The detection and mapping of large scale changes to forested landscapes is increasingly important in ecology and management. I used Landsat TM and MSS imagery to map forest cover in 1992 for a 4.2 million ha area of the interior of British Columbia with an overall classification accuracy of 79%. A combination of unsupervised and supervised classification allowed mapping of forested pixels into closed conifer, semi-open conifer, deciduous, and mixed deciduous and coniferous forest classes. The closed conifer class was further subdivided into old (> 140 years), mature (60 - 140 years), and young (< 60 years) age classes. Unsupervised classification of a combined image containing the Tasseled Cap indices from 1975 and 1992 imagery allowed accurate mapping of disturbances during the period.

I examined changes in landscape pattern from 1975 - 1992 by calculating indices that describe overall landscape pattern and that of each cover class. The B.C. biogeoclimatic ecosystem classification map was used as an overlay to calculate landscape pattern and changes by biogeoclimatic zone. Analysis showed that 11.4 % of the forested area outside provincial parks was disturbed during the 17 - year period. Disturbed areas were consistently much smaller than conifer patches in all biogeoclimatic zones and had a lower percentage of interior area. Conifer patch shape complexity varied between zones, but the size and shape of disturbed areas were similar in all zones. Results indicated the early stages of fragmentation of this landscape.

Models of stand level carbon storage and carbon stores in forest products were used in conjunction with British Columbia Ministry of Forests inventory data to calculate a carbon budget for the Engelmann spruce - subalpine fir forests of the study area. Based on inventory data, the net flux of carbon from these forests to the atmosphere averaged
0.14 Mg ha\(^{-1}\) year\(^{-1}\) between 1975 and 1992. Assuming no harvesting during this period, these forests would have been a net carbon sink averaging 0.39 Mg ha\(^{-1}\) year\(^{-1}\). Calculating the carbon budget using satellite data showed a slight net carbon sink of 0.15 Mg ha\(^{-1}\) year\(^{-1}\) for 1975 - 1992. The major difference between satellite- and inventory-based carbon budgets was due to differences in the simulation of the detrital carbon pool and timing of harvests. The satellite-based estimate failed to account for recovering areas harvested before 1975, and assumed a constant harvest rate.
Regional Changes in Landscape Pattern and Carbon Stores in the Interior of British Columbia as Determined by Satellite Imagery

by

Donald L. Sachs

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Chapter 1. Introduction

Forestry in western North America has traditionally focused on the management of forest stands, predominantly for timber production. However, the scope of forest management is broadening as it has become increasingly clear that the cumulative effects of management decisions will be felt at regional or even global scales, and have lasting legacies. The role of forests in regional biodiversity has been demonstrated by the spotted owl controversy in the Pacific Northwest. Obviously, forest management can fundamentally alter the pattern of entire landscapes in a relatively short period, and these changes can have major effects on many of the species present. During the past decade there have been calls for a shift towards ecosystem management at the landscape scale (e.g., FEMAT 1993). At the global scale, potential climate change due to rapid accumulation of radiatively-active trace gases in the atmosphere has spawned fervent study of the global carbon budget. The world's forests play an important, yet poorly quantified role in global carbon cycling, and improved understanding is essential. In the future, it seems that the effects of forest management on the overall carbon balance will have to be considered in regional and national forest planning.

Publicly owned forest lands cover approximately 60% of British Columbia, and their management is extremely important both ecologically and economically. A significant portion of lands on Vancouver Island and in southwestern B.C. have been harvested, and, as in the Pacific Northwest of the U.S., the management of remaining primary forest lands in B.C. has become increasingly controversial. The Provincial government has responded with the new Forest Practices Code, which mandates that forest management will, among other things, conserve biodiversity, and meet present needs without compromising the needs of future generations. Also enacted is a comprehensive Protected Areas Strategy that seeks to set aside 12% of provincial lands
representing all ecosystems. Clearly the role of forest management in B.C. is changing, and planning at the landscape and regional scales is now mandated.

The changing focus of management coupled with rapid pace of change in forested ecosystems presents problems for traditional mapping and inventory methods. To track the pace of human activities and address management issues at very large scales we must find ways to inventory and detect changes in broad forested areas in a relatively short period of time. Landscape and regional management require spatially explicit inventory information. There is mounting evidence that the size, shape, and orientation of landscape elements matters. Remote sensing and geographic information systems (GIS) tools are particularly well suited to address these problems. For example, satellite imagery has been used to estimate tropical deforestation rates, examine changes in the spatial characteristics of forested landscapes, and provide data for estimating regional carbon budgets.

