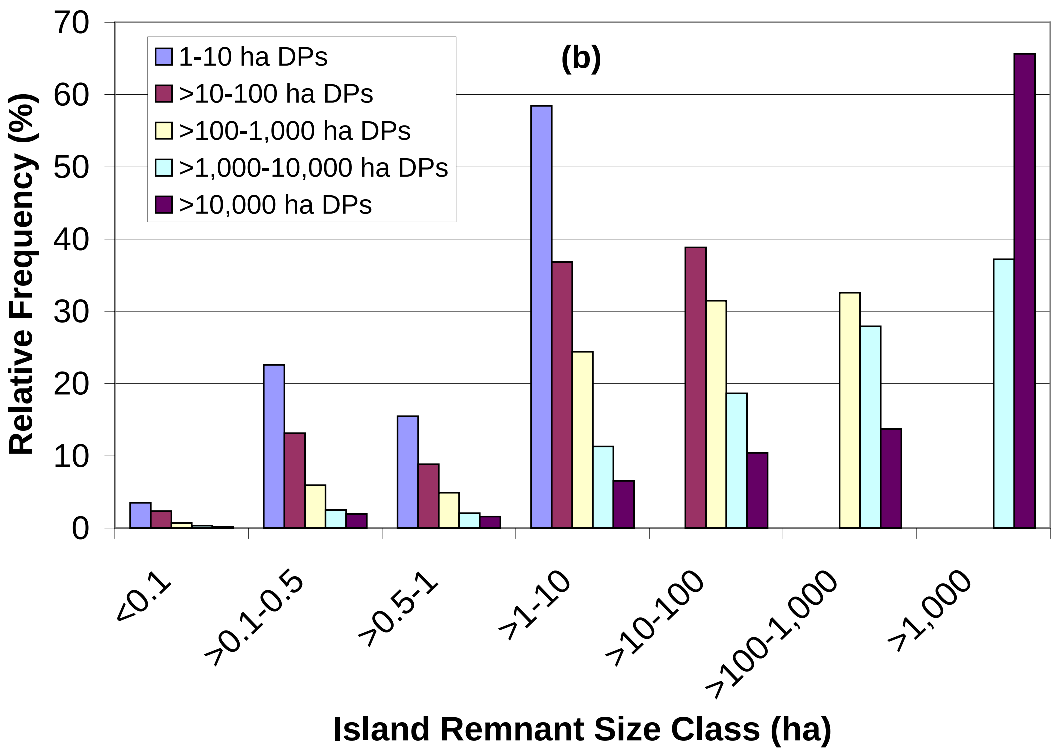
Figure 16. Summary of island remnant density and area by size-class.



While Figure 16 captures the general trend, more specific size-class distributions of islands were related to the size of the disturbed patch. Not surprisingly larger islands were more common in larger disturbed patches. For example, islands smaller than one ha accounted for 88% of island numbers, and 41% of island area, within disturbed patches smaller than 10 ha. For disturbed patches between 1,000 and 10,000 ha, small islands accounted for 80% of the number of islands, and only 5% of total island area (Figures 17a and 17b)

Figure 17. Summary of island remnant density (a) and area (b) by island size-class, and by disturbed patch size.





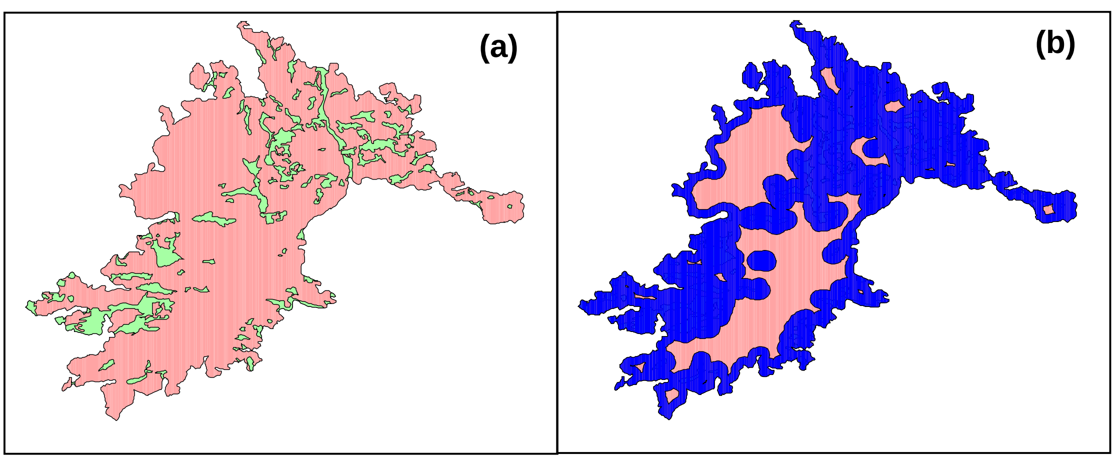
5.4.10 Disturbance Event Negative Space

Question: ***How much area within events is near live vegetation?***

The traditional way of summarizing disturbance patterns is to focus on the area, size, shape, and form of those areas which retain significant levels of live vegetation - thus the terms *remnants* and *residuals*. An alternative way of considering disturbance patterns is in terms of describing the areas that is entirely disturbed within each event – the *negative space*. This is a particularly informative technique when there are multiple sources, sizes, and types of positive space (*i.e.*, live vegetation in this case).

