

Managing the pattern of forest harvest: lessons from wildfire

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Managing forests for sustainable use requires that both the biological diversity of the forests and a viable forest industry be maintained. A current approach towards maintaining biological diversity is to pattern forest management practices after those of natural disturbance events. This paradigm hypothesizes that ecological processes will be maintained best where active management approximates natural disturbance events. The forest management model now used in most sub-boreal and boreal forests calls for regularly dispersed clearcuts no greater than 60–100 ha in size. However, the spatial characteristics of the landscape produced by this model are distinctly different from the historic pattern generated by wildfire, which was heretofore the dominant stand-replacing process in these forests. Wildfire creates a more complex landscape spatial pattern with greater range in patch size and more irregular disturbance boundaries. Individual wildfires are often over 500 ha but leave patches of unburned forest within them. The combination of these attributes is not present in recent clearcuts. Allowing a proportion of larger (i.e. > 500 ha) harvest units may provide distinct economic advantages that could outweigh the opportunity costs of leaving some patches of forest behind. For the forest type examined, further evaluation of modelling forest harvest patterns more closely after the patterns created by wildfire is required as it may achieve a good balance and strike a suitable compromise between certain ecological and economic objectives of sustainable development.

Keywords: ecosystem management; fire; landscape ecology; forest management; biodiversity.

Introduction

Land managers and the scientific community in disciplines related to forest management are now expected both to sustain ecological function and biological diversity and to maintain a viable forest industry. Historically, the protection of wilderness areas and areas relatively unaffected by resource extraction in protected areas was assumed adequate to sustain natural ecological functioning and biological diversity. Once protected, these areas were considered inviolate and management, largely restricted to measures such as fire control, was directed at the maintenance of pristine conditions. On the remainder of the land area a single species emphasis approach was usually adopted, with management guidelines typically designed to provide habitat for large game species (e.g. moose). The spatially explicit management stratification afforded by this dichotomy has proven problematic however, as the protected areas tended not to include low elevation ecosystems displaying the greatest net primary productivity, were generally not large enough to protect wide ranging species such as grizzly bears, and were often chosen for their visual majesty, recreation value and lack of conflict with established resource-based

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industry. Given these and other limitations it is unlikely that maintenance of biodiversity will be achieved by strategies confined to a small portion of the land base, but that management strategies must encompass all landscapes and address multiple temporal and spatial scales (Franklin, 1993).

In the wake of public criticism of past practices, and a better understanding of natural systems, new ideas about forest management have emerged (Maser, 1988; Franklin, 1989; Hansen *et al.*, 1991). One idea currently receiving attention is that managed disturbances should be designed to mimic natural disturbance regimes. This suggestion is derived in part from emerging evidence indicating the important and often key role that disturbance has demonstrated in forested ecosystems (Zackrisson, 1977; Van Wagner, 1978; Hessburg *et al.*, 1994). The assumption underlying this idea is that the biota of a forest are adapted to natural disturbances and thus could cope more easily with the ecological changes associated with timber harvest if they resembled those of natural disturbances (Hunter, 1993; Swanson *et al.*, 1993; Bunnell, 1995).

In some forests, natural disturbances maintain plant and animal diversity over time and space by maintaining structural complexity within stands and by influencing the size, distribution, edge characteristics, and dispersion of stands across the landscape (Zackrisson, 1977; Hansen *et al.*, 1991; Hessburg *et al.*, 1994). The size, shape, and location of individual forest patches or stands profoundly affect landscape stability and productivity (Franklin and Forman, 1987; Frank and McNaughton, 1991). Since the impacts of forest management appear now to overshadow those of natural disturbance agents, evaluating the implications on forest patch distribution of conventional and alternative forest management systems is badly needed.

In the past, the main natural disturbance shaping the forests of sub-boreal and boreal plateau landscapes dominated by lodgepole pine (*Pinus contorta*), white spruce (*Picea glauca*) and trembling aspen (*Populus tremuloides*) was stand-replacing wildfire (Rowe and Scotter, 1973; Zackrisson, 1977; Van Wagner, 1978). Previous studies have indicated that the overall area burned by a single stand-replacing wildfire may exceed 10 000 ha (Eberhart and Woodward, 1987) and the fire return interval is 80–125 years (Johnson, 1992; Bunnell, 1995). Post-disturbance natural regeneration of these stands appears to be prompt as indicated by natural stocking success of lodgepole pine after clearcutting within the sub-boreal landscape (Eremko, 1990). These landscapes are in contrast to others with similar tree species composition which can have extended fire return intervals leading to an uneven-aged stand structure (Romme, 1982; Parker, 1986) or have complex regeneration patterns due to low-intensity fire (Agee, 1994) or variability in serotiny of lodgepole pine cones (Lotan, 1976).

