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Landscape Ecology (2005) 20: 19–33
DOI 10.1007/s10980-004-0286-z
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Research article

The influence of forest harvesting on landscape spatial patterns and old-growth-forest fragmentation in southeast British Columbia

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Received 27 November 2003; Accepted in revised form 3 May 2004

Key words: Canada, Managed forests, Landscape indices, Landscape metrics, Patches

Abstract

Habitat fragmentation is considered one of the major conservation issues of recent decades. We tested predictions of landscape patterns in a 352,253-ha managed forest area in southeast British Columbia. We did this by focussing on forest fragmentation concerns among old-growth, harvest, and wildfire patches in 44 delineated landscapes using patch indices as measures of landscape pattern. We found no significant association between amount of harvesting and 15 old-growth patch indices. Comparisons among patch types revealed that amounts and spatial patterns of harvest patches diered little from amounts and spatial patterns of old-growth patches in control landscapes. Variability indices revealed similar variability between harvest patches and old-growth patches, and more variability between harvest patches and wildfire patches. Little of the evidence gathered in this study supported predictions of fragmentation of old-growth spatial patterns, or predicted dierences between harvest spatial patterns and more naturally occurring spatial patterns. We suggest these results could be due to the relatively small amounts of harvesting and old-growth forest in these landscapes, and therefore habitat amount may be a more important factor than spatial configuration of patches in these landscapes.

Introduction

Habitat fragmentation is considered one of the most serious conservation issues of recent decades (Haila 2002; Wade et al. 2003) because it is suggested to be one of the most important factors in a current species extinction crisis leading to a loss in biological diversity (Wilcox and Murphy 1985). Fragmentation is thought to be a problem within commercial forestry largely because the theory of island biogeography predicts declines in species richness within habitat fragments as they become smaller and farther apart (MacArthur and Wilson 1967; see D'Eon 2002a for review). On this basis, and for the purposes of this study, we considered forest fragmentation to be the process of dividing a forest into smaller and more isolated fragments (D'Eon 2002a). Briefly, species persistence in small isolated fragments, or patches, is predicted to be lower than in contiguous habitat because of higher vulnerability of small populations within small patches to stochastic demographic or environmental events leading to local extinctions (Pimm et al. 1988), and the ultimate result of a reduction in biodiversity. Predicted consequences of fragmentation have been widely cited (eg., Harris 1984; Saunders et al. 1991; McGargial and McComb 1995; Forman 1997).

Much of what is called forest fragmentation is based on human perception rather than rigorously derived conclusions (D'Eon and Glenn 2000). The

term 'fragmented' is often used to describe landscapes without a clear explanation of what is meant by this term or a quantitative basis for describing a landscape as fragmented. In this case, we echo the concerns of Hager and McCoy (1998) who stated that the acceptance of untested hypotheses has adverse scientific and social consequences by promoting management activities that are based on unsupported assumptions.

Commercial forest harvesting is commonly associated with forest fragmentation (Franklin and Forman 1987). It is obvious that forest harvesting generally removes mature and old forest habitat. Consequently, the number of individuals of a species dependant on this type of habitat will likely go down (either through direct mortality or migration) with large habitat removals within a specific area. However, it is not obvious that changes in spatial patterns of forest elements will be a cause of detrimental ecological effects. In fact, the scientific evidence generated thus far is inconclusive at best (Debinski and Holt 2000). This issue is likely related to confusion between the concept of habitat loss, which involves absolute habitat amount, and that of fragmentation, which deals with spatial arrangement of habitat (Andren 1994; Fahrig 1997). This confusion is no doubt fueled by the inseparability of habitat loss and fragmentation in forest management. Any potential fragmentation effects caused by forest harvesting will be accompanied by an associated habitat loss. When the distinction has been made however, spatial configuration of habitat elements was insignificant compared to habitat amount in deriving ecological effects (McGargial and McComb 1995; Meyer et al. 1998; Trzcinski et al 1999; but see Villard et al. 1999). More work is required to resolve this issue.

In this study we tested predictions of fragmentation of landscape patterns in a managed forested area in southeast British Columbia and compared harvest patch patterns to old-growth and recent wildfire patch patterns. To be clear, our work presented here involves the detection of differences and changes in spatial landscape patterns (i.e., patch geometry), and not the direct response of organisms to spatial pattern. We believe that predicted spatial pattern stemming from the fragmentation paradigm represent important prerequisites for fragmentation effects to occur on organisms living in landscapes (i.e., in order for predicted fragmentation effects on organisms to occur, the associated predicted changes in spatial pattern must occur). As such, we often use the words 'fragmentation' and 'fragmented' in this study to refer to the physical process of dividing forests into smaller and more isolated fragments or patches. As well, it is important to differentiate the situation in our study where we are addressing the fragmentation of a seral stage, from a situation of converting forest into non-forested land (e.g., urban or agriculture). As such, we use the word 'fragmentation' in reference to a process involving the change from one forest seral stage (typically 'old seral') to another (typically 'early seral') without permanent landtype conversion (i.e., amount of forest land remains constant).

We structured our tests based on two general questions. (1) Is forest harvesting fragmenting the existing forest pattern? and, (2) Is forest harvesting imposing a new and fragmented pattern on the landscape?

To investigate effects of forest harvesting on existing forest patterns we focused on old-growth forest patches because of their relatively high importance to conservation and forest management (D'Eon et al. 2002). We tested the hypothesis that forest harvesting fragments existing old-growth by targeting old-growth forests and changing their spatial arrangement. If this was true, we predicted that a significant relationship should be observed between old-growth spatial pattern and varying amounts of forest harvesting in landscapes. Specifically, we focused our tests on the following six predictions based on Forman's (1997) predicted consequences of forest fragmentation: as the proportion of harvested forest in a landscape increases (1) old-growth patch density will increase, (2) old-growth patch size will decrease, (3) the ratio of edge area to patch area of old-growth patches will increase, (4) core area of old-growth patches will decrease, (5) distance between old-growth patches will increase, and (6) dispersion of old-growth patches will increase.