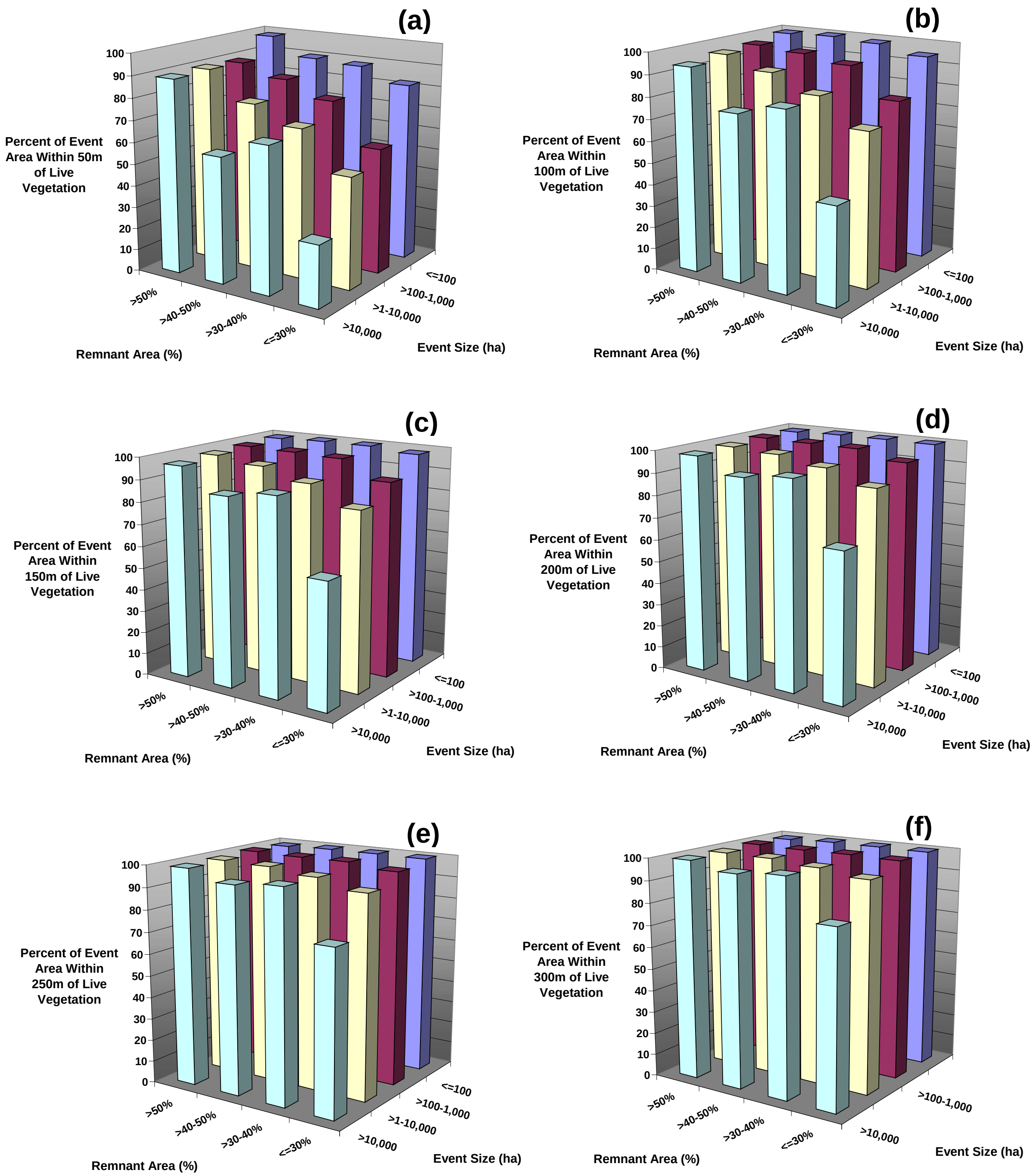
To capture the patterns of *negative space* within wildfires, I summarized the area within each event within (X) m of any form of live vegetation (either as island remnants, matrix remnants, or disturbed patch edges), where (X) equals buffer widths of 50-300m (Figure 18). I used the 30%, 40%, and 50% thresholds as approximate quartile thresholds (based on Figure 13) for overall remnant levels (Figures 19a-f).

Figure 18. Buffering out from island remnants (green areas in a), and in from disturbed patch edges creates a buffer area (blue area in b), and the negative space within an event (red area in b).



I found that on average, 77% of the area of a disturbance event was within 50 m of live vegetation, 90% was within 100m, and 95% within 150m (Figure 19a). Keep in mind that the average area in total remnants was only 41%.

Figure 19. Average percentage of event area within 50m (a), 100m (b), 150m (c), 200m (d), 250m (e), and 300m (f) to live vegetation based on event size and total remnant area



The proportional area of each event in negative space was significantly and negatively related to event size, and significantly and positively related to total remnant area. On average, more than 82% of event areas smaller than 100 ha, and/or events with more than 50% in total residuals were within 50 m of live vegetation (Figure 19a). And 95% of all small (<100 ha) and/or high survival (>50%) events were within 100 m of live vegetation (Figure 19b). All but 1% of the area of events <100 ha in size or with >50% residual were within 250 m of live vegetation.

Larger events and//or lower overall remnant areas corresponded to significantly lower negative space levels. On average, at least 80% of event areas less than 10,000 ha, and/or <40% residual levels were within 150 m of live vegetation (Figure 19c). The area within 250 m of live vegetation increased to at least 92% for the same criteria.

Very large (*i.e.*, >10,000 ha) high intensity (*i.e.*, <30% survival) wildfires were the exception to this pattern. Only 28% of their area of events >10,000 ha with less than 30% survival were within 50 m of live vegetation. Only 57% was within 150 m of live vegetation (Figure 19c), and 74% within 250 m (Figure 19e).

5.4.11 Pre-Burn Conditions

Question: ***To what degree was burn probability related to predictable land or vegetation factors?***

For the 83 fires with pre-fire data, I calculated the average probability of burning, not burning (which included matrix remnants plus class 0 islands), and partially burning (which were class 1-4 islands) within the boundary of a wildfire event using two methods of classifying the major land type (by soil moisture regime and forest vs. non-forest), and three different ways of classifying forest (by age-class, leading species, and tree density). I repeated each summary for the Foothills-Mountain (on the left side of Figures 20 and 21) and Boreal-Shield (on the right side of Figures 20 and 21) regions.

In most cases, the classes used here were created directly from fields within the inventory data. However, the Alberta vegetation inventory (AVI) and the Saskatchewan vegetation inventory (SVI) had some subtle differences that had to be reconciled. Young forest was <30 years of age, immature 30-80 years, and old >80 year of age for AVI, and young, immature+mature, and old seral-stages from SVI respectively. Dry soils included *dry* soil moisture from AVI, and D, F, and MF Smr from the SVI; wet soils included *wet* soils from AVI and VW and W Smr from SVI, and moist soils were *moist* soils from AVI, and all the remaining Smr classes from SVI. For both AVI and SVI, the first tree species listed determined whether a polygon was hardwood or softwood leading. Low tree density was a density A stand for AVI, and <30% crown closure for SVI, and high tree density was a density D stand in AVI, and >70% crown closure for SVI. Forest versus non-forest was determined using a combination of fields. All polygons with at least one tree species were forested, and those with a non-forested H-type in SVI, and a naturally non-forested type in AVI were non-forested.

Figure 20. Average probability of burning, partially burning, or surviving wildfire by major vegetation type for the F-M (a) and B-S (b) regions, and soil moisture regime for the F-M (c) and B-S (d) regions.