Although the temporal and spatial pattern generated by boreal and sub-boreal wildfires may be simpler than those generated in other natural systems, it could be relatively complex when compared to current clearcut harvest disturbances. Burning is rarely if ever completely uniform, leaving relatively small unburned or lightly burned areas containing trees that survive the fire (Eberhart and Woodward, 1987). In contrast, current clearcut harvest disturbances leave little or no mature trees within them, and are relatively evenly distributed throughout old-growth and mature forest.

The intent of this paper is to examine differences and similarities between wildfire¹ and

¹ Other natural stand replacement disturbances such as wind were not separated from wildfire but local knowledge gained from 20 years of forest operations indicate they comprise a very small proportion of the study area.

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harvested areas within a portion of the sub-boreal landscape. The paper explores the hypotheses that:

- (1) wildfires are becoming smaller and less frequent,
- (2) clearcutting has supplanted wildfire as the dominant disturbance agent in terms of area affected,
- (3) individual clearcuts are spatially different from individual wildfires, and,
- (4) the landscape mosaic produced by clearcutting varies from that produced by wildfire.

Because the data analysed was collected at a large-scale the paper focuses discussion on large-scale pattern and makes no attempt to examine finer scaled pattern generated by processes such as uneven regeneration.

Study area

The study was conducted in the plateau portion of the Salmon River variant of the moist cool Sub-boreal Spruce (SBSmk1) biogeoclimatic unit (DeLong *et al.*, 1993), extending from about 54–55°N and 122.5–124°W. The area ranges in elevation from 750–1000 m, has a mean annual precipitation of 727 mm, a mean annual temperature of 1.5°C, and covers 782 213 ha, approximately 660 000 ha of which is forested. The forest stands are generally dominated by lodgepole pine or hybrid white spruce (*Picea glauca* × *engelmannii*), with localized areas dominated by trembling aspen or black spruce (*Picea mariana*), and scattered patches dominated by Douglas-fir (*Pseudotsuga menziesii*) or paper birch (*Betula papyrifera*). The study area was selected, in part, for relative uniformity of topography. This was done to minimize the confounding effect of topography on wildfire behaviour. In addition, it was assumed that ignition patterns were random, that the probability of wildfire occurrence was equal across all age classes, and that the study area was large enough such that only a small portion of the landscape burned in any individual year. These assumptions are similar to those made in other studies conducted in boreal forests (Maruoka and Agee, 1994). Individual stands within the study area, as determined by 20–30 years of operational and research experience and stand cruise data, are normally even-aged. Regeneration after wildfire was assumed to be prompt, therefore the fire age and extent could be derived from the mosaic pattern of different aged stands (Agee, 1994). This assumption is supported by age data collected from stands of known fire origin from the British Columbia Ministry of Forests (BCMOF) fire history database.

Methods

A number of databases were generated for this study using a Geographic Information System (GIS). These databases were created in order to examine different aspects of temporal and spatial pattern of wildfire and harvesting at different scales. Table 1 summarizes some information about these databases and the data sources that were used to generate them.

Landscape study

To examine spatial and temporal characteristics of wildfire and harvesting across the entire study area maps and databases were generated from forest inventory age information by a Geographic Information System (GIS).

Table 1. Summary of sources and attributes generated for the databases

Database	Source	Attributes generated				
		Age class	Area (ha)	Perimeter (m)	Topographic position	Interior forest
Landscape - wildfire	Inventory age class	x	x	x		
Landscape - harvesting	Inventory age class	x	x	x		
Island remnant	Mapped wildfires		x	x		
Island remnant by topographic position	Mapped wildfires		x	x	x	
Habitat comparison	Mapped wildfire and harvesting database					x

For the forest inventory, forest age of a map polygon was interpreted from 1:15 840 black and white aerial photographs by comparing height, crown closure and crown condition with adjacent polygons. Periodic ground-truthing was performed to verify stratification and standards require polygon age to be within 15% of actual age (British Columbia Ministry of Forests, 1992). The estimated age of each polygon was subsequently assigned to an age class (Table 2) and entered into a database along with the digitized boundaries to allow for use in a GIS.

