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Preliminary Analysis on the Impacts of Mountain Pine Beetle Management on the Spatial Pattern of Forests—MPBI Working Paper.

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Abstract

The current mountain pine beetle epidemic in British Columbia led to new forest management scenarios and the removal of large areas (>1000 ha) of forest. New harvesting blocks may well be within the range of natural disturbance for this region providing sufficient retention of forest cover. Change to forest composition causes change to forest spatial pattern, which impacts a number of ecological processes such as wildlife habitat use, and hydrology. We show that in mountain pine beetle infested regions, intense harvesting is impacting the spatial pattern of forest and non-forest cover. The result is a forested landscape that has become increasingly fragmented over a short-period of time. Forest edge habitat along with the number of forest patches is increasing, while the average area of these patches is decreasing. Large non-forest components are being created as a result of large harvesting operations and a conglomeration effect with prior cut-blocks. Future work must investigate the impact of the changing spatial pattern on local wildlife and various physical processes.

Keywords: mountain pine beetle, landscape metrics, forest fragmentation, salvage harvesting

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1 Introduction

The mountain pine beetle (*Dendroctonus ponderosae*) is the most significant biological agent of mortality in mature pine stands in western North America (Shore et al. 2006a). Mountain pine beetle primarily attacks mature lodgepole pine (*Pinus contorta*) stands (Safranyik & Carroll 2006), which account for approximately 25% of the standing forest inventory within the province of British Columbia, and as much as 50% within the interior regions (Wagner et al. 2006). As such, epidemic mountain pine beetle populations have significant impacts on the forest harvest sector, as well as the environmental services of the natural landscape (Wagner et al. 2006).

After boring holes into the bark at the base of a pine tree, the mountain pine beetle feeds on phloem, thus degrading the passageway for water and nutrients to the crown, leading to eventual tree mortality (Safranyik & Carroll 2006). The mountain pine beetle population can remain at incipient levels until suitable climatic and habitat conditions are achieved, facilitating the outbreak of an epidemic (Taylor & Carroll 2004). Fire suppression, and preferential harvesting of non-pine species in the years leading up to the 1990's, has created an abundance of mature pine forest stands in the interior regions of British Columbia (Taylor & Carroll 2004). Accompanied by a warming climate, which hinders mortality in mountain pine beetle populations (Carroll et al. 2004), this convergence of factors has allowed the current mountain pine beetle epidemic to reach an unprecedented magnitude and extent. The current outbreak is expected to expand causing mortality in a significant proportion of the province's merchantable pine timber within the next decade (Westfall, 2007).

Knowledge of the biology and dispersal of the mountain pine beetle has led to two different methods for mitigating mountain pine beetle success: direct control and preventive management. Direct control includes tactics aimed at suppressing beetle populations, while the goal of preventive management is to increase stand vigour or reduce the amount of susceptible pine within a landscape (Shore et al. 2006b). Preventive management efforts were under-employed in the early phase of the current epidemic (Wilson, 2004), and direct control measures are ineffective at the epidemic stage (Safranyik et al. 1974). In response to the magnitude of the current epidemic, harvesting efforts are being focused on the salvage of standing dead timber, and less on the suppression of the spread of mountain pine beetle (Eng 2004).

Standing dead timber, resulting from mountain pine beetle infestations, has an associated shelf-life, or time for which it remains economical for harvest (British Columbia Ministry of Forests and Range 2007b). As a result, there has been economic pressure on the government of British Columbia to allow increased harvest for the short-term salvage of standing dead timber (Wagner et al. 2006). Government initiatives have been put in place to increase the allowable annual cut (AAC) in the hardest hit forest management districts (British Columbia Ministry of Forests and Range 2007b), and to redirect existing harvest activities to mountain pine beetle infested areas (Wagner et al. 2006). In doing so, larger timber volumes are being harvested over shorter time periods primarily through large-scale salvage operations. Large-scale salvaging of mountain pine beetle infested stands provides the opportunity to open up large (>1000 ha) sections of forest, which are within the range of natural disturbance, if proper legacies of standing forest are maintained (Eng 2004). This change in harvesting is in contrast to the situation prior to the outbreak which was characterized primarily by cut-blocks of <60 ha (Eng, 2004).