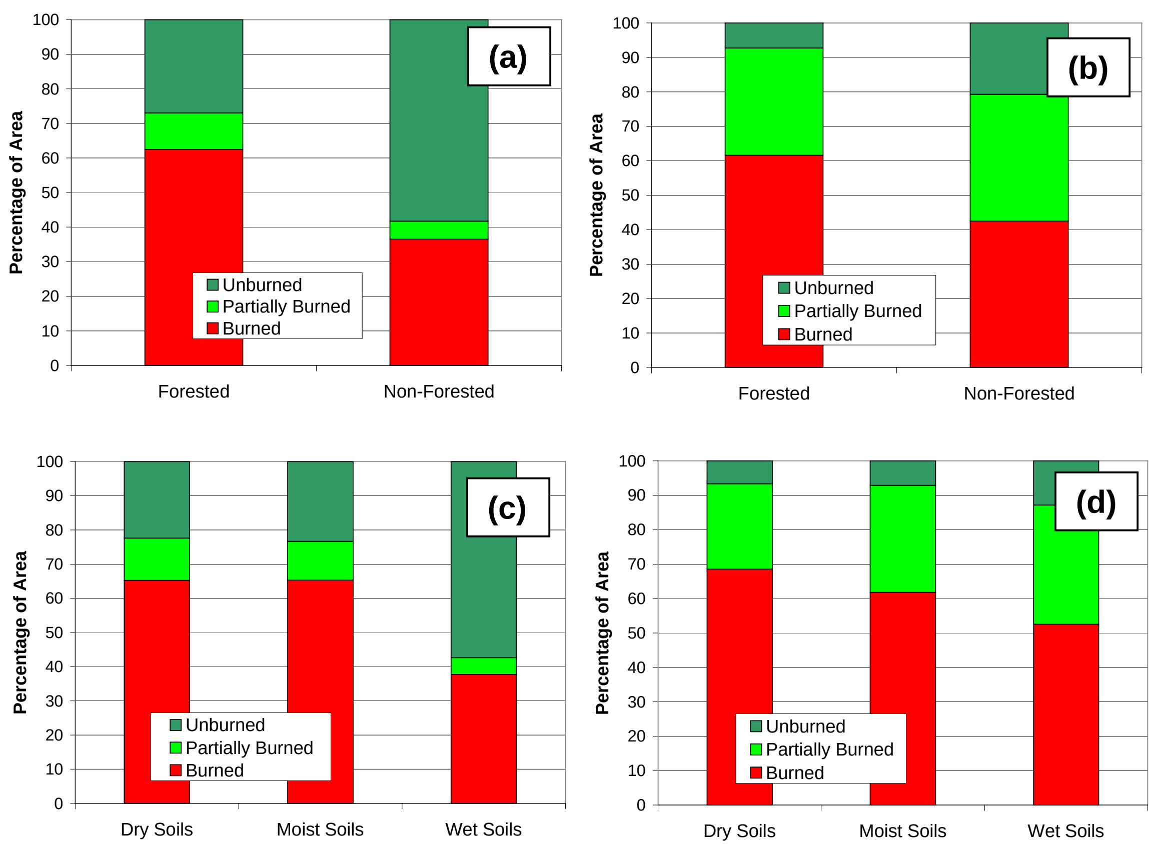
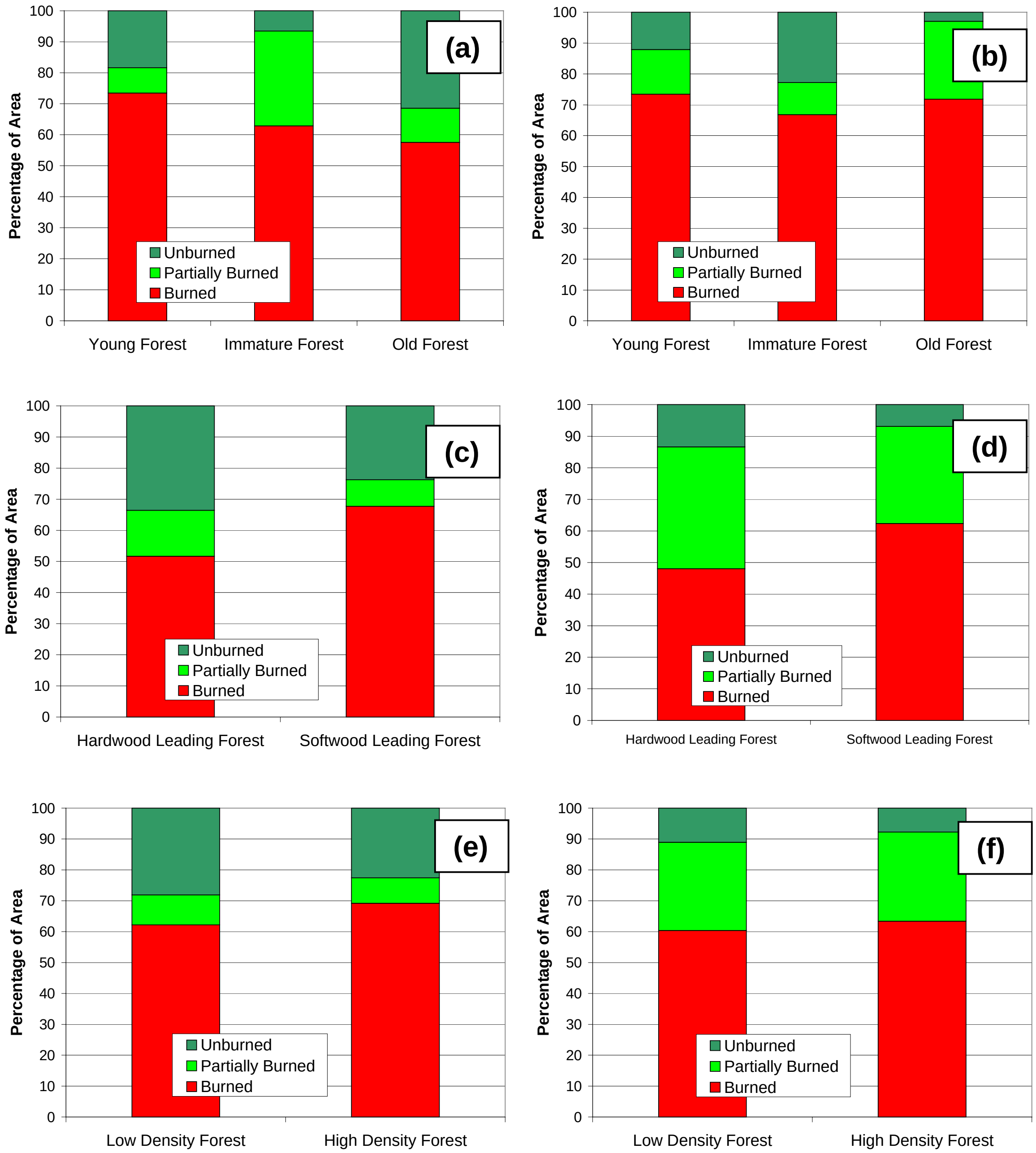


Figure 21. Average probability of burning, partially burning, or surviving wildfire by seral-stage for the F-M (a) and B-S (b) regions, leading species for the F-M (c) and B-S (d), and stand density for the F-M (e) and B-S (f) regions.