To examine the pattern and scale of wildfire, all cultural disturbances such as harvesting and land clearing were absorbed into the age classes that surrounded them using a combination of automated and manual GIS functions. The GIS subsequently generated a map and database based on these modified age class polygons. Minimum harvesting age for the stands in the study area is 120 years (i.e. age class 7 and greater) and the harvest units tend to be dispersed throughout large even aged patches of mature forest.

Table 2. Age limits for inventory forest age classes used by the Ministry of Forests in British Columbia

Age class	Age limits (years)
1	1-20
2	21-40
3	41-60
4	61-80
5	81-100
6	101-120
7	121-140
8	141-250
9	> 250

Reconstruction of these areas required little subjective interpretation. Land areas cleared in activities not related to forestry (approximately 5% of the study area) were more difficult to reconstruct and required some interpretation where the disturbance cut through adjoining age classes. Forest cover maps based on air photography from 1957-63 were used to assist in this process.

In order to compare the amount of wildfire and harvesting in a 20-year time interval, age class 8 and 9 data were excluded. This was done due to the larger age range for these age classes, as well as the increased probability that any area of this age might be the product of more than one disturbance and thus skew disturbance size and shape relationships. However, age class 8 and 9 stands were colour-themed as one type and joined together for mapping purposes. This afforded perspective as to the size and spatial distribution of mature and old-growth stands within the study area, and was used to determine age class adjacency during the selection of individual wildfires for more intensive study (see individual wildfire methods).

Wildfire literature for forest types similar to the study area (Van Wagner, 1978; Johnson, 1992) suggests that fire pattern is random and thus recent wildfires will re-burn portions of older wildfires thus masking their full extent. In order to correct for this, it was assumed that the amount of a particular age class of wildfires which would be re-burned would be directly proportional to their extent in the landscape. Thus, the total area of disturbance for any one time interval (20 years for this study) was proportionally allocated to each previous 20 year interval and an estimated total area of disturbance for each interval determined. For the purposes of this correction, the forest age intervals were truncated at 200 years of age. This assumption is supported by extensive operational timber cruising data and data (84 plots) collected in this area for development of a fire behaviour model (Anderson, 1994, unpublished data). This correction was applied in order to provide an estimate of the total area burned within the study area in any one time period. Using this 'rollback' correction technique, it is assumed that the magnitude of the error of the estimate will increase with time interval from present.

The harvesting database was produced by selecting map polygons with a harvesting history. Since clearcut harvest patterns were of most interest, older partially cut areas and road clearings not included in the harvest units were removed and a GIS was used to generate a map and database from the remaining polygons.

Both naturally disturbed and harvesting databases contained area (ha) and perimeter (m) data. Using this data shape index for both harvested and wildfire openings was calculated as:

$$SI_x = p_x / 2 \sqrt{\pi \cdot \sqrt{[a_x \cdot 10000 (\text{m}^2 \text{ha}^{-1})]}}$$

where SI = shape index, p = perimeter (m), a = area (ha) (Patton, 1975).

Individual wildfires

Using the natural disturbance map from the landscape study, nine individual age class 3 (41-60 years old) wildfires were selected for further study. These nine wildfires represented a small proportion of the total number of wildfires but covered a relatively broad range of disturbance sizes (159-2239 ha). More recent wildfires were not considered because fire suppression activities and subsequent salvage logging would have significantly altered their character. Older wildfires were not included because of difficulties in distinguishing between unburned patches of older forest and the surrounding forest when

both are similar in height. The nine stands were selected such that they were primarily surrounded by much older forest (i.e. > 100 years difference) in order to decrease the probability that errors would be made in boundary delineation.

Black and white aerial photographs (1:15 840) were examined to determine the topographic position and size of unburned patches of mature timber within the nine age class 3 wildfire initiated disturbances. The boundaries of the wildfire and residual patches of living mature timber within the wildfire; henceforth termed 'island remnants', were outlined on the aerial photographs and then digitized for later use in the GIS. Island remnants of less than 0.2 ha or areas of dispersed residual stems with a crown cover of less than 15% were not outlined. A database containing area and perimeter data of the island remnants was generated by the GIS.