To investigate future forest patterns imposed upon landscapes by harvesting, we focused on spatial patterns of current harvest patches. We did this under the assumption that existing clearcuts regenerate and represent future forest patterns. Specifically, we compared spatial patterns of existing clearcuts to patterns of existing old-growth and recent wildfire patches. Patterns of old-growth forest and wildfire were assumed to represent the inherent natural heterogeneity of these landscapes. We tested the hypothesis that forest harvesting creates a future fragmented forest pattern. We predicted that if this was true, then harvest patch patterns would reflect a more fragmented (from Forman 1997) spatial

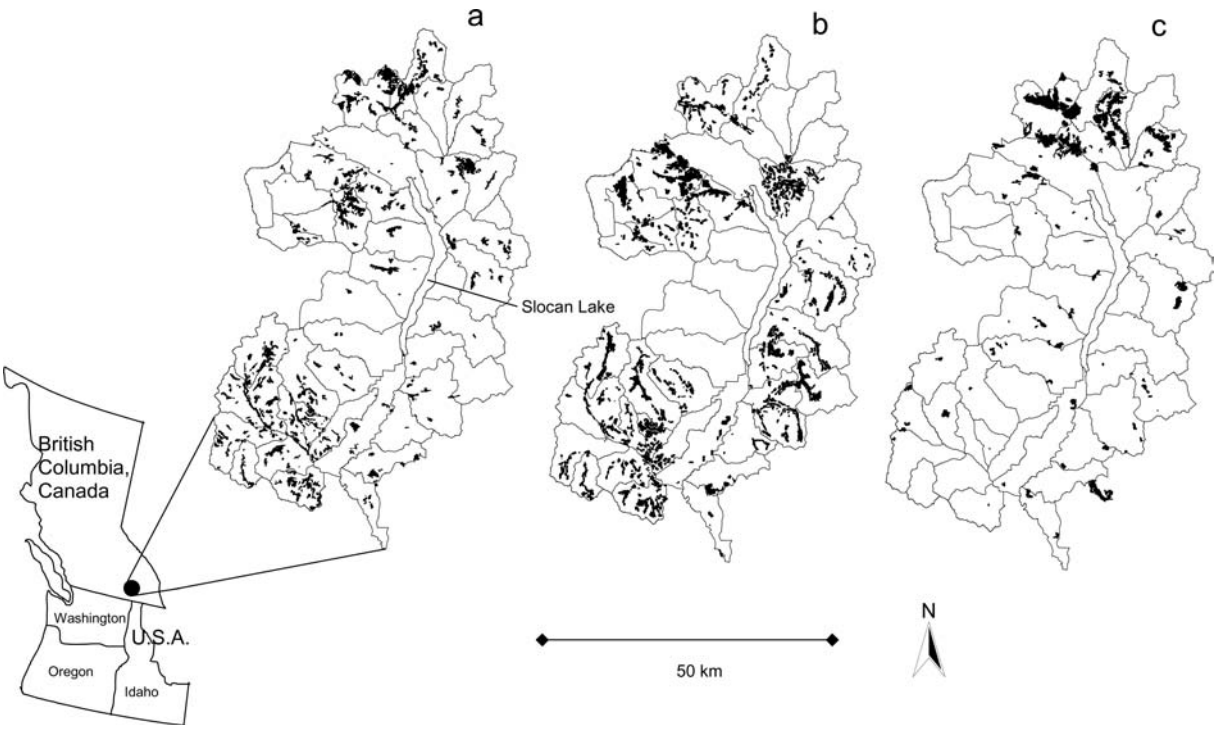

Figure 1. Landscapes and patches used in a landscape pattern study within the Slocan River Valley of southeast British Columbia, Canada (49°N, 117°W). Thin black lines are landscape boundaries for 44 landscapes used in the analysis. Solid black polygons are patches. a) old-growth patches (>140 or 250 year old forest depending on ecosystem classification), b) harvest patches (clear cut ≤ 40 years old), c) wildfire patches (fire ≤ 40 years old). Control landscapes are landscapes without harvest patches in map b. Source data is British Columbia forest service digital forest cover, 1998.

pattern than patterns of existing old-growth and wildfire. Further, we tested the hypothesis that forest harvesting creates a simplified and less heterogeneous spatial pattern of forest elements by creating more uniformly shaped and spaced patches (Krummel et al. 1987; Turner and Rusher 1988). If true, we predicted that clearcut patterns would display less variability than patterns of old-growth and recent wildfire patches.

Study Area

Landscape pattern data were derived from a managed forest area of 352,253 ha within the Slocan Valley of the Selkirk Mountains in southeast British Columbia, Canada (49°N, 117°W; Figure 1; see D'Eon 2002b for more study area description). Terrain within this mountainous area is generally steep and broken with slope gradients often exceeding 80%. Elevation ranges from 525 m along the main Slocan Valley bottom to 2800-m mountain peaks.

Forests made up 75% of the land cover within the entire study area. Approximately 96% of these forests were dominated by coniferous species with forest canopy closure varying from 0% in recent clearcuts and natural openings to 100% in dense stands (British Columbia Ministry of Forests data). Forests of this area are within the *Interior Subalpine and Southern Columbia* regions described by Rowe (1972) and are predominantly within three forest biogeoclimat-icsubzones described by Braumandl and Curran (1992) and natural disturbance regimes described by BC Ministry of Forests and Ministry of Environment, Lands and Parks (1995): Interior Cedar Hemlock Dry Warm (ICHdw) subzone at low elevations, Interior Cedar Hemlock Moist Warm (ICHmw) subzone at mid elevations, and Englemann Spruce Sub-alpine Fir (ESSF) subzone at higher elevations. Briefly, ICHdw forests are ecosystems with frequent stand-initiated fire events (mean interval ~150 yrs); ICHmw forests are ecosystems with infrequent stand-initiating fire events (mean interval ~200 yrs); ESSF forests are ecosystems with rare stand-initiating fire events (mean

interval ~250 yrs). Alpine parkland predominates above 2000-m elevations. Within one representative landscape (Lemon Creek drainage), mid and mature forests (age > 80 yrs) made up 80.1% and early seral vegetation (< 40 yrs) including logged stands made up 16.2%. Logged stands made up 80.5% of early seral vegetation in Lemon Creek drainage.

Logging within the Slocan Valley began in the late 1800s but was primarily confined to localized selective harvesting. Large-scale commercial logging began around 1950. Side drainages of the Slocan Valley have since been managed for forest harvesting and road building to varying degrees. Many areas within the main valley corridor and a large provincial park, however, have been excluded from forest harvesting. A narrow band of low elevation area along the main valley bottom is privately-owned land and has been partially deforested for agricultural and urban development purposes. This private land (~ 5% of the size of the study area) was excluded from the analyses because our focus was on the effects of commercial forest management and forest patterns within the managed industrial forest landbase. Routine forest fire suppression in the area began in the late 1930s (unpublished data, J. Parminter, BC Ministry of Forests, Victoria, BC).