In general, the two major land classifications created greater (and more significant) burn probability differentials. The probability of non-forest burning in the F-M region was 37%, compared to 63% for forest (Figure 20a). For the B-S region, the same probabilities were 42% and 62% respectively (Figure 20b).

It is also interesting to note that probability of a residual surviving intact (*i.e.*, with no mortality) was more than double for non-forested areas than it was for forested areas, for both landscapes (Figure 20a, b). The relationship to soil moisture regime was also quite strong. Wet soils in the F-M region had a 38% chance of burning, compared to 65% for both dry and moist soils (Figure 20c). The response to soil moisture was more gradual in the B-S region. The probability of areas of wet soils burning within a given event was 52%, compared to 61% for areas of moist soils, and 68% for areas of dry soils (Figure 20d).

Differences in the probability of burning completely (*i.e.*, a class 5 polygon) within an event were less pronounced for different forest types and ages. Burn probabilities for different forest age-classes in the B-S region were not significantly different between young, immature, and old forest (Figure 21b). For the F-M region, old forest actually had the lowest probability of burning completely at 58%, and young forest the highest (74%) (Figure 21a). However, F-M old forest also had the highest probability of not burning at 32%. Immature forest in the F-M region had the highest probability of being partially burned (31%, versus 8% and 11% respectively for young and old forest). Hardwood leading areas had lower burn probabilities than softwood leading areas for both regions (48% versus 62% for the B-S, and 52% versus 68% for the F-M) (Figure 21c and d). However, when partially burned areas were considered, the differences were less significant. In the B-S region, hardwood-leading areas only had a 13% chance of surviving intact within a wildfire event, versus 7% for softwood leading areas. Similarly, hardwood-leading areas had a 33% chance of burning, compared to 24% for softwood-leading areas in the F-M. Forest density class was unrelated to the probability of burning in all cases (Figure 21e and f).

5.4.12 Fire Return Intervals

Question: ***Based on the results from this study, is it possible to define site-specific fire return intervals?***

A *fire cycle* is the long-term (*i.e.*, hundreds of years) average number of years required to burn the number of hectares across a given landscape (Johnson 1992). Thus, over the term of a single fire cycle, some areas of that particular landscape burn more than once, and others not at all.

Fire cycle is a landscape-scale metric in that the details of what happens within individual wildfires are either ignored, or averaged. Finer-scaled burning dynamics are captured by the mean fire return intervals (FRI), which capture the relative rate of disturbance of various vegetation types (Tande 1979). FRIs are normally estimated based on empirical field sampling (Mathieu et al. 2008) such as detailed tree-ring analyses (Amoroso et al. 2011).

I created Estimated Fire Return Intervals (EFRI) for the various fuel-types tested here based on an assumed relationship between the fire cycle as a coarse-scale estimate of the general frequency of fire over hundreds of years, versus the more local return interval of fire locally. Thus, it should be theoretically possible to apply knowledge of burn probabilities within wildfires to generate FRI estimates based on known (or various estimated) fire cycles.

To do so, I assumed three different average fire cycles for the F-M and B-S landscapes; 60, 80, and 100 years. Given our current understanding of this large study area this range of fire cycles is reasonable – *note that I am not suggesting that one or another fire cycle is more appropriate for either region.* For each fire cycle, I multiplied the probability of burning for each of the six (mostly) fixed landscape parameters from Section 5.4.12 for both the F-M and B-S regions by the average fire cycle (Table 8). Note that I considered areas that were partially burned (*e.g.*, polygon codes 1-4) as being *disturbed* for this calculation.

Table 8. Estimated fire return intervals (EFRI) for the F-M and B-S regions based on six major vegetation-types and three fire cycle assumptions.

	Estimated Fire Return Intervals (EFRI)					
Region	**Dry Soils**	**Wet Soils**	**Non-Forested**	**Forested**	**Hardwood Leading**	**Softwood Leading**
			Assuming a 60-Year Fire Cycle			
F-M	77	141	144	82	90	79
B-S	64	69	76	65	69	64
			Assuming a 80-Year Fire Cycle			
F-M	103	188	192	110	120	105
B-S	86	92	101	86	92	86
			Assuming a 100-Year Fire Cycle			
F-M	129	235	240	137	150	131
B-S	107	115	126	108	115	107

Note that all of the EFRI estimates were higher than the fire cycles. This is not surprising. Given the assumptions and method of calculation, it would be impossible for any of the EFRIs here to be lower than the average fire cycle since EFRIs include residual areas. Having said that, it is surprising to see some EFRIs so close to the fire cycle. In particular, the ERFI of areas of dry soil moisture were consistently the lowest of all vegetation types tested here (Table 8). Assuming an 80-year landscape fire cycle, the EFRI of areas of dry soil moisture was 86 for the B-S, and 103 for the F-M. In sharp contrast, areas with wet soils moisture had an FRI of 188 and 92 years respectively for the F-M and B-S regions (Table 8).

The difference in EFRI between the F-M and B-S regions is worth noting. Assuming the same fire cycle, the FRI of non-forested areas for the F-M were double that of the B-S region (Table 8). A similar pattern was noted across all six vegetation types tested.