There exist significant economic (British Columbia Ministry of Forests and Range 2007b), and ecological (Chan-McLeod 2006; Lindenmayer et al. 2004) benefits to intact mountain pine beetle attacked stands due to understory characteristics and the existence of non-pine species. Following mountain pine beetle mortality, stands of mixed species, or those with a developed understory, show a short-term spike in growth, often leading to the propagation of late successional species (Hawkes et al. 2004). These tolerant species provide a source of habitat (Chan-McLeod 2006), and future timber supply (British Columbia Ministry of Forests 2007b) more rapidly than those areas which have been cleared during salvage logging. As such, salvage harvesting efforts are being focused predominantly on those stands composed of at least 70% pine, with little to no understory composition (British Columbia Ministry of Forests 2007a).

New approaches to harvesting mountain pine beetle infested forests are changing the spatial pattern of the forest on the landscape. The spatial pattern of a forested region can be interpreted as a combination of the proportion of forest and its configuration on a landscape (Riitters et al. 2002). The spatial pattern of forest has been demonstrated to impact a variety of natural processes. For instance, clear-cut harvesting, a key cause of change in the spatial patterns in forests (Franklin & Forman 1987), impacts hydrologic regimes (Foster et al. 1997; Helie et al. 2005), soil quality and nutrient retention (Dahlgren & Driscoll 1994), the carbon budget (Kowalski et al. 2003) and wildlife habitat use (Bunnell et al. 2004; Tinker & Knight, 2000).

The objective of this research is to quantify the effects of the current mountain pine beetle epidemic and its management on the spatial pattern of the forest / non-forest landscape components. Of focus will be the change in forest pattern between two time periods, 2000 and 2006, representing the early and peak stages of the current mountain pine beetle epidemic (British Columbia Ministry of Forests and Range 2007b). To meet this objective, an existing landcover dataset representing forest conditions in the early epidemic stage (2000) is obtained. Data from 2000 are updated to reflect forest conditions in 2006 using remotely sensed imagery. Landscape pattern indices (LPIs) are then employed to quantify the spatial pattern of the forested landscape in both time periods for an area within the Prince George Forest District, which is known to be impacted by the mountain pine beetle. It is expected that, as a result of management activities, the forested landscape will be becoming increasingly fragmented. Given that these trends in salvage harvesting are expected to continue at least over the short-term, the future impacts to the spatial pattern of the forested landscape will also be considered.

2 Material and Methods

2.1 Study area

Located within the interior plateau of British Columbia, the Prince George forest district (Figure 1) is situated primarily in the Sub-Boreal Spruce (SBS) biogeoclimatic zone (Meidinger & Pojar 1991). The SBS biogeoclimatic zone is characterized by extreme climatic fluctuations across seasons, with hot, moist summer months, followed by long, dry and cold winters. White spruce (*Picea glauca*), subalpine fir (*Abies lasiocarpa*), and lodgepole pine (*Pinus contorta*) are the dominant forest species within this region. The Prince George forest district has experienced severe timber losses from mountain pine beetle infestations, and has been designated as a region where an increased AAC will be prescribed for the short-term future (British Columbia Ministry of Forests and Range 2007b). The significant increase in salvage activities over the short term provides a suitable case study for evaluating the effects of mountain pine beetle management activities on the spatial pattern of the forested landscape.

Within the Prince George forest district, two sites were chosen as representative of landscapes impacted by the mountain pine beetle, both through infestation and resultant salvage and mitigation activities. Each site is 40 km by 40 km (160 000 ha) in size, which is large enough to contain several large openings (1000 ha) without being impacted by non-forestry activities or urban environments. Sites were selected using previous mountain pine beetle survey data that indicated existing attack locations (Nelson et al. 2006) and through visual interpretation of satellite imagery. Study site 1 was intensely infested by the mountain pine beetle and includes several large-scale salvage harvesting activities, which occurred between 2000 and 2006. Study site 2 has also been heavily impacted by mountain pine beetle infestations, but contains less harvesting. It is expected that based on the observed differences in harvest intensity, seen in sites 1 and 2, there will be greater impacts to the spatial structure of the forested landscape in site 1.