The above natural and human-induced events have created landscapes within the study area (outside of private land) that can be generally characterized as dispersed clearcuts within a predominantly mid to mature (~80 to 140 yrs) forest matrix. We believe the study area to be a good example of managed forests in the interior mountainous regions of western Canada where multiple resource values have created very diverse landscapes experiencing much public pressure and concern. Old-growth management is a major concern in this region with potential fragmentation effects from industrial forest harvesting at the center of the debate.

Methods

Experimental design

The study area was delineated into 44 distinct landscapes (mean area = 8,006 ha; range = 2,735 – 15,479 ha; SE = 442 ha; Figure 1). Landscape boundaries were based on drainage patterns and were generally drawn along heights of land that distinguished two adjacent watersheds. We used the word 'landscape' rather than 'watershed' because landscape is a more universally applicable term without hydrological assumptions. The size and scale of landscapes were chosen based on a natural disturbance regime that usually confines wildfires to a single drainage. As well, individual drainages usually form the basis of forest management planning cells in this area, and were therefore useful land units for investigating both natural and forest management landscape spatial patterns. Following terminology provided by Hurlbert (1984), landscapes were then considered the experimental unit in a mensurative experiment with treatments based on past forest harvest levels ranging from 0 to 34.5% of the forest in a harvested state. Landscapes with no past harvesting were considered replicate control landscapes.

We recognize an inherent violation of an assumption of homogeneity among landscapes within similar treatments, particularly those considered controls, and thus an inherent pseudoreplication problem (Hurlbert 1984). In this case, our assumption that our replicate landscapes are exact or similar replicates and truly independent in time and space, is no doubt false to varying degrees, and therefore represents psuedoreplication defined by Hurlbert (1984). However, we concur with Hargrove and Pickering (1992) and McGarigal and Cushman (2002) that a classical experimental approach is virtually impossible at very broad scales and that careful pseudoreplication leading to induced and qualified conclusions is necessary (e.g., Mladenoff et al. 1993). As pointed out by Hargrove and Pickering (1992), pseudoreplication is less about study design, and more about the misrepresentation or misanalysis of data that results in the presentation of misleading conclusions. To this end, we make no presumptions about having a perfect design, and recognize and present the limitations of our data. However, we made every attempt to define landscapes based on similar ecology and close proximity. All landscapes are within one large drainage basin and many shared common boundaries. Consequently, all landscapes share a common biogeoclimatic zone structure, underlying geology, and climate (Braumandl and Curran 1992).

Landscape index calculations

Three patch types were identified within the study area: harvest patches (clear cuts harvested within the

Table 1. Landscape indices used in pattern analyses for 44 landscapes within the Slocan Valley Basin in southeast British Columbia. In each case, variables were screened and transformed if closer to normal distributions could be obtained based on skewness and kurtosis indicators.

Index name	Description
PROP	Proportion of the forest in a landscape within a patch type
PATDEN	Patch density expressed as number of patches per 100 ha of forest within a landscape
COREDEN	Core density expressed as the amount of core area per 100 ha of forest within a landscape, assuming a 50 m edge effect from the patch perimeter
EDGEDEN	Edge density expressed as the amount of edge area per 100 ha of forest within a landscape, assuming a 50 m edge effect on either side of a patch perimeter
AVGPERM	Average patch perimeter length in meters within a landscape
AVGAREA	Average patch area in hectares within a landscape
AVGCORE	Average patch core area in hectares within a landscape, assuming a 50 m edge effect from a patch perimeter
AVGEDGE	Average patch edge area in hectares within a landscape, assuming a 50 m edge effect on either side of a patch perimeter
MEDAREA	Median patch area in hectares within a landscape
NN	Average distance between patches and the nearest patch within a landscape based on closest straight-line perimeter to perimeter distance in meters.
DISPER	Dispersion pattern index indicating spatial arrangement of patches (Clark and Evans 1954), ranging from 0 to 2.1491. Values close to 0 indicate aggregated spatial pattern, values close to 1 indicate a random distribution, and values approaching 2.1491 indicate even spacing among patches, or maximum dispersion.
EDGERAT	Ratio calculated as the amount of edge area divided by the amount of patch area for a given patch type in a landscape
CORERAT	Ratio calculated as the amount of core area divided by the amount of patch area for a given patch type in a landscape
SHAPE	An area to perimeter ratio from Paton (1975). A value of 1.0 indicates a perfect circle. The index increases from 1.0 with shape complexity and departure from a perfect circle.
FRACTAL	Variation of an area to perimeter ratio using an estimator of the fractal dimension of a line where the fractal dimension = 2logP/logA; where P = patch perimeter, A = patch area (Krummel et al. 1987, Ripple et al. 1991). A value of 1.0 indicates a straight line and increases with shape complexity to a theoretical maximum of 2.0 where a line becomes plane filling.

past 40 years), old-growth patches (forest stands > 140 years for Interior Cedar Hemlock dry warm stands and > 250 years for all others; old-growth definitions consistent with BC Ministry of Forests and BC Ministry of Environment, Lands and Parks [1995]), and wildfire patches (wildfire within past 40 years). All source data were derived from 1998 British Columbia provincial forest cover map information in digital format. The three patch types were chosen for their ecological and management importance, and because they can be reliably distinguished from aerial photographs and have therefore have been relatively accurately delineated in the provincial forest cover database. Harvest, wildfire, and old-growth polygons (patches) were identified using polygon descriptors within the database.

For each landscape and each patch type the following indices were calculated (Table 1): total proportion of the forested land within a patch type (forested land was defined as any polygon with a forest descriptor in the database), patch density (number of patches per 100 ha of forest), core density (amount of interior patch area per 100 ha of forest assuming a 50-m edge effect extending inwards from a patch perimeter), edge density (amount of edge patch area per 100 ha of forest assuming a 50-m edge effect either side of a patch perimeter), average patch perimeter length, average patch area, average patch core area, average patch edge area, median patch area, average nearest neighbor distance (minimum perimeter to perimeter distance between a patch and the closest other patch within a landscape), patch dispersion index, edge ratio (total edge area divided by total patch area), core ratio (total core area divided by total patch area), shape index, fractal dimension. An assumed edge effect of 50 m from a patch perimeter has been used in previous research of this nature (McGargial and McComb 1995) and has been suggested to be a good approximation of edge effects in western North American forests (Kremsater and Bunnell 1999). Patch dispersion (Clark and Evans 1954; also see Ripple et al. 1991) is a measure of non-randomness of patch arrangement and departs from1.0 which represents a random pattern, to <1.0 indicating a trend towards patch aggregation, or >1.0 indicating a trends towards regular or uniform

Table 2. Summary statistics[†] for 15 untransformed landscape indices used in landscape pattern analyses among 44 landscapes within the Slocan Valley Basin in southeast British Columbia.