In this thesis project I examined the utility of satellite imagery for addressing large scale forest management questions in the interior of British Columbia. Chapter two describes a project that used LANDSAT satellite imagery from 1975 and 1992 to map forest cover and delineate disturbances over an area of 4.2 million ha in the Southern Interior of B.C. I assessed the accuracy of the map, and compared the estimate of disturbance with that from B.C. Ministry of Forests inventory data for the area from the same time period. I then used the land cover map to examine the changes in landscape pattern during 1975 - 1992 and compared the results with those from other temperate forested areas.

The third thesis chapter examines the carbon budget for the Engelmann-spruce subalpine fir (ESSF) forests, which cover approximately one-third of the 4.2 million ha study area, and occur at the highest elevations. As more accessible forest types have been logged, an increasing amount of the allowable cut in B.C. is coming from ESSF forests. It is important to determine whether these forests are a net source or sink for atmospheric carbon, as they are currently managed. I compared carbon budget estimates based on the satellite map from chapter one, with estimates based on B.C. Ministry of Forests inventory data. Others have estimated carbon budgets from either inventory or satellite data, but there has been no comparison of the two techniques for the same area.
I also compared the results with other regional and national carbon budget estimates. Finally, I projected the inventory data forward to the year 2015 at the current harvesting rate to estimate the future carbon flux from these forests.

Introduction

The detection and mapping of large scale changes to forested landscapes is increasingly important in ecology and management. Forest harvesting can fundamentally alter landscape patterns (Franklin and Forman 1987, Ripple et al. 1991, Skole and Tucker 1993, Spies et al. 1994, Wallin et al. 1994, in press) with potential impacts on biological diversity (Harris 1984, Rosenberg and Raphael 1986, Lehmkuhl et al. 1991). Forest clearing and management play an important role in the global carbon budget, and large scale changes in forest cover have resulted in releases of C to the atmosphere (Houghton et al. 1983, Harmon et al. 1990, Dale et al. 1991, Dixon et al. 1994, Cohen et al. in press). Carbon sequestration by temperate forests has also been postulated as part of a missing sink for 1.8 Pg carbon unaccounted for in current global carbon budget estimates (Tans et al. 1990).

Recent advances in satellite remote sensing and GIS technologies have made the detection of broad scale landscape change much easier, and such techniques have been used increasingly in forested landscapes (Iverson et al. 1989). Satellite imagery has been used to document long-term deforestation rates in the tropics (Nelson et al. 1987, Sader and Joyce 1988, Green and Sussman 1990), and more recently to document the pattern of landscape change by measuring landscape characteristics such as total edge, or the size and arrangement of patches in tropical (Skole and Tucker 1993) and temperate forests (Hall et al. 1991, Ripple et al. 1991, Spies et al. 1994, Turner et al. in press, Zheng et al. in press).

Public forest lands in British Columbia cover approximately 60 % of the province, and their management is extremely important both ecologically and economically. As in the United States, the management of these lands has become increasingly controversial, and many have argued for a shift to ecosystem or landscape
management (Johnson and Agee 1988, Salwasser 1988, Galindo-Leal and Bunnell 1995). The B.C. government has responded with measures such as the new Forest Practices Code (Government of B.C. 1994) and a comprehensive Protected Areas Strategy (Lewis et al. 1994). Remote sensing can be of great use for planning at landscape scales, and in the Pacific Northwest region of the U.S. remote sensing techniques have been used extensively to map remaining old-growth stands in the Oregon Cascades (Morrison et al. 1991, Congalton et al. 1993, Cohen et al. 1995), document levels of cutting (Cohen et al. submitted), quantify changes in landscape pattern imposed by forest management (Ripple et al. 1991, Spies et al. 1994), and estimate net carbon flux from forest lands (Cohen et al. in press). Remaining old-growth reserves in coastal British Columbia have been mapped by photo-interpretation of true color prints of satellite images (MacKinnon and Eng, 1995), but little has been done in the interior forests that cover a large portion of the province. Therefore, this project was undertaken to use satellite imagery for: (1) mapping forest lands in the interior of British Columbia, (2) detecting major changes in forest cover over a long time period, (3) documenting the resulting changes in landscape pattern, and (4) comparing the results with data from other forests of the region and world.