2.2 Data and pre-processing

2.2.1 2000 Forest Data

As part of the Earth Observation for Sustainable Development of forests (EOSD) program, circa year 2000 landcover mapping, has been completed for the entirety of Canada's forested regions (Wulder et al. 2003). The spatial resolution of the EOSD dataset (25m), originally derived from Landsat imagery, has been identified as an applicable scale at which to monitor landcover conditions for national and regional assessment in both the United States (Vogelmann et al. 2001) and Canada (Wulder et al. 2003). Using a top of atmosphere (TOA) radiometric correction method for Landsat-7 ETM+ imagery, an automated unsupervised classification system was employed to compile landcover information. For further details on the processing and compilation of the dataset, refer to Wulder et al. (2004). The accuracy of the EOSD dataset has been examined in select areas (including the Prince George region), using differing methods and has demonstrated the validity of this dataset (Remmel et al. 2005; Wulder et al. 2006; Wulder et al. 2007). The EOSD data are comprised of a nested class hierarchy with 23 classes at the highest level of categorical detail, which can be aggregated down to three classes: forest, non-forest, and other (Wulder & Nelson 2003). The forest class contains coniferous, mixed and broadleaf forests distinguished as dense, sparse, or open. Also included in the forest class are forested wetlands. The non-forest class contains cover types such as herbs, grasslands, bryoids, and wetlands. It also includes the exposed land class associated with clear-cuts and agricultural activities. The "other" landcover class component includes regions which are non-vegetated, including water bodies, snow/ice, and rock/barren land. Areas where the imagery has cloud and shadow are included in the "other" class. It is important to note that the landcover definitions used in this study represent a snapshot in time. Forest and non-forest components are temporally dynamic and much of the non-forest delineated in this study is expected to return to forest over time.

2.2.2 2006 Forest Data

In order to consider how the spatial pattern of forest changes due to mountain pine beetle infestation and related harvesting, a similar data set was required for 2006. Two Landsat-7 ETM+ scenes (Path: 48, Rows: 22 & 23) used in the EOSD landcover program from the year 2000, were obtained. As the Landsat-7 ETM+ sensor is no longer fully functional, Landsat-5 TM scenes for the same Path, and Row numbers were acquired for the year 2006. The 2000 images had been previously processed to TOA reflectance as per EOSD guidelines (Wulder et al. 2004). The 2006 images were also processed to TOA reflectance using the same methods as the EOSD program, and an image-to-image normalization procedure was conducted to facilitate a change detection process (Hall et al. 1991; Han et al. 2007).

The enhanced wetness difference index (EWDI) was used to update forest conditions from the year 2000 to 2006. Based on the wetness component of the tasseled cap transformation (TCT) (Crist & Cicone 1984; Kauth & Thomas 1976), the EWDI has demonstrated success in the detection of forest harvest conditions (Franklin et al. 2001; Franklin et al. 2002), and mountain pine beetle red attack (Skakun et al. 2003). High-positive values of the EWDI are representative of harvested forested regions, while mid-to-high-positive values are representative of mountain pine beetle red attack (Skakun et al. 2003). Although the EWDI is sensitive to both changes in vegetative loss and recovery, its application in monitoring forest regeneration requires development (Franklin et al. 2001). Forest regeneration in previously cut areas was assumed to be negligible based on the temporal scale used in this study (6 years). Only those regions, where harvesting has occurred, were considered to have undergone change.

Using the EWDI values, harvested regions were delineated and stored in a binary grid representing areas of forest loss and no-change. Locations with no-change were given the same class (forest / non-forest / other) as existed in 2000. For locations of forest loss, the non-forest class was assigned. The result of this process was a 2006 dataset of forest / non-forest / other classes at a 25m spatial resolution.

2.2.3 Landscape Metrics

Landscape metrics, often used in quantifying the spatial pattern of forests (e.g., De Clercq et al. 2006; Frohn & Hao 2006; Haines-Young & Chopping 1996; Trani & Giles Jr. 1999), were calculated for the 2000 and 2006 datasets classified as forest / non-forest / other. Hundreds of landscape metrics exist and are applicable when analyzing the spatial patterns in categorical data (such as forest / non-forest). Most landscape metrics analyze the properties of specific patch types or of the landscape as a whole (Boots 2003; Gustafson 1998). Often when using landscape metrics to quantify forest pattern, a large number of metrics are computed and researchers determine which provide interesting insights for analysis. Our approach is to base the selection of metrics on knowledge of how beetle and management related processes are impacting the spatial patterns of the forest, as this aids in interpretation (Li & Wu 2004). Four metrics were chosen based on the work pertaining to the use of landscape metrics in forested regions (De Clercq et al. 2006; Frohn & Hao 2006; Haines-Young & Chopping 1996; Trani & Giles Jr. 1999) and definitions of applicable metrics based on sensitivity studies (Hargis et al. 1998; Li et al. 2005; Riitters et al. 1995).