Index	Patch type					
	Old-growth		Harvest		Wildfire	
(see Table 1)	*n*	Mean (±SE)	*n*	Mean (± SE)	*n*	Mean (± SE)
PROP	44	6.2 (0.90)	44	9.2 (1.19)	44	4.9 (1.85)
PATDEN	44	0.18 (0.079)	44	0.20 (0.027)	44	0.35 (0.006)
COREDEN	44	2.6 (1.08)	44	5.4 (0.79)	44	2.8 (1.17)
EDGEDEN	44	6.9 (0.90)	44	7.5 (0.91)	44	3.2 (1.13)
AVGPERM[§]	41	3204 (218.7	38	3623 (328.4)	33	4654 (663.2)
AVGAREA[§]	41	24.5 (2.43)	38	42.9 (6.37)	33	55.4 (11.13)
AVGCORE[§]	41	11.4 (1.48)	38	26.7 (4.88)	33	35.2 (8.31)
AVGEDGE[§]	41	28.8 (1.96)	38	34.2 (3.08)	33	42.3 (5.95)
MEDAREA[§]	41	10.7 (0.92)	38	17.1 (1.08)	33	33.7 (8.79)
NN[¶]	40	796 (254.1)	37	448 (117.6)	24	3258 (1087.9)
DISPER[¶]	40	0.43 (0.083)	37	0.30 (0.050)	24	1.05 (0.280)
EDGERAT[§]	41	1.3 (0.07)	38	0.9 (0.31)	33	1.7 (0.42)
CORERAT[§]	41	0.41 (0.019)	38	0.56 (0.018)	33	0.46 (0.038)
SHAPE[§]	41	1.9 (0.4)	38	1.6 (0.1)	33	1.9 (0.1)
FRACTAL[§]	41	1.3 (0.01)	38	1.3 (0.01)	33	1.3 (0.01)

[†]Sample size based on the number of landscapes (total $n = 44$) used to calculate the mean; [§]Only landscapes containing ≥1 patches used in calculation; [¶]Only landscapes containing ≥2 patches used in calculation.

spacing. Mean shape index (Paton 1975) and fractal dimension (Ripple et al. 1991) are variations of an area to perimeter ratio where 1.0 represents a perfect shape (perfect circle or straight line) and larger numbers represent increasing departure from perfect shapes and increased shape complexity. Coefficient of variation (CV) was calculated on applicable data and used as a relative measure of variability from the mean within a landscape.

This suite of landscape indices was chosen to specifically deal with a vector database (standard format for provincial forest cover database). They were chosen based on their direct association with testable predictions, as well as their familiarity within the literature for ease of comparison with other work. While some overlap and redundancy may occur among indices, it is widely recognized that there is currently no single and universally applicable measure of landscape spatial pattern (and therefore no single measure of fragmentation) that provides all relevant information, and that most phenomena are best described with a suite of indices (Hulshoff 1995; Riitters et al. 1995; Garrabou et al. 1998).

Patch indices were calculated within an ARCINFO geographic information system platform using uniquely programmed algorithms. Patches were defined using pre-existing stand boundaries in vector format in the forest cover database. The minimum mapping unit in the data was 2 ha (i.e., all polygons ≥2 ha). Since the predominant harvest method in this area, up to and including the last year of data in this study, was large-patch clearcut harvesting (mean harvest patch size = 42.9 ha; Table 2) there is little to no chance that harvest blocks smaller than 2 ha occur and are not considered in our study.

Data analysis

All data analyses were performed using SYSTAT 8.0 (SPSS 1998) statistical software. Tests were considered significant at $\alpha = 0.05$. Non-normal data distributions were assessed using skewness and kurtosis indicators and transformed using logarithm, square-root, and arcsine transformations to produce more normal distributions (transformation type chosen based on best results; Fowler et al. 1998).

To test the fragmentation predictions from Forman (1997), we tested for differences in landscape indices between 2 levels of harvesting, 6 landscapes with no harvest patches (i.e., harvest = 0%) were compared to 12 landscapes (top quartile) with the highest levels of harvesting in this study (13.1 to 34.5% of forest harvested) using student's *t*-test. We used only the top quartile in this case (rather than comparing 0% harvest

Table 3. Mean old-growth patch indices between landscapes with no forest harvesting and high amounts of harvesting for 44 landscapes in the Slocan Valley drainage of southeast British Columbia.[†]

Old-growth index (see Table 1)	Harvest Amount	Mean	SD	*t*	p
PROP (SQ)	zero	2.07	0.45	−0.564	0.581
	high	2.38	1.28		
PATDEN (SQ)	zero	0.35	0.04	−1.055	0.307
	high	0.46	0.25		
COREDEN (SQ)	zero	1.35	0.30	−0.612	0.549
	high	1.57	0.85		
EDGEDEN (SQ)	zero	2.32	0.54	−0.598	0.558
	high	2.67	1.37		
AVGPERM (LG)	zero	7.93	0.23	0.073	0.943
	high	7.91	0.54		
AVGAREA (SQ)	zero	4.53	0.79	−0.0.57	0.956
	high	5.64	1.50		
AVGCORE (SQ)	zero	2.99	0.75	0.004	0.997
	high	2.99	1.35		
AVGEDGE (SQ)	zero	5.02	0.56	−0.142	0.889
	high	5.09	1.18		
MEDAREA	zero	10.29	5.59	0.907	0.378
	high	8.11	4.26		
NN (LG)	zero	5.56	1.08	−0.840	0.414
	high	6.04	1.16		
DISPER (LG)	zero	−1.70	1.05	−1.720	0.106
	high	0.94	0.76		
EDGERAT (LG)	zero	0.22	0.14	−0.531	0.603
	high	0.30	0.35		
CORERAT (AR)	zero	0.44	0.08	0.403	0.693
	high	0.42	0.16		
SHAPE	zero	1.84	0.13	−0.826	0.422
	high	1.91	0.20		
FRACTAL	zero	1.321	0.01	−2.092	0.054
	high	1.338	0.02		