Methods

Study Area

This study examined an area of approximately 4.2 million ha located in south central British Columbia ranging from Barriere in the south to McBride in the north (Figure 2.1). Most of the area is Crown Land and is managed by the Kamloops, Cariboo and Prince George Forest Regions of the B.C. Ministry of Forests. Also included are all of Wells Gray and parts of Bowron Lakes and Mt. Robson Provincial Parks. The landscape is quite varied and encompasses eight zones of the biogeoclimatic ecosystem classification (BEC) (Meidinger and Pojar 1993) ranging from dry forests of the Ponderosa Pine (PP) zone in the south to the high elevation forests of the Engelmann Spruce Subalpine-fir (ESSF) zone and alpine tundra (AT) in the northern half of the area.
Average annual precipitation varies from as little as 31 cm in the PP zone to as much as 200 cm in some portions of the ESSF zone. Mean annual temperatures range from 8 °C in the south to -1 °C in portions of the AT zone.

Figure 2.1. Location of study area in British Columbia.
Table 2.1. Biogeoclimatic (BEC) zones found in study area.

<table>
<thead>
<tr>
<th>BEC Zone</th>
<th>Leading Species</th>
<th>% of Study Area</th>
</tr>
</thead>
<tbody>
<tr>
<td>ESSFdry</td>
<td>Engelmann spruce, subalpine fir</td>
<td>3</td>
</tr>
<tr>
<td>ESSFwet</td>
<td>Engelmann spruce, subalpine fir</td>
<td>28</td>
</tr>
<tr>
<td>ICH</td>
<td>western red cedar, western hemlock, white spruce, Engelmann spruce,</td>
<td>21</td>
</tr>
<tr>
<td></td>
<td>subalpine fir, western larch, Douglas-fir, western white pine, black</td>
<td></td>
</tr>
<tr>
<td></td>
<td>cottonwood, trembling aspen, paper birch</td>
<td></td>
</tr>
<tr>
<td>IDF</td>
<td>Douglas-fir, lodgepole pine, ponderosa pine</td>
<td>11</td>
</tr>
<tr>
<td>MS</td>
<td>hybrid white spruce, lodgepole pine, subalpine fir</td>
<td>3</td>
</tr>
<tr>
<td>PP</td>
<td>ponderosa pine, Douglas-fir</td>
<td>&lt;1</td>
</tr>
<tr>
<td>SBPS</td>
<td>lodgepole pine, white spruce, trembling aspen</td>
<td>5</td>
</tr>
<tr>
<td>SBS</td>
<td>hybrid white spruce, lodgepole pine, trembling aspen, paper birch,</td>
<td>8</td>
</tr>
<tr>
<td></td>
<td>Douglas-fir</td>
<td></td>
</tr>
<tr>
<td>AT</td>
<td>alpine tundra - no trees except krummholz forms</td>
<td>21</td>
</tr>
</tbody>
</table>