The characterization of spatial pattern can generally be divided into two components: *composition* and *configuration* (Gustafson 1998; Li & Reynolds 1995). Percent forest cover (%FC) was chosen to characterize how the amount or composition of forest cover has changed over time. In forested landscapes changes to composition from disturbances are often the driver of changes to spatial pattern (Foster et al. 1998; Franklin & Forman 1987). Three measures of configuration were chosen to evaluate how mountain pine beetle harvesting activities are changing the arrangement of forest on the landscape. Edge density (ED), in m/ha, was chosen to enable the quantification of edge properties, which have been identified as a key component in promoting wildlife and plant diversity in forested landscapes (Ranney et al. 1981). Number of patches (NP) for forest and non-forest patches, along with mean patch area (MPA), in ha, for forest and non-forest patches were used to evaluate how the patch components of forest and non-forest are changing through time. Both number of patches and patch area have been identified as key factors for measuring the level of fragmentation in forested landscapes (De Clercq et al. 2006; Fahrig 2003; Gergel 2007).

Each of the two study sites were segregated to create three scales of analysis resulting in one 40 km^2 landscape, nested with sixteen 10 km^2 landscapes, and sixteen-hundred 1 km^2 landscapes, for each year (see Figure 2). The 40 km^2 landscapes contain 2560000, 25 m^2 pixels, while the 10 km^2 and 1 km^2 landscapes contain 160 000 and 1600 pixels respectively. The landscape metrics identified above were calculated for both time periods, for each site, and each landscape at all three scales using the APACK 2.23 software package (Mladenoff & DeZonia 2004). In addition, the change between the metric results for 2006 and 2000 was compiled, where change represents the landscape metric value in 2000 subtracted from the landscape metric value in 2006. The co-efficient of variation was also calculated for the smaller landscape (1 km^2 & 10 km^2) scales. The co-efficient of variation can be used for evaluating the variability in the smaller landscape (1 km^2 & 10 km^2) metric values. It is expected that trends observed at the broadest scales may not be seen at finer scales. Results are interpreted based on the known harvest characteristics distinguishing the two study sites, and the onset of increased management due to mountain pine beetle infestations between the years 2000 and 2006.

3 Results and Discussion

3.1 Results

Study site 1 showed an 18.1–18.2 % decrease in percent forest cover (%FC) (see Table 1) related to the intense harvesting activities occurring in this region. The edge density (ED) in site 1 increased at all three scales 10.3–10.6 m/ha overall (see Table 2). While the number of forest patches (NP-F) showed a large increase between 2000 and 2006 (Table 3), decreases in the mean forest patch area (MPA-F) (Table 4) were also seen. The number of non-forest patches (NP-NF) showed a substantial decrease in site 1 (Table 5), while the average area of these patches (NP-NF) increased overall (Table 6).

The results for study site 2 showed similarities with the results seen in site 1. An overall decline in %FC was once again seen, however the magnitude of this loss was reduced (see Table 1). ED was initially larger in site 2 than in site 1, but showed an overall smaller increase between 2000 and 2006 (see Table 2). NP-F (see Table 3) increased and MPA-F (see Table 4) decreased as was observed in site 1. The magnitude of these changes are substantially lower than those in site 1. NP-NF increased in site 2 (Table 5) which is the opposite to what was seen in site 1. MPA-NF increased (see Table 6), which is what was observed in site 1.

The smallest scale of analysis in this study (1 km^2 landscapes), contains a higher level of variation in all measures than the 10 km^2 scale. Several of the 1 km^2 (100 ha) landscapes used in this study could easily be contained within a single large-scale harvesting operation and could therefore be subject to a wide range of decreases to %FC (97%–0%), as seen in Figure 3. This suggests that at the 1km^2 scale, spatial pattern characteristics are likely changing non-uniformly across the study region.

Further examination of the spatial distribution of change in forest pattern using the 1 km^2 landscapes indicates that, although general trends can be discerned at a broad scale, local trends may in fact be different. For example, finer scale landscapes with the largest declines in %FC may not coincide with those with the largest increase in ED (see Figure

3), which contradicts the relationships seen at larger scales. This type of non-relationship can also be seen when examining some of the other general trends observed. Areas showing the greatest increase in NP-F in site 1 do not necessarily correspond with areas showing the largest decreases in MPA-F (see Figure 4). This suggests that the general trends observed at the broad landscapes scales used for many analyses, may not be translatable to smaller landscape scales.

3.2 Discussion

Fragmentation can be broadly defined as the breaking up of a habitat, ecosystem, or land-use type into smaller parcels (Forman 1995). It was the aim of this study to use metrics relating to studies in forest fragmentation in order to quantify the spatial pattern effects seen as a result of changing harvesting scenarios in response to the mountain pine beetle. The largest component of the process of forest fragmentation is forest loss, however forest can be removed in a variety of ways resulting in many different spatial configurations (Fahrig 2003). It is therefore necessary to quantify the resulting configuration of forest and non-forest components. Based on the literature, edge density, number of patches, and mean patch area were identified as principal measures of configuration in forested landscapes.