[†]Zero harvest $n = 6$; high harvest landscapes defined as top quartile of all landscapes and ranges from 13.1 to 34.5% of forest harvested; $n = 12$. Only landscapes containing old-growth patches used for mean and ratio index calculations. Data transformations indicated in brackets (SQ = square root; LG = logarithm; AR = arcsine).

to >0% harvest) to increase the chances of obtaining a significant difference, and therefore provide a more rigorous test. In all cases, pooled variance was used in *t*-tests since variances were not significantly different. To investigate the assumption of homogeneity among control replicates and our ability to detect differences between control and treatment landscapes in these tests, we compared variability in the proportion of old-growth in landscapes among control landscapes relative to treatment landscapes. We also investigated the sensitivity of our results to control sample size (n) using a series of sample elimination experiments. We did this by systematic removals of control landscapes and subsequently repeating all *t*-tests (Table 3) with control n ranging from 6 to 2, and then recording any differences in results and conclusions with the original full-sample analyses. We removed control landscapes in 3 ways: (1) beginning with the most variable landscape (i.e., farthest from the mean) and proceeding to the least variable, (2) beginning with the least variable landscape, and (3) a random removal.

Linear regression techniques were also employed to test fragmentation predictions from Forman (1997) concerning relationships between amounts of forest harvesting and fragmentation effects on old-growth forest patterns. Relationships between the total amount of area within a patch type and landscape configuration can be highly correlated (Fahrig 1997). To evaluate harvesting and old-growth patch configuration relationships, independent of old-growth area, partial regression techniques were used to remove area effects (McGarigal and McComb 1995, Trzcinski et al. 1999). In this procedure, the amount of old-growth within each landscape was

regressed against each of the landscape indices measuring old-growth patch configuration. Analysis of residuals was then used to remove confounding area effects for indices showing a significant relationship (regression F-test ratio < 0.05) with old-growth area. In these cases, residuals were regressed against harvest level to test our predictions. Original index values were regressed against harvest level where no area effects were detected. Relationships that should logically pass through the origin were evaluated for use of a zero-intercept model. In all cases however, full regression models were used (rather than zero-intercept models) due to significant differences in y-intercepts from zero (Kozak and Kozak 1995).

Testing for differences in patch patterns associated with the hypothesis that harvesting creates a fragmented pattern of current harvest patches was performed using one-way analysis of variance to test for differences among patch types. Testing for difference in patch patterns associated with the hypothesis that harvesting simplifies the landscapes, variability among patch types was tested using one-way analysis of variance to test for differences in variability among patch types. Mean values of old-growth patch indices were calculated from 6 control landscapes with zero harvest levels. In this way old-growth parameters were derived free of confounding harvest effects on old-growth patches. In cases when significant differences among patch type means were detected Bonferroni post-hoc tests were employed to distinguish differences between patch types (Miller 1985).

In the calculation of mean index values among landscapes, n varied depending on the applicability of the data. In the case of PROP, PATDEN, COREDEN, and EDGEDEN, which are indices related to sheer amounts of habitat, all landscapes were used in calculating means since zeros were considered valid data in these cases. In all other cases, only landscapes containing ≥1 patch (or ≥2 patches in the case of NN and DISPER) of the patch type were used in the calculation of means to avoid spurious results from including zeros (i.e., non-data).

Results

The number of old-growth patches within landscapes ranged from 0 to 40; 0 to 67 for harvest patches; and 0 to 13 for wildfire patches. Old-growth patches were present in 41 of 44 landscapes and accounted for 0.1 to 23.3 % of the forested area in landscapes with old-growth patches (mean PROP = 6.2%, SE = 0.90, n = 44). Harvest patches were present in 38 of 44 landscapes and accounted for 1.3 to 34.5 % of the forested area in landscapes with harvest patches (mean PROP = 9.2%, SE = 1.19, n = 44). Wildfire patches were present in 33 of 44 landscapes and accounted for 0.1 to 62.9 % of the forested area in landscapes with wildfire patches (mean PROP = 4.9%, SE = 1.85, n = 44). Table 2 provides summary statistics for all indices among the three patch types.

No differences were observed between mean old-growth patch index values for 6 control landscapes with no harvesting and 12 landscapes containing the highest levels of harvesting in this study ($-2.092 \leq t \leq 1.16$, all $p > 0.054$; Table 3). Variability in the proportion of old-growth among control landscapes was lower (mean = 4.5%, range = 2.1 to 6.2 %, SE = 0.726, CV = 39.8%) than that for treatment landscapes (mean = 6.5%, range = 0 to 23.3%, SE = 1.028, CV = 97.6%). The results of sample elimination experiments yielded insignificant t-tests in all control landscape removal experiments with the exception of Nearest Neighbour which was significantly lower among control landscapes when control n was 4 in removal experiment (1) and when control n was 2 in removal experiment (3).

Significant relationships were observed in 10 of 12 linear regressions between the amount of old-growth in landscapes and individual old-growth patch indices (PATDEN, COREDEN, EDGEDEN, AVGPERM, AVGAREA, AVGCORE, AVGEDGE, NN, EDGERAT, CORERAT; all $r^2 > 0.244$, all $P < 0.001$). To eliminate confounding area effects, residuals from the 10 significant linear regressions were then used to test fragmentation predictions against amounts of harvesting, while the two indices without confounding area affects (MEDAREA and DISPER) were used without alteration (Figure 2). All subsequent linear regressions between old-growth patch indices (or their residuals) and the amount of harvesting in landscapes (PROP_H) were insignificant (all $r^2 < 0.094$, all $p > 0.054$; Table 4).