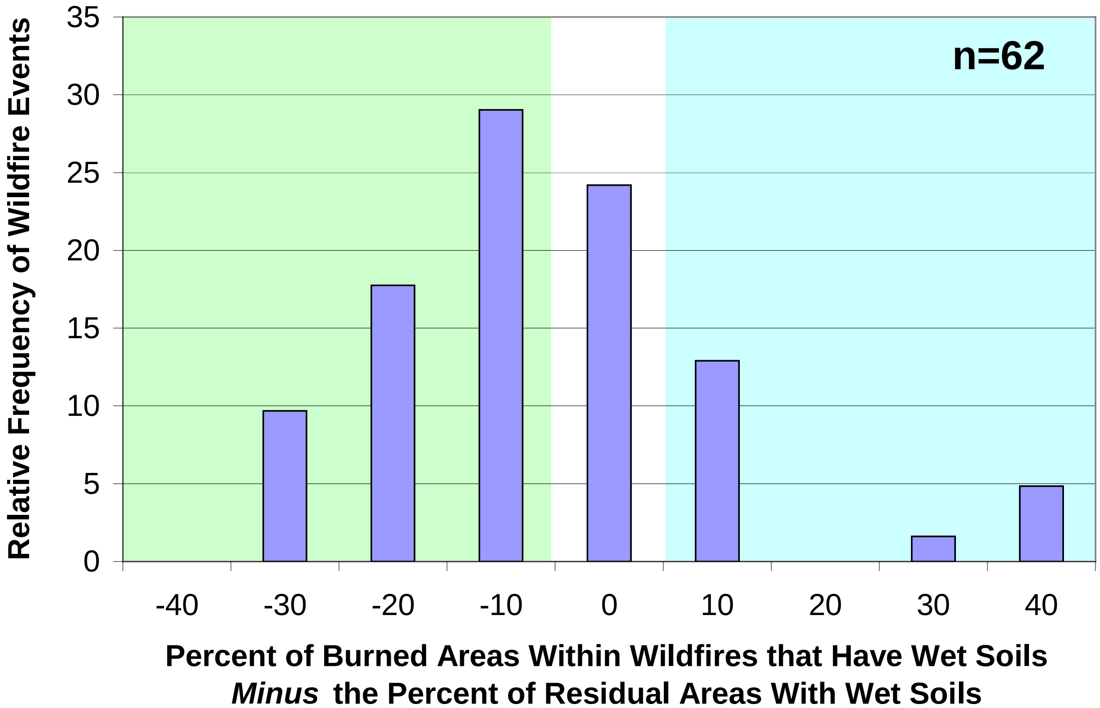
5.4.13 Between Fire Variation

Question: ***How much variation is there from one fire to another in terms of burning patterns?***

The summary graphs in sections 5.4.1 through 5.4.8 reveal the tremendous natural range of variation for these metrics. However, the previous two sections give the impression that all wildfires behave similarly as regards to burn probabilities for different fuel types. To test this hypothesis, and as a supplement to the previous section, I calculated for each event the Burn Probability Difference (BPD) for wet soils, which is the percent of the burned area (i.e., polygon code 5) defined as having wet soil moisture minus the percent of all remnant areas (polygon codes 0-4) defined as having a wet soil moisture regime (Figure 22). A negative difference identifies events in which wet soils were less likely to burn than other soil types, and a positive difference defines those events in which wet soils are *more* likely to burn. Only those events >10 ha in size, with at least 5% of the event area in wet soils were included. I repeated the calculation for merchantable forest (based on an 80-year threshold assumption for merchantable).

The results reveal that for 24% of the wildfires, areas of wet soils were just as likely to burn as areas of moist or dry soils – represented by the “0” bar in Figure 22). Wet soils in 57% of the wildfires were more likely to burn to varying degrees, and wet soils were more likely to burn in 20% of the wildfires (Figure 22).

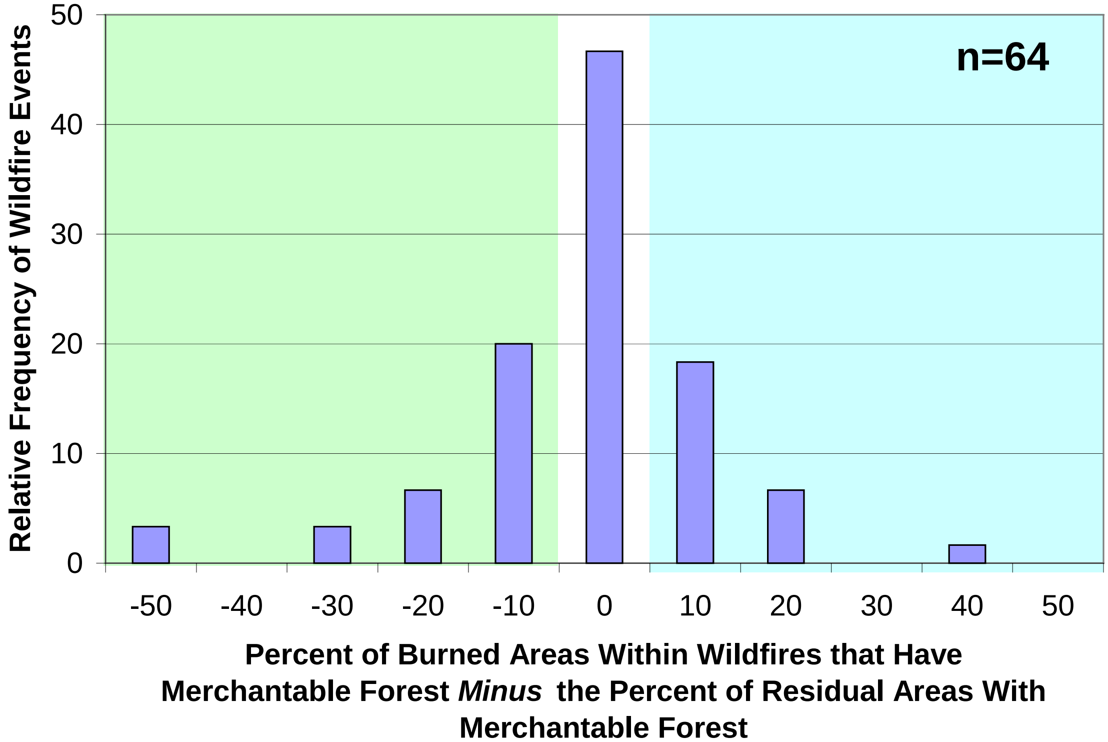
Figure 22. Relative frequency of the percent of burned areas within wildfires that have wet soil moisture, minus the percent of residual areas within wet soils. The green zone represents fires in which wet soils are *less* likely to burn to varying degrees, and the blue zone captures those fires in which wet soils are *more* likely to burn (relative to areas with moist and dry regimes).



The general trend noted in Figure 22 aligns with the results from Figure 20c and 20d in the previous section; areas with wet soils tended to burn (significantly) less than other soil types. However, Figure 23 suggests that this relationship varied between fires. In fact, only 57% of the wildfires in the sample followed this general rule of thumb. Another 20% of the samples showed the opposite trend, and sometimes strongly so.