The island remnants were also categorized by their landscape position. Where an island remnant occupied more than one landscape position it was subdivided. Subsequent to subdivision and categorization a second database was generated by the GIS. The landscape positions used were: 'draw', defined as a moisture receiving area where the ground slopes up on at least two sides; 'wetland', defined as those areas with water present at or near the surface throughout the year and with little if any coniferous cover present; 'wetland edge', defined as closed forest (crown cover > 15%) adjacent and contiguous with a wetland; and 'ridge', defined as that portion of the landscape occupying a height of land and from which the ground drops off on at least two sides. These four landscape positions are thought to offer some interpretation as to the reasons why stands may survive wildfire. Some stands, however, were not as easily interpreted. The term 'other' was used if the patch did not fit any of the aforementioned landscape positions.

Paired colour photographs were taken along pre-defined transects within four of the nine mapped wildfires, using a helicopter mounted 70 mm camera boom. The transects were designed such that they intersected as few of the previously identified residual patches as possible. The transects were flown at an altitude of 120 m and 120 photograph pairs, evenly distributed between the four wildfires, were taken. Photograph pairs were then analysed to determine the number of live and dead standing remnant trees (i.e. trees left from the previous stand) within a circular 0.2 ha plot. Broken off remnant trees which were beneath the height of the canopy of the younger forest could not be accounted for using this method.

Habitat comparison

A comparison of landscape habitat attributes was conducted between one of the mapped individual wildfires and an area of recent harvest openings (< 10 years old). Areas of individual harvest openings dispersed in mature forest were summed until the total area equalled an approximate match ($\pm 2\%$) with the disturbance area of the mapped wildfires. The size of the analysis area for the wildfire scenario was then increased to equal the total area required to include the harvest openings of the recent harvesting scenario. For purposes of comparison, this additional area was considered to represent mature forest. Calculations of the amount of 'interior forest' for these two scenarios was conducted using a GIS. 'Interior forest' was defined as the amount of mature forest remaining after the removal of different 'edge zones' (100, 200 and 500 m) and was calculated using GIS distance functions. 'Edge zones' are used to reflect the distance into the mature forest that is subject to the effects (e.g. climatic) of being close to an edge. The width of these edge

zones covered the range suggested in studies of edge effect on microclimate and vegetation dynamics (Oke, 1978; Chen *et al.*, 1992).

Results and discussion

An estimated high of 23.1% of the total forested area within the SBSmk1 was disturbed by stand-replacing wildfire over a given 20-year period (Fig. 1). This value occurred from 1911–30, a time period when a small proportion (1.2%) of the total is accounted for by the 'rollback' correction. The number and size of natural disturbances have decreased substantially in the past 40 years (i.e. since 1950), likely in response to fire suppression activities (Figs 1 and 2). In the five 20-year periods before 1950 the estimated percentage of total forested land disturbed ranged from 7.4–23.1% with an average of 15.7%. Between 1970 and 1990, 12% of the forested land was disturbed by clearcutting, while only 0.15% was disturbed by wildfire. Clearly, harvesting is replacing wildfire as the dominant disturbance process shaping the pattern of forest age on the landscape.

Upon examination there are some clear differences in landscape and individual disturbance patterns between wildfire and harvesting. One key difference affecting landscape pattern is the selectivity of forest age of harvesting. Whereas wildfire is non-selective as to the age of forest it will disturb (Johnson, 1992; Van Wagner, 1978), harvesting only occurs in older stands (i.e. > 120 years). Since the economic rotation age of these forests is unlikely to change significantly, this basic difference will remain. However, there are other characteristics of wildfire landscape pattern which may be emulated by altering forest management practices.