Edge habitat has been identified as an important habitat structure for many species of wildlife, and influences a number of physical processes that occur in forested landscapes (Ranney et al. 1981). Both regions examined in this study showed increasing edge as a result of both intense and moderate harvesting activities. The intensely harvested region (site 1) showed a larger increase in ED than the moderately harvested area (site 2). Despite this, within study site 1 the largest increases in ED were not necessarily in those areas with the largest changes in %FC. Previous work has shown that in two class landscapes (this study used three, but the 'other' class was much less prevalent than 'forest' and 'non-forest') the relationship between edge density and proportion of one class follows a symmetrical curve which peaks when each of the classes are in equal proportion (50% each) (Remmel et al. 2002). Thus, a landscape comprised of 80% forest and a landscape comprised of 80% non-forest, are expected to contain a similar amount of edge. It is likely that this phenomenon is what we are observing (Figure 3). Changes in ED are dependant on the nature of the changes to %FC.

Number of patches and average patch size are measurements that have been traditionally associated with measuring fragmentation (Gergel 2007). The number and size of forest patches located within a landscape will have impacts on the type, quality, and quantity of habitat available for wildlife in the region (Andren 1994). The importance of large patches of forest habitat for maintaining viable wildlife populations has been identified in conjunction with the theory of island biogeography (Harris 1984). The size of forest patches as measured by MPA-F has decreased as a result of salvage-harvesting activities in the each of the sites. The level of decrease in MPA-F was noticeably larger in site 1 where more intense harvesting was observed. As a result, there may be negative implications for species requiring larger contiguous patches of forest cover. An increasing number of forest patches have been identified as key components in quantifying fragmentation of forested landscapes (Trani & Giles Jr. 1999). Both sites in this study showed increasing number of forest patches, indicating as expected, that harvesting activities are further fragmenting these forested areas.

Many studies in forested landscapes are often focused on measuring quantities relating to the amount and spatial structure of forest components. In this study we also investigated how the changes were affecting the non-forest components. In site 1, where the removal of forest cover has been higher, the change in non-forest patch characteristics have provided some interesting results. Here the number of non-forest patches has actually decreased in number between 2000 and 2006, despite an increase of 18.2% in non-forest cover. The average size of these patches has increased. Harvesting practices have been leading toward larger (>1000 ha) openings in the forest cover (Eng, 2004). However, what is also being seen is the removal of portions of forest that had originally been left as buffers between existing cut-blocks, and non-forested regions. This has created a conglomeration effect between the remnant cut-blocks from prior to 2000, and the regions harvested since. This may be a result of existing logging infrastructure (i.e., roads, landings), which would facilitate more economical and timely harvesting opportunities. In site 2, where the level of non-forest added to the landscape was noticeably smaller (4.7%), this conglomeration effect was not seen. Here the number of non-forest patches increased, as well as their mean patch area. The magnitude of the changes in site 2 were noticeably smaller then those seen in site 1. Intense harvesting of pine forests in response to the mountain pine beetle are causing spatial pattern changes to the non-forest patches not seen in typical harvesting scenarios.

4 Conclusions

In response to the mountain pine beetle epidemic in British Columbia, the clearing of large portions of the forested land base is occurring in select regions. There exist noticeable changes to the spatial pattern of the forested landscape when comparing intensely harvested (site 1) and moderately harvested (site 2) landscapes. Large scale harvesting may be creating large openings in forest cover not seen since the implementation of forest management and intense fire suppression activities. The size of these openings may well fall within the natural variation for interior British Columbia coniferous forests. Despite this, forest management in this region has traditionally comprised the creation of a mosaic of smaller openings (<60 ha) within a largely forested landscape (Eng 2004). Regions of largely pine composition forest are being targeted for intense salvage harvesting opportunities. It is in these areas where the spatial pattern effects of these activities will be most noticeable. When considering large scale harvesting opportunities, forest management should include proper legacy components as identified in Eng (2004). Of equal importance may be the maintenance of forest islands between new harvesting blocks and previously cleared areas where sufficient forest cover has yet to be developed. The spatial pattern effects of the large forest openings created by salvage harvesting activities are magnified when coupled with pre-existing areas of little or no forest cover (i.e., prior cut-blocks, grasslands).