The BDP results for merchantable forest reveal a more balanced distribution. This is not surprising given that “merchantable” is in this case a simplified version of forest age which was not a significant factor in burn probability from the previous section. The new information revealed in Figure 23 is the relative proportion of the fires that follow this general trend. In this case, almost half of all fire events showed no significant difference in the probability of burning for merchantable forest. In another 33% of the sample events merchantable forest is less likely to burn than other fuel types, versus 27% for those in which merchantable forest is more likely to burn (Figure 23).

Figure 23. Relative frequency of the percent of burned areas within wildfires that are merchantable forest (i.e., >60 years of age) minus the percent of residual areas within merchantable forest. The green zone represents fires in which merchantable forest is *less* likely to burn to varying degrees, and the blue zone captures those fires in which merchantable forest is *more* likely to burn (relative to non-merchantable forest).



6.0 Discussion

6.1 Spatial Language Implications

The spatial language developed for this study is unique in that it objectively defines a simple repeatable system for delineating a wildfire event boundary. No other known such spatial language exists, although more general pattern guidelines are common (*e.g.*, FSC 2004). Given that recent evidence suggests that subtle differences in the delineation of a wildfire boundary can generate significantly different pattern outcomes (Andison 2012), clearly defining a robust spatial language is a priority. The ultimate value of these results both as a scientific contribution, and as a decision-support tool for management, relies on the degree to which the results can be understood, effectively communicated and integrated.

Conceptually, the chosen language captures the *general area of influence of a wildfire* (Burton et al. 2008), meaning that all areas within the boundary have approximately an equal chance of burning. The boundary is akin to that which might be drawn around a wildfire from 20,000 feet in the air. In many ways, *event* boundaries are not unlike those that were drawn on fire history maps (Andison 2012).

Technically, the spatial language introduced here is a three level *event-disturbed patch-residual* conceptual model of a wildfire, as opposed to the more conventional two level *fire-residual* model. A three-level model is more logically consistent with what we know about forest fire behaviour. The presence of multiple disturbed patches in very close proximity could only originate from two possible sources; a) simultaneous lighting strikes, or b) 'spot-fires' caused by wind-borne embers carried beyond the existing fire boundaries (Diaz-Delgado and Pons 2001). The fact that almost all of the fires have one very large disturbed patch suggests that one disturbed patch has been burning much longer than its neighbours, supporting the spot-fire theory. Furthermore, wildfire spotting distances of <300 m have been commonly observed (Kill et al. 1977). That distance corresponds to a buffer distance of 150 m, which captures over 95% of the wildfires in this study. Spotting distances beyond one kilometre have been recorded, but only in association with extreme fire behaviour (Albini et al. 2012). Note that most of the wildfires for which inter-patch distances exceed 300 m are in the Foothills region. The Foothills area is renowned for strong, unidirectional Chinook winds (Burrows 1903), which provides one possible explanation of the more extreme distances between disturbed patches observed in this study.

Originating processes aside, in the end, it only matters that multiple disturbed patches are common enough in these data to be considered as a fundamental disturbance pattern. This particular wildfire pattern phenomenon has until now been undocumented.

A by-product of this particular language is a new class of residual that I have called “matrix remnants”. Until now, the terms *residual* and *remnant* refer to a single residual class, often synonymous with *islands* or *remnants* regardless of location, type, or survival (Eberhart and Woodard 1987, Delong and Tanner 1996, Burton et al. 2008, Soverel et al. 2010). The technical distinction between the two types of residuals in this study is that island remnants are those that can be obviously and unambiguously mapped, and matrix remnants are generated from the event-defining algorithm.

Whether the island-matrix distinction has biological relevance is unknown. On one hand, the physical differences between the two range from subtle (*i.e.*, a few meters) to dramatic (*i.e.*, large corridors of undisturbed forest). On the other hand, island remnants are partially burned and/or are physically isolated, compared to unburned, physically connected matrix remnants. The distinction between corridors and stepping-stones is important for some species (Monkkonen and Mutanen 2003), while overall residual (spatial) complexity is more important for other species (Schmiegelow et al. 1997).

6.2 Regional Fire Patterns

Although many significant differences in burning patterns between ecological regions were found, they were not absolute. One of the most notable, and surprising, similarities was with overall remnant levels. The average level of remnant area per event varied widely from one fire to the next, but on average it never deviated significantly from the Rocky Mountains to the Saskatchewan-Manitoba border, and from the 50th parallel to the NWT border. Perhaps even more surprising is that average remnant levels in this study were 41% by area.

This high level of survival found here aligns with neither current dogma, nor previous local research findings. The disturbance regime of the boreal forest is generally thought to be *stand-replacing* (Johnson 1992), which refers to a very high degree of internal morality. The only available published mortality threshold for stand-replacing events suggests that the lower limit is 80% (Brown 2000). According to this criterion, less than 10% of the fires in this study would qualify, which suggests that one of our more fundamental assumptions of boreal wildfires is in error. Boreal-wide studies of natural wildfire patterns agree with the findings from this study in terms of overall remnant levels (Burton et al. 2008, Madoui et al 2010, Soveral et al 2010). A recent pilot study in the foothills area found evidence of stand-maintaining fires (Amoroso et al. 2011), which is consistent with these findings. However, more local comparisons from western boreal Canada report wildfire residual levels below 10% (Delong and Tanner 1996, Eberhart and Woodard 1987).

There are several possible explanations for these differences, but it is interesting to note that the two western boreal studies relied on inventory maps and/or recent wildfires to generate their maps, while the three latter studies used a combination of existing fire maps and satellite imagery to generate their raw data. The data collected in this study are far more likely to align with that collected for the latter studies.

That more recent wildfires do not seem to follow this same pattern of moderate survival is unknown, although it is an excellent question. Although no one has studied the burning patterns of controlled wildfires, one might hypothesize that fire control activities would be more effective on those areas of a wildfire burning at lower severities, thus ultimately creating fire patterns with less residual area. It is also possible that climate change is already influencing fire severity (Flannigan et al. 2000).

The fact that wildfires in the foothills-mountains tend to be more patchy, but with less partially burned areas relative to those fires in the boreal-shield is likely a function of some combination of weather, topography, and fuel-type. Local winds in the foothills area such as Chinooks (Burrows 1903) are likely to cause more spotting than in the boreal-shield. Fire behaviour also becomes increasingly influenced by the greater topographic complexity in the foothills-mountain areas. And finally, the F-M landscape tends to be dominated by pure conifer stands, which are associated with high burn probabilities. The vegetation mosaic in the B-S landscape is a combination of fuel-types associated with high (*i.e*., black spruce and pine dominated areas), moderate (*i.e*., mixedwood and deciduous dominated areas), and low (*i.e*., wetlands) burn probabilities.