One factor controlling forest age pattern across a landscape is the size of stand-replacing disturbances. In the landscape studied, wildfire disturbances show a higher proportion of area in large (> 500 ha) and small (< 50 ha) forest age patches than harvesting (Fig. 3). In the past 40 years smaller wildfires have remained common while larger ones have become

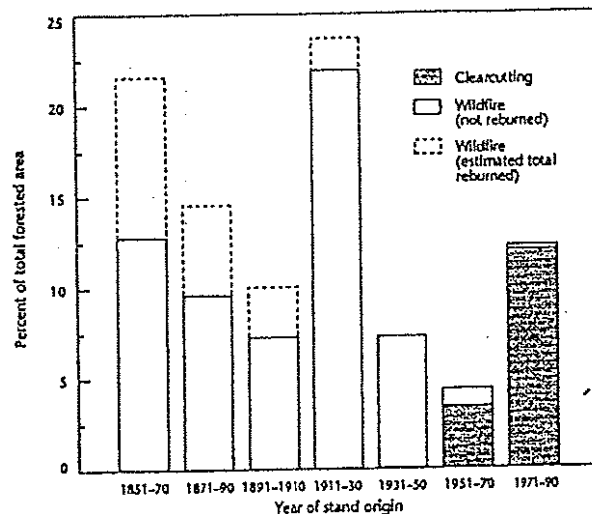


Figure 1. Percentage of total forested area disturbed by wildfire and clearcutting by 20-year period. (Note: for wildfires the area within the dashed line indicates the estimated additional amount returned by subsequent wildfire.)

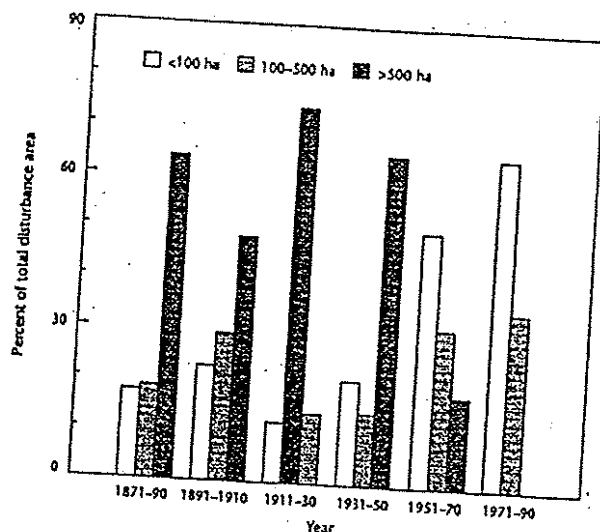


Figure 2. Percentage of total wildfire disturbance area by size class of wildfire within different 20-year periods.

rare (Fig. 2). Thus it is the area previously disturbed by larger wildfires which is now being replaced by disturbances related to harvesting.

In the four 20-year periods before effective fire control, 50-75% of the total naturally disturbed area was comprised of wildfires larger than 500 ha (Fig. 3). By comparison, only 38% of the harvest openings were larger than 500 ha, with a higher proportion being between 50 and 500 ha (Fig. 2). The largest harvest opening is 3238 ha versus 34 273 ha for a wildfire generated opening. Current management guidelines in the study area limit individual clearcuts to 80 ha. Most larger harvest openings result from unanticipated salvage operations required to recover timber damaged by blowdown and spruce beetle (*Dendroctonus rufipennis*). These openings represent exceptions to extant management plans. Since both blowdown (Stathers *et al.*, 1994) and spruce beetle attack (Safranyik *et al.*, 1990) are somewhat predictable it seems appropriate to allow larger harvest units to be part of the plan. Not only would this allow for a more natural range of patch size to be planned, but it would also reduce uncertainties and their concomitant costs. The overall result would be management by intent rather than management by default.

A three pass harvesting system is currently employed in the study area. This system requires that cutblocks be dispersed regularly throughout the mature forest (i.e. > 120 years) such that reserve areas are left between to accommodate two future passes. Before a portion of the reserve area can be cut in the second pass, trees on the first pass blocks must have reached 3 m in height. This is intended to provide hiding cover for ungulates and provide visual screening. This 'green up' period takes approximately 15-20 years within the study area. The final pass occurs when green up is complete on the second pass blocks. This dispersed clearcutting system results in an unnatural pattern on the landscape (Bunnell, 1992) and is leading to the homogenization of disturbance patch size and spatial pattern. Given our poor understanding of ecosystem function, deviations from the natural disturbance pattern could adversely impact biodiversity. Bunnell (1995) found that the structure of vertebrate fauna was adapted to the spatial extent and distribution of seral