When we compared average measures of patch structure among patch types (old-growth, harvest, and wildfire) we found a lower mean fractal dimension for harvest patches than old-growth patches (Table 5). When compared to wildfire patches, harvest patches consistently had higher values for 4 indices associated with habitat amount: PROP, PATDEN, COREDEN, EDGEDEN(Table 5). As well, harvest patches had

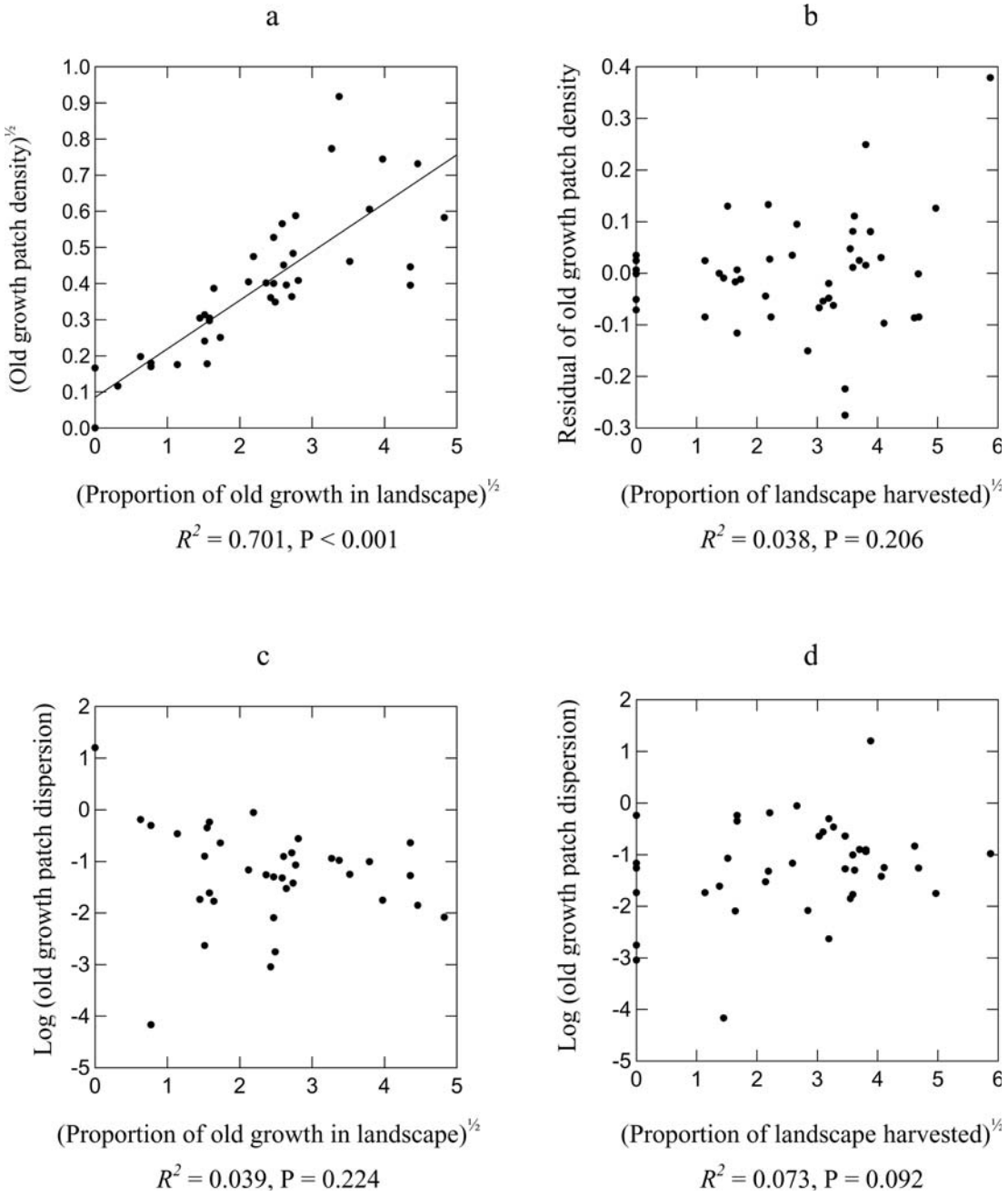

Figure 2. Examples of regression analyses between amount of old-growth and old-growth patch indices (see Table 1), and amount of harvesting and old-growth patch indices for 44 landscapes in the Slocan Valley of southeast British Columbia. All patch index values were regressed against the proportion of old-growth (square root transformed) in landscapes (e.g., a, c); if a significant regression ($p < 0.05$) occurred with the proportion of old-growth (a), the residuals of this regression were used in an a subsequent regression with the proportion of harvesting in a landscape (b) to eliminate confounding area affects; if a non-significant regression occurred with proportion of old-growth (c), the original index was used in a subsequent regression with proportion of harvesting in a landscape (d).

lower SHAPE and FRACTAL values compared to wildfire patches (Table 5). Harvest patches also had lower average nearest neighbor (NN) distances and dispersion index values than wildfire patches (Table 5). Other index values between harvest and wildfire patches were not statistically significantly different.

Table 4. Linear regressions between old-growth patch indices and the amount of harvesting (proportion of forest in clearcuts less than 40 years old) in 44 landscapes in the Slocan Valley drainage of southeast British Columbia.

Old-growth Patch Index† (see Table 1)	*n*	r^2	p
PATDEN_RES (SQ)	44	0.038	0.206
COREDEN_RES (SQ)	44	0.000	0.924
EDGEDEN_RES (SQ)	44	0.001	0.856
AVGPERM_RES (LG)	41	0.003	0.747
AVGAREA_RES (SQ)	41	0.001	0.829
AVGCORE_RES (SQ)	41	0.004	0.706
AVGEDGE_RES (SQ)	41	<0.01	0.990
MEDAREA	41	0.008	0.584
NN_RES (LG)	40	0.094	0.054
DISPER (LG)	40	0.073	0.092
EDGERAT_RES (LG)	41	0.047	0.174
CORERAT_RES (AR)	41	0.022	0.359

†Only landscapes containing old-growth patches used for average and ratio calculations, and only those containing ≥2 old-growth patches used for nearest neighbor and dispersion index calculations; sample size therefore varies. 'RES' indicates a new variable using residual values from a regression with the original index and the amount of old-growth in a landscape to account for confounding area effects. Data transformations indicated in brackets (SQ = square root; LG = logarithm; AR = arcsine).

All mean values between wildfire and old-growth patches were similar with two exceptions. PATDEN was significantly lower for wildfire patches compared to old patches, and NN was significantly higher for wildlife patches compared to old patches (Table 5).

Three of 6 coefficient of variation indices had significant differences among mean values by patch type (Table 6). Coefficient of variation indices of harvest patches were not significantly different than those of old-growth patches in every case (Table 6). Post-hoc tests revealed higher coefficients of variation of harvest patches than wildfire patches for AVGAREA, AVGCORE, and AVGEDGE (all Bonferroni $p < 0.012$). No differences between old-growth and wildfire patch coefficient of variation indices were significant.