Based on the observed results from the sites examined in this study, expected changes to British Columbia's interior forests include: large decreases to forest cover in regions targeted for salvage harvesting, increasing amounts of forest edge, forest patches that are increasing in number while decreasing in size, and non-forest patches that are increasing in size and decreasing in number. Future work should examine specific implications of these changes on keystone wildlife species in these regions. There may also be impacts on various physical processes that need to be further quantified. New methods for quantifying the spatial pattern qualities of categorical data, such as local indicators for categorical data (LICD), may lead to opportunities for comparing landscapes through time (Csillag & Boots 2005), and should be explored. Finally, it seems that the relationships seen at the broadest scale may not transfer to finer scales. This has implications in the choice of scope for the evaluation of spatial pattern, and how results can be interpreted. If small landscapes are used to map the distribution of spatial patterns across a larger region, these relationships need to be considered.

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Table 1. Percent forest cover (%FC) results.

Percent Forest Cover (%)		Site 1				Site 2		
		2000	2006	Change		2000	2006	Change
1 km x 1 km								
(n=1600)	mean	72.3	54.2	-18.1	mean	76.8	72.1	-4.7
	CV	0.28	0.47	-1.19	CV	0.23	0.26	-2.31
10 km x 10 km								
(n=16)	mean	72.3	54.1	-18.2	mean	76.8	72.1	-4.6
	CV	0.09	0.14	-0.27	CV	0.07	0.09	-0.65
40 km x 40 km								
		72.3	54.1	-18.2		76.7	72.1	-4.7

Table 2. Edge density (ED) results.

Edge Density (m/ha)		Site 1				Site 2		
		2000	2006	Change		2000	2006	Change
1 km x 1 km								
(n=1600)	mean	76.0	86.3	10.3	mean	97.5	103.3	5.8
	CV	0.60	0.51	1.57	CV	0.52	0.48	1.94
10 km x 10 km								
(n=16)	mean	77.4	88.0	10.6	mean	98.2	104.1	5.9
	CV	0.14	0.11	0.36	CV	0.14	0.13	0.52
40 km x 40 km								
		77.6	88.1	10.5		98.4	104.4	6.0

Table 3. Number of forest patches (NP-F) results.

Num. Patches (F)		Site 1				Site 2		
		2000	2006	Change		2000	2006	Change
1 km x 1 km								
(n=1600)	mean	5.5	7.6	2.1	mean	4.5	4.9	0.4
	CV	1.03	0.80	1.37	CV	0.89	0.85	2.96
10 km x 10 km								
(n=16)	mean	301.3	470.0	168.7	mean	209.6	238.1	24.5
	CV	0.46	0.31	0.29	CV	0.35	0.35	0.86
40 km x 40 km								
		4583	7160	2577		3164	3594	430

Table 4. Mean forest patch area (MPA-F) results.

Mean Patch Area (F) (ha)		Site 1				Site 2		
		2000	2006	Change		2000	2006	Change
1 km x 1 km								
(n=1600)	mean	36.5	20.2	-16.3	mean	36.0	31.2	-4.8
	CV	0.95	1.36	-1.59	CV	0.90	0.98	-3.01
10 km x 10 km								
(n=16)	mean	30.0	12.7	-16.3	mean	40.4	33.5	-6.9
	CV	0.45	0.38	-0.59	CV	0.43	0.43	-0.74
40 km x 40 km								
		24.9	11.9	13.0		76.7	72.1	-4.6

Table 5. Number of non-forest patches (NP-NF) results.

Num. Patches (NF)		Site 1				Site 2		
		2000	2006	Change		2000	2006	Change
1 km x 1 km								
(n=1600)	mean	8.3	7.6	-0.7	mean	10.3	10.7	0.4
	CV	0.71	0.78	-3.66	CV	0.62	0.95	3.79
10 km x 10 km								
(n=16)	mean	591.4	491.4	100.0	mean	751.0	766.3	15.3
	CV	0.31	0.34	-0.42	CV	0.24	0.24	1.68
40 km x 40 km								
		9185	7592	-1593		11660	11875	215

Table 6. Mean non-forest patch area (MPA-NF) results.

Mean Patch Area (NF) (ha)		Site 1				Site 2		
		2000	2006	Change		2000	2006	Change
1 km x 1 km								
(n=1600)	mean	6.9	15.7	8.8	mean	3.9	4.5	0.6
	CV	1.81	1.46	2.20	CV	2.02	1.90	7.34
10 km x 10 km								
(n=16)	mean	5.3	10.6	5.3	mean	3.2	3.7	0.5
	CV	0.50	0.44	0.49	CV	0.42	0.39	0.86
40 km x 40 km								
		4.8	9.5	4.7		3.0	3.6	0.6