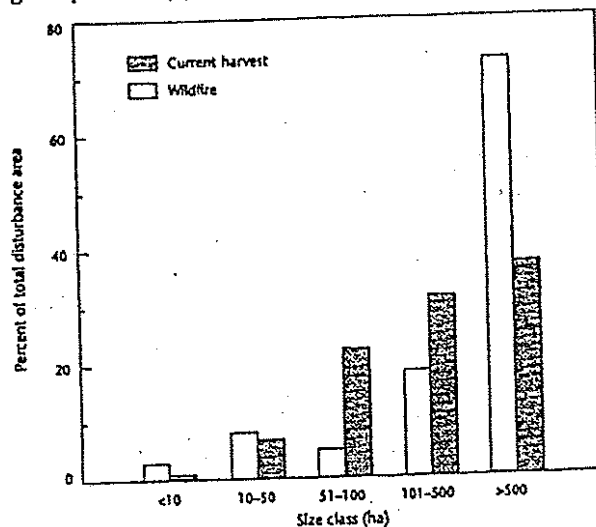


Figure 3. Percentage of total disturbance area in different size classes for wildfire and current harvesting.

stages of forests in British Columbia. The study showed a positive relationship between fire size and proportion of native vertebrate fauna associated with early seral stages.

When compared to the dispersed smaller cutblock model, larger harvest units associated with a more natural harvesting pattern could reduce the active road density over the landscape. This would occur because a significant amount of internal road system of larger units could be deactivated once harvesting and initial silviculture operations are completed. Decreasing active road densities should benefit wide-ranging species such as grizzly bears and wolves, which are negatively impacted by high-density road networks (McLellan, 1990; Thurber *et al.*, 1994).

Historically, restricting maximum cutblock size to less than 80 ha was promoted as a means of ensuring favourable conditions for species requiring edge (e.g. moose) as the ratio of perimeter (i.e. edge) to area increases as patch size decreases. Apart from patch size, the main factor affecting amount of edge is the shape, with irregular shapes resulting in more edge. The shape index (Patton, 1975), an indicator of shape complexity, was consistently higher for wildfires than for harvested openings (Fig. 4). Shape index increased with wildfire size while harvest opening shape index remained constant up to 500 ha and then dropped. The increase in shape index with wildfire size agrees with the findings of Eberhart and Woodward (1987) who applied fire shape index to large fires in Alberta. Increasing the complexity of shape of larger harvest units could yield similar amounts of edge to the smaller, more geometric harvest units currently employed.

Whereas some species require edge, others such as some forest-dwelling migratory songbirds appear to require interior forest habitat that is not affected by edge (Terbough, 1992). Smaller dispersed clearcuts result in a fragmented forest condition which reduces the amount of forest interior (Hansen *et al.*, 1991; Chen *et al.*, 1992). The harvesting and wildfire scenario simulation analysis that compared the wildfire and harvest patterns depicted in Fig. 5 indicated that wildfire disturbance resulted in more interior forest habitat, especially for 200 and 500 m edge zones (Table 3). Allowing for a proportion of

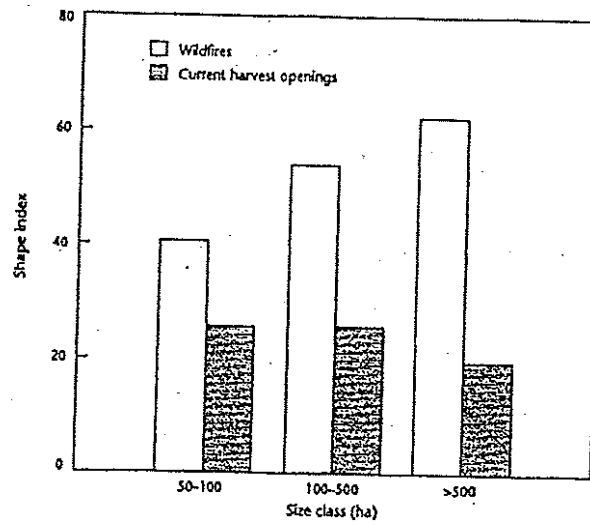


Figure 4. Shape index for wildfires and current harvest openings in different disturbance size classes.

larger harvest units accompanied by large leave areas should be an appropriate management strategy for interior forest species. This management strategy has been recommended for woodland caribou (Racey *et al.*, 1991).