Discussion

Old-growth spatial patterns relative to harvesting levels were not consistent with predicted fragmentation patterns described by Forman (1997:407). Landscapes with higher harvest levels did not have old-growth patches that were consistently smaller and farther apart, nor did they have more old-growth edge and less core area than landscapes with little or no harvest levels. Although landscapes are inherently variable, we are confident in our results due to their insensitivity to changes in control sample size within the sample elimination experiments. The failure to detect patterns consistent with old-growth fragmentation in this case presents several possibilities: (1) old-growth fragmentation is not occurring, (2) amounts of harvesting and old-growth are too low to detect a fragmentation pattern, or (3) our measures and tests of fragmentation are inadequate to detect a fragmentation pattern.

The relatively low level of forest harvesting in our study area is very relevant to this issue. Many have predicted critical thresholds in landscape patterns, after which significant effects on ecological process are realized. The overall level of harvesting for all landscapes combined in this study was 9.4% (i.e., 9.4% of the landscape was in a state of recent harvest [<40 yrs]). Studies focusing on amount of forest harvesting as an element of fragmentation cite levels of harvesting that are much higher than those found here. For example, percolation theory (Stauffer 1985) predicts a breakdown in landscape connectivity when > 40 % of habitat is removed (O'Neill et al. 1988). Using a checkerboard simulation model, Franklin and Forman (1987) found patch size remained high until 30% of the forest had been cut and then leveled off when 50% had been cut. On this basis it is very plausible that current levels of harvest in our study area are too low to either detect old-growth fragmentation patterns, or are below a threshold under which fragmentation does not occur. Our results are consistent with Sachs et al. (1998) who investigated forest spatial patterns in southcentral British Columbia. They concluded that a disturbance rate of 8.4% (majority of disturbance was attributed to harvesting) outside parks during a 17-yr period represented the very early stages of fragmentation in this region – well below the thresholds discussed by Franklin and Forman (1987) and harvest rates reported elsewhere in similar studies (e.g., Mladenoff et al. 1993; Spies et al. 1994; Wallin et al. 1996).

As well, the amount of old-growth forests in our study area can be considered relatively low (6.2%; all landscapes combined). Fire patterns in our study area in the last century have produced a current seral stage distribution heavily skewed towards mid-successional forests (unpublished data, BC Ministry of Forests, Castlegar, BC). Patches of old-growth in this case already tend to be relatively small and widely

Table 5. Mean patch indices among old-growth[†] (O), harvest (H), and wildfire (F) patches within 44 landscapes in the Slocan Valley Basin of southeast British Columbia.[§]

Index (see Table 1)	Patch type mean							
	O	H	F	*n*	F	p	Post-hoc test	Bonferroni *p*
PROP (SQ)	2.073	2.625	1.335	94	7.012	0.001*	H – O	1.000
							H – F	0.001*
							F – O	0.892
PATDEN (SQ)	0.354	0.383	0.150	94	20.521	< 0.001*	H – O	1.000
							H – F	< 0.001*
							F – O	0.025*
COREDEN (LG)	0.618	0.965	−0.593	94	10.593	< 0.001*	H – O	1.000
							H – F	< 0.001*
							F – O	0.258
EDGEDEN (SQ)	2.318	2.398	1.169	94	10.082	< 0.001*	H – O	1.000
							H – F	< 0.001*
							F – O	0.143
AVGPERM(SQ)	52.873	58.551	62.832	77	0.777	0.463		
AVGAREA(SQ)	4.527	6.179	6.376	77	0.965	0.386		
AVGCORE(LG)	2.137	2.978	2.881	77	1.521	0.225		
AVGEDGE(SQ)	5.016	5.691	5.992	77	0.694	0.503		
MEDAREA(LG)	2.226	2.746	2.617	77	0.603	0.550		
SHAPE (SQ)	1.355	1.275	1.376	77	5.739	0.005*	H – O	0.470
							H – F	0.004*
							F – O	1.000
FRACTAL (LG)	0.278	0.250	0.273	77	8.007	0.001*	H – O	0.046*
							H – F	0.001*
							F – O	1.000
EDGERAT (LG)	0.221	−0.115	0.166	77	3.251	0.044*	H – O	0.391
							H – F	0.063
							F – O	1.000
CORERATIO (AR)	0.444	0.599	0.494	77	3.703	0.029*	H – O	0.185
							H – F	0.062
							F – O	1.000
NN (LG)	5.558	5.554	7.114	67	14.231	< 0.001*	H – O	1.000
							H – F	< 0.001*
							F – O	0.012*
DISPER (LG)	−1.703	−1.538	−0.633	67	6.665	0.002*	H – O	1.000
							H – F	0.003*
							F – O	0.068

[†]Old-growth statistics calculated from landscapes with zero harvest level (n = 6); [§]All degrees of freedom = 2. Only landscapes containing patches within the patch type were used in the analyses, therefore *n* is variable. Post-hoc tests performed only in cases of significant *F* test result. Statistical significance indicated (*) at $\alpha = 0.05$. Data transformations indicated in brackets (SQ = square root; LG = logarithm; AR = arcsine).

dispersed in harvested and non-harvested areas. Fragmentation, by definition, only occurs when the number of patches increases by the breaking apart of contiguous patches (Fahrig 1997). If whole patches are removed, then fragmentation under this definition is actually decreased since the number of patches decreases (Ripple et al. 1991). Therefore, if old-growth patches are small and can be removed in their entirety by harvesting (i.e., attrition [Forman 1997:407]), landscape indices used to measure fragmentation could reflect a decreasing fragmentation pattern. This phenomenon may explain to some extent the results of our study.

Our failure to detect an old-growth fragmentation pattern could also be related to our questionable ability to quantify and therefore adequately test for fragmentation effects. Hundreds of quantitative measures of landscape pattern have been proposed (Gustafson 1998). However, accepted quantitative definitions of central concepts in landscape ecology such as forest fragmentation remain elusive (Hulshoff 1995; Schumaker 1996; Davidson 1998; Hargis et al.