Image Processing

Four satellite images were used in the project. The TM (LANDSAT Thematic Mapper) images were acquired on June 23, 1992 (WRS 46/23 and 46/24) and the two MSS (LANDSAT Multispectral Scanner) images were acquired in 1975 on July 5 (WRS 50/23) and July 23 (WRS 50/24). TM image 46/24 had been precision geocoded (25m cell size) and clipped before we received it. We georeferenced both TM scenes to B.C. Ministry of Forests 1:20000 forest cover maps (UTM NAD 27), resampled TM image 46/23 to a 25m cell size using nearest neighbor rules, and joined the two scenes. The two MSS images were georectified to the 1992 TM images. Because the MSS images were from different dates, we chose a series of very bright and very dark targets in the overlap region of the images and determined that the two scenes were spectrally very similar and that no radiometric correction (e.g., Hall et al. 1991) was necessary before joining the images. All georeferencing was done using third-order transformations with a root mean square error of less than 1 pixel. The TM images were transformed into the TM Tasseled Cap brightness, greenness, and wetness axes (Crist et al. 1986), and the MSS images were transformed into the brightness and greenness axes of the MSS Tasseled Cap (Kauth and Thomas 1976). Previous work has shown that the Tasseled Cap transformation facilitates accurate mapping of forest cover (Cohen et al. 1995) and forest disturbances (Cohen et al. in press, in review) in the Pacific Northwest.
We used the unsupervised classification algorithm ISODATA (ERDAS 1994) with the Tasseled Cap TM images to separate pixels containing vegetation from non-vegetated pixels. Cover classes associated with spectral classes were assessed using a combination of extensive ground knowledge, air photos, and forest cover maps. After several iterations we were able to stratify the non-vegetated pixels into cover classes containing water, clouds, snow and ice, and an open class representing areas of bare soil and rock. Assuming the spectral signatures of all remaining pixels were dominated by vegetation, we performed a supervised classification of these pixels using quadratic discriminant functions (SAS 1994) assuming equal prior probability of class membership. Discriminant function analysis is a multivariate technique commonly used in remote sensing for classification. In this method the mean digital values of subsets of pixels of known classes are extracted from the image and linear or quadratic functions are fit to maximize divisions between classes (Tom and Miller 1984, Lark 1994, Ghitter et al. 1995). Using air photos and forest cover maps we digitized 268 training polygons on the TM Tasseled Cap image. Training polygons included closed conifer, deciduous, and mixed forest stands of various ages. We also selected training polygons representing three types of open conifer stands. These included alpine stands, dry open stands of the Ponderosa Pine and Interior Douglas-fir zones (Meidinger and Pojar 1993), and selectively logged stands containing only a remnant of the overstory. The remaining training polygons represented low deciduous vegetation (avalanche tracks and recent brush-covered clearcuts), alpine meadows, and grasslands. For each training polygon we extracted the mean values for brightness, greenness, wetness, and cosine of the solar incidence angle. In addition, for each of the forest stands we entered data on the species mix, disturbance date (if any), age class, and crown closure taken from the forest cover maps.

Our goal was to produce a classification scheme that would prove useful for forest ecology applications such as landscape modeling and carbon budget estimation. At a minimum we wanted to separate the forested areas into pure conifer stands, pure hardwoods, and mixed stands containing both hardwoods and conifers. If possible, we hoped to make further divisions based on stand age or crown cover. We tried a number of methods of grouping the training polygon data based on age class and crown cover
class. At each attempt we checked the classification accuracy using the crossvalidation option in SAS (SAS Institute 1994). Finding it impossible to discriminate reliably between grasslands and alpine meadows, or between the three types of open conifer stands, we combined these classes into grass/meadow and semi-open conifer classes. It was also impossible to reliably separate mixed and deciduous stands by age or cover class. We were able to classify closed conifer stands into three groups based on age class. The age of division between the young and medium age class stands varied depending on the predominant species. For stands dominated by western hemlock, cedar, spruce or balsam-fir, the break between young and medium aged stands occurred at age 60 (age class 4 on the forest cover maps). For all other stands, dominated by lodgepole pine or Douglas-fir the age break came at 40 years (age class 3). This makes ecological sense in that our medium age class represents stands that have closed completely; overstory trees thus dominate the site. Stands generally close faster in the zones of the study area that are dominated by lodgepole pine and Douglas-fir. The best break between medium and old conifer stands was at age 140 in all test stands. This is at least partially an artifact of the age classification scheme used on B.C. Ministry of Forests cover maps. Age classes 1 - 7 use 20-year intervals, but age class 8 stretches from 141 - 250 years. Apparently most of the stands in the study area take on older stand characteristics sometime in this age range regardless of dominant species. In a GIS modeling operation we applied the final discriminant functions to all of the unclassified pixels containing vegetation to produce a classified vegetation map.

To identify major areas of vegetation change between 1975 and 1992, we produced a combined 5-band image containing the Tasseled Cap axes from the 1975 MSS and 1992 TM images. The classification of combined images has been shown to be as accurate as image-differencing for change detection in Northwest conifer forests (Cohen and Fiorella (submitted)). We used the ISODATA algorithm to perform an unsupervised classification on the combined 5-band image. Output clusters were interpreted by examining the Tasseled Cap images from 1975 and 1992. Clusters were assigned to three classes: disturbed, regrowth, and no change. We were conservative in assigning clusters to all classes. Clusters containing more than one output class were reclassified until we had maximized class differences. The disturbed class consisted of
many large patches that appeared to match the obvious cut block boundaries. However, areas in the regrowth class often consisted of many small patches interspersed with areas of no change inside older cut blocks. Since it would have been difficult to use the regrowth information in any landscape pattern analysis, we decided to combine the regrowth and no change classes because we were most interested in analyzing the pattern of cutting. Pixels from the disturbed class were merged with the 1992 cover map in a GIS overlay operation to produce a 1992 cover map that included disturbances.