Island remnants within the perimeter of large fires may provide valuable wildlife habitat and seed sources for natural regeneration (Eberhart and Woodward, 1987). Detailed mapping of the nine individual wildfires showed that island remnants comprised 3–15% of the total disturbed area while individual island remnants ranged in size from <1–73 ha. The proportional amount of island remnant area and size of individual remnants increased

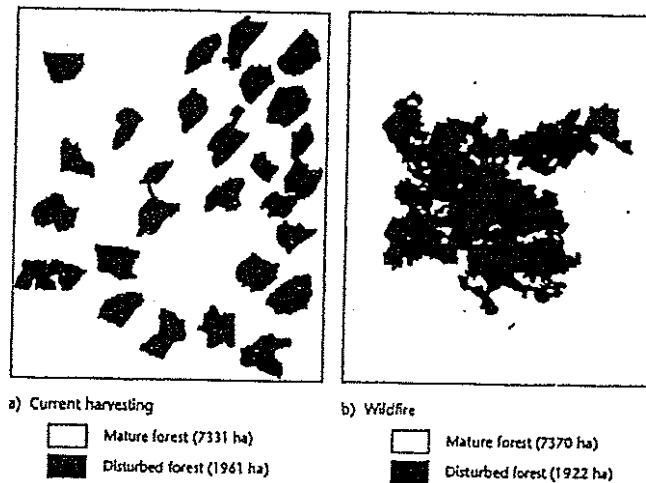


Figure 5. Illustration of (a) current harvest units, and (b) individual wildfire used for habitat comparison study.

Table 3. Amount of interior forest area for wildfire and current harvesting scenarios using different edge zones

Edge zone	Interior forest habitat (ha)	
	Wildfire	Current harvesting
100 m	7131	6613
200 m	6369	5112
500 m	5452	2086

with the size of wildfire (Fig. 6 and Table 4). The increase in relative amount agrees with Eberhart and Woodward's study (1987) which indicated that the amount of island remnant area per 100 ha increased up to fires 2000 ha in size. This increase is likely the result of a combination of the higher probability that larger wildfires will encounter low fuel areas and topographic positions that produce island remnants, and the variable winds associated with prolonged burning times (Foster, 1983; Eberhart and Woodward, 1987). The dominant topographic positions that island remnants were associated with varied considerably among the nine wildfires studied, but the amount of island remnant area associated with draws was consistently high (Fig. 7). This is most likely attributable to a combination of the wetter conditions experienced in draws and their relative frequency of occurrence. Island remnants associated with undefined topographic positions (i.e. 'other' category) were also common and may be due to irregular fire skips on level to gently sloping terrain. These irregular skips may be attributed to the complex wind patterns associated with fire. These data lend support to the explanations for the existence of island remnants outlined in Eberhart and Woodward (1987) which include combinations of

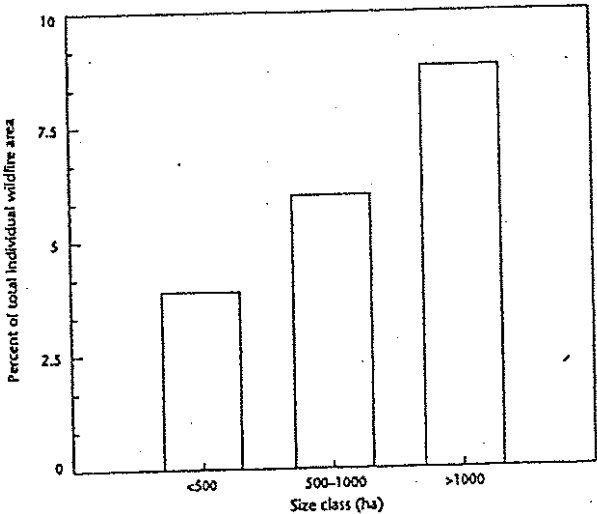


Figure 6. Percentage of total area of wildfire occupied by island remnants by size class of wildfire.

Table 4. Average and range in proportion of area of island remnants by size class of remnant for wildfires < 1000 ha and > 1000 ha

	% of total area in island remnants by size class (ha) of remnant						
Fire size (ha)	<1	1-2	2-5	5-10	10-20	20-50	> 50
<1000	28	21	32	17			
n=4	(17-46)	(12-35)	(27-40)	(0-44)			
>1000	6	11	20	11	17	27	8
n=5	(2-11)	(6-20)	(11-32)	(2-17)	(8-29)	(10-43)	(0-32)

topography and soil moisture, changes in wind during the fire, and factors such as water bodies or rock which disrupt the horizontal continuity of the fuel bed.