Table 6. Mean coefficient of variation among old-growth[?] (O), harvest (H), and wildfire (F) patches for 44 landscapes in the Slocan Valley drainage of southeast British Columbia.[§]

Coefficient of variation index (see Table 1)	Patch Type O	H	F	*n*	F	p	Post-hoc test	Bonferroni *p*
CV_AVGAREA	118.57	130.30	83.25	67	5.320	0.007*	H – O	1.000
							H – F	0.006*
							O – F	0.499
CV_AVGCORE	150.67	152.87	103.01	67	5.438	0.007*	H – O	1.000
							H – F	0.006*
							O – F	0.245
CV_AVGEDGE	87.47	95.05	60.39	67	4.506	0.015*	H – O	1.000
							H – F	0.012*
							O – F	0.555
CV_SHAPE	27.05	28.13	20.99	67	2.220	0.117		
CV_FRACTAL	2.67	2.46	1.97	67	2.710	0.074		
CV_NN	110.67	95.62	103.18	58	0.434	0.650		

[†]Old-growth statistics calculated from landscapes with zero harvest level (n = 6); [§]All degrees of freedom = 2. Only landscapes containing patches within the patch type were used in the analyses, therefore *n* is variable. Post-hoc tests performed only in cases of significant *F* test result. Statistical significance indicated (*) at $\alpha = 0.05$. Data transformations indicated in brackets (SQ = square root; LG = logarithm.

1998). D'Eon and Glenn (2000) demonstrated that landscape indices are of limited use in distinguishing between human perceptions of fragmentation or are not sensitive enough to differentiate the critical elements of landscape spatial heterogeneity. Further, the spatial scale at which fragmentation is measured is commonly the scale of human perception, but any potential ecological impacts of fragmentation will occur at different scales for different organisms. So while fragmentation as a conservation issue continues to harbor concern on theoretical grounds, it is unclear if current methods of quantifying landscape pattern adequately address these concerns (Tischendorf and Fahrig 2000).

Comparisons among the three patch types revealed that amounts and spatial patterns of harvest patches differed very little, if at all, from amounts and spatial patterns of old-growth patches within control landscapes, thus falsifying our hypothesis that harvesting creates a more fragmented pattern. All mean values of harvest patch indices were similar to mean values of old-growth patch indices with one exception. The fractal dimension, a measure of patch perimeter complexity, was lower in harvest patches suggesting that harvest patches tended to be of more regular shapes than old-growth patches. However, important indicators of spatial pattern such as patch size and spacing were similar and thus indicate no significant deviation from a more natural patch pattern represented by old-growth patches. Fire patches revealed more discrepancies when compared to harvest patch patterns. Indices associated with habitat amounts indicated more total harvesting within the landscape than area represented by wildfire. We suggest that active fire suppression in recent decades may be a confounding factor leading to a reduction in the amount of recent wildfire in these landscapes than would be present without fire suppression. Considering however, indices associated with spatial pattern only, wildfire patches reflected more complex shapes than harvest patches. This is similar to the case with old-growth patches, where the same pattern was observed. Although, wildfire patches tended to have higher spacing index values (nearest neighbour and dispersion) suggesting that harvest patches are more aggregated and therefore less fragmented, from this point of view.

Variability indices revealed similar variability between harvest patches and old-growth patches, and surprisingly more variability between harvest patches than wildfire patches. This result rejects our hypothesis that harvesting tends to simplify landscape patterns by resulting in patches with less inherent variability than more natural patches. The opposite appears true in this study. We suggest that this phenomenon is due to shifts in forest policy in the four previous decades. Early logging patterns tended towards large continuous clearcuts often hundreds of hectares large. This pattern was reversed towards the 1980s and has resulted in a more recent pattern tending towards smaller dispersed clearcuts often legally restricted to a maximum size (Franklin and

Forman 1987; Province of British Columbia 1996). The combination of these shifting patterns appears to have resulted in an overall harvest pattern that displays high spatial variability in these landscapes.

To sum, little of the evidence presented here demonstrated an obvious old-growth fragmentation pattern that can be attributed to forest harvesting in these landscapes. Clear predictions pertaining to old-growth patch spatial patterns as a result of forest harvesting were falsified. Further, future forest patterns represented by current harvest patches are similar to current spatial patterns of old-growth and wildfire, with the exception of patch shape which appears to be more regular in harvest patches. While patch shape may influence interception rates of individual organisms moving through landscapes (Gutzwiller and Anderson 1992; Hamazaki 1996), the contribution of patch shape to the issue of forest fragmentation is not clear. Spatial configuration aside, habitat amount and related indices in this study accounted for more differences between harvest patterns and more natural patterns. On this basis we agree with Fahrig (1997) and Trzcinski et al. (1999) and suggest that total habitat amount is more of a concern to the fragmentation issue than spatial configuration of patches in this case. An important consideration here is that our data represent the detection of spatial patterns, and not direct effects of spatial pattern on individual organisms. While we did not detect significant spatial patterns consistent with the fragmentation paradigm, this finding does not preclude effects on organisms related to other causes and other spatial pattern issues (e.g., D'Eon et al. 2002).

A limitation of this study, as discussed, was the relatively low proportion of harvesting and old-growth in these landscapes. We recommend therefore that additional empirical tests of the fragmentation paradigm similar to those presented here be performed in landscapes with larger amounts of harvesting and old-growth forest. Another limitation of this study is our singular focus on spatial pattern, without the consideration of temporal patterns and trends. Since fragmentation is often thought of as a process occurring through time, temporal trend analyses represent an important avenue for future study in this landscape. Another limitation of the study was the use of wildfire patches to reflect attributes associated with a naturally occurring landscape pattern. Despite debate over its influence, active fire suppression may have served a role in modifying wildfire patterns in recent decades (Baker 1993). As well, in considering only wildfire patches <40 yrs old, we may not have captured the full range of wildfire-patch spatial characteristics that might result from considering patches over a longer time period. An ideal repetition of this study, therefore, would be performed in landscapes with an unsuppressed fire regime over a longer time period. Finally, a finding that harvest patches have similar spatial configuration to more naturally-occurring patches (at least to the extent that we have measured it) could be explained by the naturally-heterogeneous nature of forests in these landscapes. Abrupt changes in terrain and other biophysical attributes associated with steep mountainous areas produce a landscape mosaic that is inherently heterogeneous and therefore may reflect an already naturally fragmented spatial pattern that does not differ from harvest patterns. The most relevant question then becomes 'is harvesting creating a more fragmented pattern than the inherent heterogeneity of the landscape?' Repeating a study of this nature in other less heterogeneous landscapes such as boreal forests may produce differing results.

Acknowledgements

We thank Slocan Forest Products for their support of this research. Graduate student support to R. D'Eon was provided by the University of British Columbia Faculty of Forestry, the National Engineering Research Council of Canada, and a Canfor Corporation fellowship in forest wildlife management. D. Mack performed all GIS analyses and landscape metric calculations. We are thankful to L. Fahrig for her insightful comments and advice concerning confounding area effects, and W. Reid for his helpful comments on landscape variability and study design.

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