One of the more significant, and intriguing differences in burning patterns between the foothills-mountain area and the boreal shield is the proportion of partially disturbed residuals. Partially burned remnants in the boreal-shield account for double that of fires in the foothills-mountains (29-31% vs 15-16% respectively). This suggests that significant areas of boreal-shield landscape experience lower intensity burning similar to that associated with surface fires (Michaletz and Johnson 2008). Partially killed vegetation creates the opportunity for multiple age cohorts, and high levels of structural complexity (Amoroso et al. 2011). This is particularly intriguing because the complexity of the boreal-shield landscape is usually attributed to the interaction of vegetative species characteristics and micro-site (Pare and Bergeron 1996, Kabzems and Garcia 2004). These findings suggest that the disturbance regime may be equally responsible for the observed complexity. The results also provide an alternative explanation for the seeming paradoxical co-existence of high fire frequencies and the presence of significant areas of multiple age cohorts (Cumming et al. 2000) potentially confounding estimates of historical fire cycles

The potential exists for future fire regime divisions based on distinctive burning patterns. In particular, the differences noted in the Taiga Shield wildfire patterns suggest that further sampling (to the north into NWT) may reveal another distinct fire regime based on burning patterns. Also, although technically there were several fires in the Rocky Mountain region, all of these fires occurred in the transition zone to the Foothills region. We did no sampling further west into the mountain parks. Given the sophisticated fire management policies within most of the mountain Parks, there is an ideal opportunity to augment these data with some natural wildfires. The likelihood of finding another distinctive fire regime is quite high considering the significant corresponding changes in topography, vegetation, and weather conditions.

6.3 Spatial Complexity

These results suggest that boreal landscapes are far more complex than we imagine, and at multiple scales. Almost half of the fires in this study had multiple disturbed patches. Most were relatively small, and within 2-300 m of a single very large disturbed patch, although in rare cases they were more than 1km distant. This phenomenon was particularly prevalent in the F-M region.

Spatial complexity is created within disturbed patches by a combination of convoluted fire edges, and islands of complex shapes and various sizes. The negative space analyses suggested that live vegetation is fairly well dispersed spatially within wildfires, which potentially serves as a robust refugia network for critical non-motile species such as mycorrhizae (Read et al. 2004) and lichen (Boudreault et al. 2000). Furthermore, the fact that the vast majority of the area within wildfires was within 50-100 m of live vegetation has significant implications for old forest bridging habitat (DeLong and Kessler 2000) and re-colonization sources for small mammals (Banks et al. 2010). The 50 m buffer threshold is of particular interest to boreal mxedwood dynamics given that it represents an upper limit for seed dispersal for white spruce (Solarik et al 2010).

Fine-scale spatial complexity is generated by partially burned remnants is of particular interest. The structural and compositional complexity created by this phenomenon over the first few years may be critical for many specialized species and functions (Pinzon et al. 2011). Furthermore, those areas that survive beyond 10 years post-fire (Angers et al. 2011) have the potential to become multi-aged stands.

When these various sources of complexity are considered cumulatively over several decades and across millions of hectares, it is not difficult to imagine the boreal landscape as being highly diverse in terms of composition and structure. One of the ways that this diversity likely translates into function is as regards old growth. For example, areas with significant surviving tree and vegetation cohorts are more likely to create the necessary structural and compositional diversity to qualify as “old growth”, over less time relative to areas subjected to high severity disturbances. Furthermore, given both the area and density of surviving remnants, some proportion of those surviving patches will already be *old* from a seral perspective. Such areas are likely to be very small, but ubiquitous across a given landscape. By focusing on only very large older forest patches as regards “old forest” strategies, we may inadvertently be ignoring a far more critical *old forest* element of boreal landscapes.

Old forest issues and definitions aside, these results suggest that fine to intermediate scales of diversity is both common and important to boreal ecosystems. In other words, the tremendous range and level of variation noted in these wildfires strongly suggests that we have identified a critical part of fine-scale biodiversity for boreal landscapes.

6.4 Management Implications

The findings of this study have several implications for management agencies that are considering, or have committed to, adopting natural patterns as management guides on some level. Although natural pattern strategies are typically associated with forest management activities (Work et al. 2003), the implications of these results apply equally to forest management, parks, fire management, water management, oil and gas exploration and development, regulatory agencies, as well as more strategic teams interested in the cumulative effects of disturbance footprints.

6.4.1 Integrating New Patterns

Most forest landscape natural resource management agencies are still in the preliminary stages of integrating natural patterns into management activities. Within disturbance events, the metrics of interest are usually limited to the total area and size of island remnants. In strong contrast, this research generated a number of new pattern metrics.

There are both challenges, and opportunities associated with integrating new natural pattern metrics. For example, *disturbance events* are currently associated with a single disturbed patch, and what I am calling *matrix remnants* is not commonly recognized as undisturbed residual area (although there are notable exceptions, see OMNR 2001 and FSC 2003). Thus the areas between harvest blocks (for example) are not recognized or counted as disturbance remnants. However, since harvesting tends to occur in clusters, or *compartments*, there is an opportunity to expand the management definition of an event to become the equivalent of a compartment, and harvest blocks as disturbed patches.

Creating partially disturbed areas is another management challenge. Traditional forest harvest machinery has a limited capacity to create partially disturbed areas through feathering, and harvest corridors are not entirely satisfactory as a natural equivalent, and expensive to install. The opportunity in this case might be to begin integrating low to moderate severity prescribed fire as a post-harvesting treatment. This of course creates another, significant challenge relating to local values at risk, and ultimately, agency liability.

Perhaps the greatest concern over the long term is that we (humans) have a tendency to add new regulatory metrics and measures to the point where the big picture becomes lost in the details. One can only imagine the size of a "guideline" necessary to translate the research results from this study alone. On the other hand, this is incentive to begin to consider alternative forms of a regulatory system. In particular, this sort of information is well suited to a results-based system whereby the only requirement is to create within-event diversity at multiple scales (measured by X, Y, and Z) by whatever means available. In the end, the management concern we are trying to resolve is that most cultural disturbance activities are artificially simplifying boreal landscapes. At this stage, it is more important to find a way of instilling the concept of complexity than the specifics of exactly how it is created, or in what form (Potvin and Bertrand 2004).

6.4.2 Variation is Relevant

High levels of natural variability are a recurring theme in this study. At best, a moderate amount of this variability is explained by various biotic and abiotic factors. While the analysis was not exhaustive, it was comprehensive enough to suggest that a substantial amount of the variation was due to local fire weather conditions. This is important for two reasons. First, it represents an intrinsic type of variation in that it precise burning patterns are impossible to predict based on any combination of fixed local attributes. Thus, regardless of how many independent variables involved, *variation* in and of itself is an important natural pattern that we need to recognize and embrace.