The 70 mm transect data indicated that there were only 0.75 live and 0.26 dead dispersed remnant trees per hectare when island remnants were not considered. This lack of dispersed structure remaining from the previous stand demonstrates the relative importance of the island remnants as long-standing sources of large live trees and snags within the younger forest. The usefulness of island remnants for wildlife was demonstrated by Gasaway and DuBois (1985) in a study of the ecological impacts of large fires on moose. They found that 67% of the moose observed within a fire perimeter were located in unburned island remnants which covered only 15% of the total fire area.

The preceding discussion demonstrates some spatial differences between wildfire and harvest-generated openings. The long-term effects of these differences, if any, are as yet unclear. If we accept the premise that the landscape will continue to be subjected to cultural disturbance, then our data suggests it is important to examine if biodiversity in that landscape will be maintained best by activities that more closely approximate natural

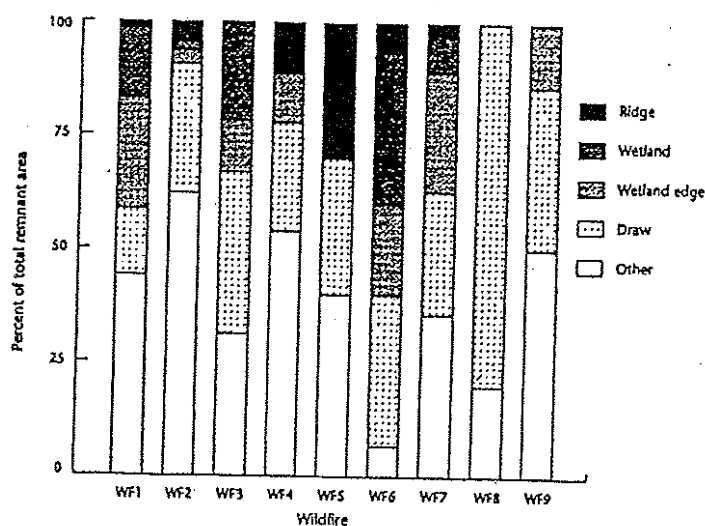


Figure 7. Percentage of total area of island remnants associated with different landscape features.

disturbance. Our results indicate a need for further investigations of the impact of patterning forest harvest after wildfire. It is also possible that certain economic advantages may also result from this shift in management direction. Allowing larger harvest units could result in more efficient use of roads and landing systems, thereby reducing both the costs and potential soil degradation associated with these activities. Other efficiencies could be realized by centralizing a higher proportion of the harvesting and silviculture operations in larger harvest units rather than smaller units dispersed across a large geographic area.

Although current clearcut harvesting practices have been said to mimic wildfire, some distinct differences are evident when comparing certain spatial characteristics. This study indicates that wildfire disturbances result in a greater proportion of the landscape in large and small patches, and that individual wildfire disturbances have more complex shapes and retain patches of mature forest within them. Although the study was limited to a portion of the sub-boreal forest, the applied analysis is likely to be applicable to large portions of sub-boreal and boreal forests where large stand-replacing wildfire was frequent and where lodgepole pine is the dominant tree species. Caution must be used in extrapolating the implications of this data too widely, however, as other lodgepole pine forests differ in both fire return (Romme, 1982; Parker, 1986) and intensity of wildfire (Agee, 1994). It should be noted that this paper could not address other considerations required to proceed towards more ecologically based forest management. These include stand level considerations such as retention of large woody debris and diversity and pattern of tree species regeneration.

The implications of patterning forest harvest on the spatial patterns of wildfire needs to be tested. Because the history of harvesting in sub-boreal and boreal forests is both recent and localized, the potential still remains for fundamental changes in resource management philosophy. This is critical since research has demonstrated that larger harvest units are difficult to enact in thoroughly fragmented forests (Wallin *et al.*, 1994). It is clear that the window of opportunity for landscape level hypothesis testing exists in the present, and is likely to disappear in the future. Given our current state of knowledge, particularly the lack of long-term empirical data to support our present landscape management model of dispersed clearcutting, the implementation of larger harvest units which more closely mimic some of the spatial characteristics of wildfire as described in this paper appears to be ecologically justifiable and is unlikely to involve more potential costs. This is important when one of the main goals of current forest management policy is to achieve a balance between ecological and economic sustainability.

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