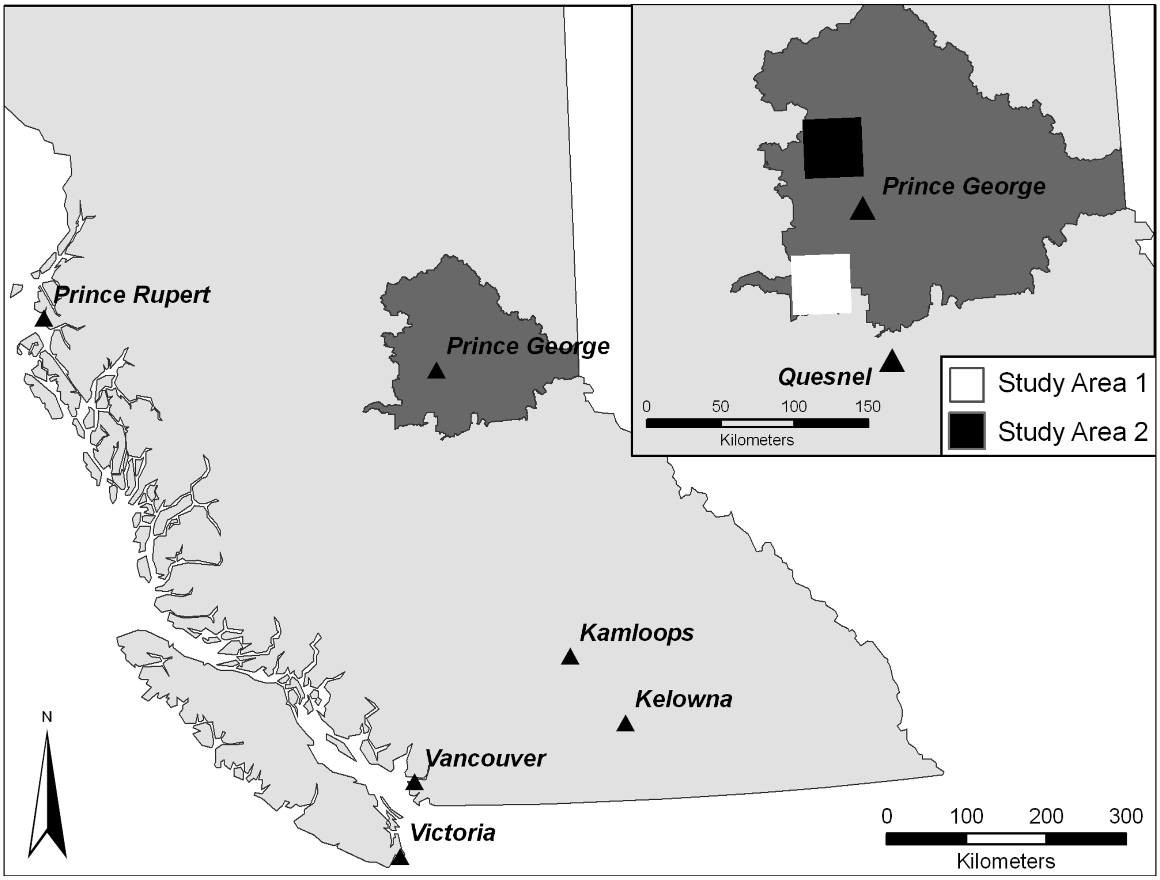

Figure 1. Study area map delineating two study sites within the Prince George forest district in the interior region of British Columbia.

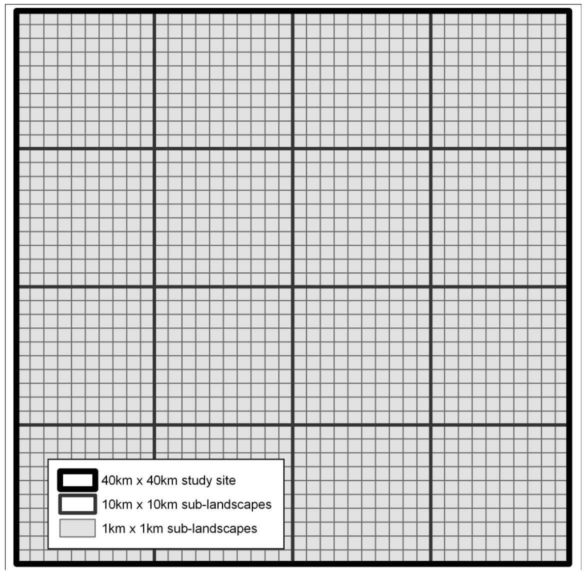

Figure 2. Creation of three scales of analysis for the calculation of landscape metrics.