The second reason why it is important to know that fire weather is significantly related to variability is that it we can better understand, and potentially predict how future changes in climate will be reflected in within-fire patterns. Fire severity is thought to be one of the more sensitive and immediate fire regime parameters to be affected by climate change (Flannigan et al. 2000). Most climate change models predict an increase in fire severity, which translates into lower survival and thus less fine-scale complexity. Given the compelling relationship between ecosystem heterogeneity and resilience (Drever et al. 2006) this could establish a negative feedback loop by which the next generation of natural disturbances simplify the landscape, which makes them more susceptible to natural disturbances.

The results from this study also raise some new questions as regards climate change. For example, *Will it result in a fundamental shift in fuel-type preferences?* This question may be of particular concern because one possible interpretation of the historical variability in burning preferences for areas of wet soils noted in Figure 22 is that while historical burning conditions usually favoured upland forest fuel-types, there are occasions when it was the reverse. Such occasions might be associated with early spring or late fall timelines, or prolonged droughts, all of which allow the fine fuels to become very dry and more flammable. Climate change predictions include both a lengthening of the fire season, and an increase in the probability of drought conditions (Flannigan et al. 2000). In other words, the susceptibility of wetlands to burning may significantly increase, which will in turn not only create a new suite of fire control problems, but a very different, potentially unfamiliar, landscape pattern.

Towards understanding the climate-fire pattern relationship better, one possible extension of this research would be to collected detailed local fire weather data for each fire, and compare those values to various burning parameters.

6.4.3 Spatial Language

If nothing else, the sheer volume and complexity of the results from this study demonstrate the need for clarity and consistency in a spatial language. For example, even subtle changes to the delineation of an event boundary can have significant implications for several pattern metrics (Andison 2012). The wide range of remnant area results found in boreal wildfire studies noted earlier is largely due to spatial language differences. A clear and concise spatial language will not only benefit the translation to management, but also future research on ecological impacts.

6.4.4 Whole Landscape Thinking

Natural resource frameworks tend to partition management activities based on the resource of value; timber, individual species habitat, water, recreational use, minerals, gas, and so on. This creates silos of responsibilities and management activities on different bits of the ecosystem. Thus, we also tend to study, track, and monitor parts of the landscape ecosystem. This research clearly demonstrates that disturbance is a natural process that affects all parts of the landscape, albeit sometimes differently so. The danger of continuing to manage landscape pieces in isolation is that it will ultimately create unintended, and often negative consequences.

Of specific concern is the vast area of the boreal that is non-forested (most of which is wetland). Although the estimated EFRIs for non-forested areas calculated in this study are relatively high, fire is clearly a process in these areas. While we have studied the potential impacts of fire control on the upland part of the landscape, we understand very little about the role of fire on non-forested areas. In the boreal transition forests of Michigan (where fire control has been in place for much longer than most areas of Canada) fire control has lengthened fire frequencies to the point where it has negatively affected wetland function (Cleland et al. 2004). As they point out, although of great practical value, in terms of biological function, the fact that fire control is particularly effective in wetland areas has been a distinct disadvantage.

Allowing more fire to burn in non-forested areas of the landscape through managed wildfire and prescribed fire programs would seem to be the ideal solution to this issue. However, prescribed fires are quite expensive, and are generally associated with very specific ecological and/or habitat objectives, or to help manage landscape-scale natural disturbance threat. Many jurisdictions in Canada are discussing the possibility of managing wildfires, but the cost in this case is increased risk and liability given the ubiquitous nature of most values-at-risk across much of Canada’s boreal forest. Still, it is encouraging that such debates are taking place.

6.4.5 Links to Fine Filter Values

The original vision of using the natural range of variation (NRV) as natural resource management tool was to provide managers with coarse filter pattern guidance as the foundation for developing plans for disturbance activities

(Franklin 1993). Under the auspices of EBM, attributes of disturbance patterns have become significant biological indicators in and of themselves (FSC 2004). However, the also there exists an assumed, and largely untested relationship between patterns, fine-filter responses, and biological health. The choice of which coarse filter indicators to use and how must thus be balanced with an understanding the associated fine filter processes.

The practical value of natural pattern knowledge is thus twofold; 1) as stand-alone measures of biodiversity and/or ecosystem health, and 2) reality checks for fine filter objectives. For example, the assumption that areas that have experienced any disturbance within the last 40 years is unfavourable habitat for woodland caribou (Environment Canada 2012) has not been tested against this particular version of wildfire patterns.

Ideally, one would like to blend coarse filter futures and fine filter predictions into a single research-management system that a) defines a desired future ecosystem condition, b) makes specific predictions about fine-filter responses, and c) measures and compares those responses to the predictions in order to gain new knowledge (Rempel et al. 2004, Schwilk et al. 2009). One could also support a two pronged approach to NRV research; a) baseline natural pattern research, and b) fine filter research to address the hypotheses generated by the coarse-filter research. This study alone provides a significant number of testable research hypotheses for fine filter ecological research. Regardless of the specific vehicle(s), the two should always be intimately linked (Cury 2004).

7.0 Conclusions

This research highlights the peril of dismissing the value of historical disturbance patterns. Most of the existing policies and practices in the boreal forest of Canada today assume that natural disturbances were “stand replacing”, and boreal landscapes are relatively simple as regards spatial patterns. The occurrence of a small number of severe, expensive, and highly visible fires that were beyond any reasonable level of fire control capacity over the last several years (Sweeney et al. 2012) has only perpetuated this perception. In contrast, this study suggests that historical boreal wildfires were in fact highly complex entities, creating significant levels of structural diversity at multiple scales.

If nothing else, this study dispels the notion of the boreal forest as a structurally simple, even-aged forest-based landscape ecosystem. It also raises many other very good questions. For example, the fine-scale structural and compositional complexities that natural wildfires create translate into high levels of diversity at landscape scales. Given that ecosystem diversity almost always translates into resilience, to what degree are we creating landscapes that less far less likely to respond to perturbations such as MPB outbreaks and climate change? Similarly, given the vast difference between our perceptions of disturbance patterns in the boreal versus the reality as understood through this study, are our interpretations of the future expectations of the delivery of various fine-filter values realistic? Furthermore, what are the risks of continuing to marginalize the so-called non-merchantable bits of the landscape as regards planning? And lastly, and perhaps more importantly, given the existing tenure overlap on any given boreal landscape, what is the capacity of any single resource management agency to address these new, and very good, questions alone?

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