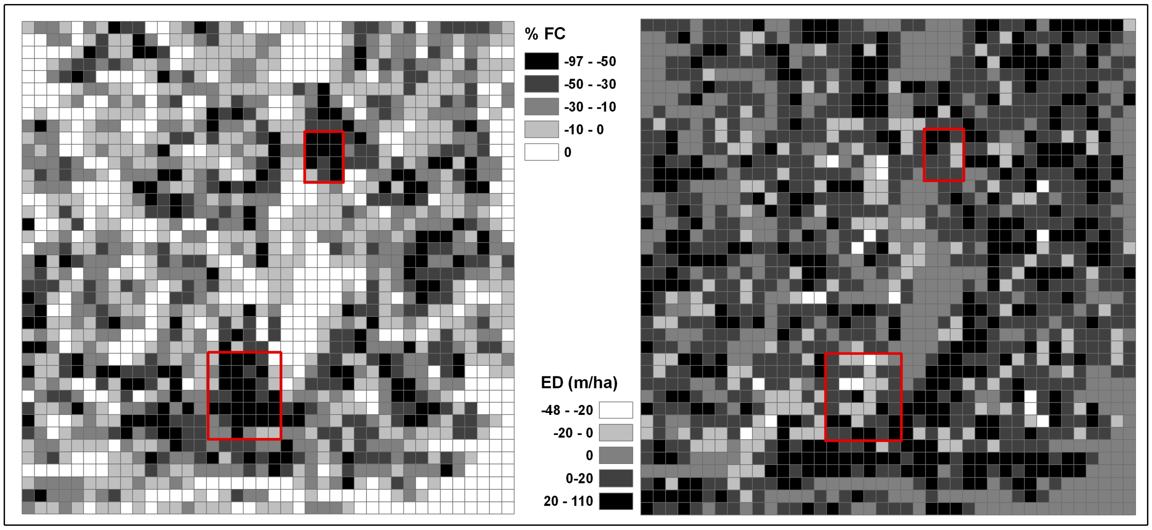

Figure 3. Distribution of change in %FC (left) and ED (right) using 1km x 1km landscapes for study site 1. Highlighted by the red boxes, dark areas in the left image indicating large decreases in %FC that do not necessarily coincide with the regions where the largest increases in ED (dark areas on right image) occur.

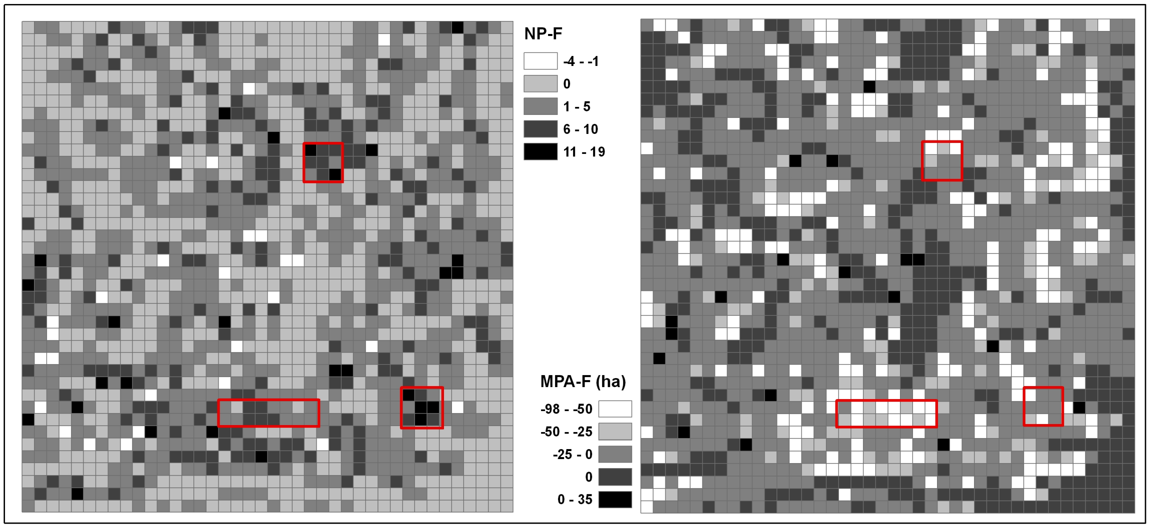

Figure 4. Distribution of change in NP-F (left) and MPA-F (right) using 1km x 1km landscapes for study site 1. Highlighted by the red boxes, dark areas in the left image indicating large increases in NP-F do not necessarily coincide with the regions where the largest decreases to MPA-F (light areas in right image) occur.