We used the 1992 TM-based map in conjunction with the 1975 MSS Tasseled Cap image to make a cover map for 1975. We masked all pixels classified as unchanged from the previous analysis and used them as a 1975 base map. We then performed an unsupervised classification of the 1975 MSS Tasseled Cap image on only the pixels classified as disturbed from the change image. It was impossible to use a supervised approach because we lacked accurate 1975 ground data. Due to the poorer spatial and spectral resolution of MSS imagery, and the lack of good ground data it was difficult to separate deciduous from mixed stands, or to make any age divisions in the closed conifer class. Therefore, we used a combined deciduous and mixed forest class and a combined mature and old closed conifer class in our 1975 map.

To examine changes in land cover and landscape pattern in actively managed areas separately from those in parks we digitized the boundaries of the three major parks in the area and overlayed them on the 1975 and 1992 maps. We also obtained a digital copy of the B.C. Biogeoclimatic Zone map from Research Branch, B.C. Ministry of Forests, registered it to the 1992 TM imagery, and used it as an overlay to examine landscape pattern and changes by biogeoclimatic zone. As a final step we used a rule-based merging algorithm to smooth the final 1975 and 1992 maps to a minimum mapping unit of 2 ha (Ma 1995). The same algorithm was used to merge large disturbed areas to a 10 ha minimum mapping unit when possible if they straddled two BEC zones.
Accuracy Assessment

Accuracy of the 1992 map for all classes except the disturbed areas was assessed by selecting a series of 278 random points on the map using the classification accuracy tool in the ERDAS Imagine software (ERDAS, 1995). Each point was identified on forest cover maps or air photos and its true cover type determined. Accuracy of the disturbed class was assessed in two ways. First, we selected a series of 80 random points on the disturbance image, allocating 40 points each to the disturbed and no change classes. Each point was examined on the original 1992 and 1975 Tasseled Cap images to see if any change had occurred.

We used the B.C. Ministry of Forests 1995 Provincial Forest Inventory for a second, independent assessment of the accuracy of detecting disturbance. The inventory is organized by 1:20000 forest cover map sheets. We used SAS software to select all map sheets that covered the study area, and all forest cover polygons that had a logging date, or wildfire burn date, between 1975 and 1992, and sorted the result by BEC zone. This produced the maximum estimate of disturbed area from the inventory. However, some inventory polygons had logging dates that were later than the stand establishment date. This occurs if the stand was selectively logged and the understory was considered established before the final overstory harvest. We removed all these stands from our original estimate to produce a low estimate of the area disturbed. The true inventory estimate should lie between our low and high estimates as stands with only slight overstory removal would probably appear as unchanged in our classification if the understory were still dominated by tree canopies.

It was impossible to assess the accuracy of the 1975 land cover map because we lacked ground data for 1975. However, a large portion of the 1975 map is based on the 1992 map, and we did assess the accuracy of that map as well as our ability to delineate major vegetation disturbances. Separation of conifer dominated stands from hardwood or mixed stands was not difficult in the 1975 imagery, the difficulty was in further subdividing these classes. We used the 1975 map solely for the landscape pattern analysis, and there we only assessed the conifer and disturbed patches.
Landscape Pattern Analysis

We obtained a digital copy of the B.C. Biogeoclimatic Zone map from Research Branch, B.C. Ministry of Forests, registered it to the 1992 TM imagery, and used it as an overlay to examine landscape pattern and changes by biogeoclimatic zone. We examined the land cover pattern by calculating indices that describe the overall landscape pattern and that of each cover class using the FRAGSTATS computer program (McGarigal and Marks 1995). The cover class statistics calculated included mean patch size, largest patch size, and percentage of interior area based on a 100 m edge width. Three additional cover class indices relating to patch shape were calculated. The first index was area-weighted mean patch fractal dimension (AWMPFD). This is calculated as the average fractal dimension of patches of a cover class, weighted by patch area so that larger patches are given more weight:

\[
AWMPFD = \sum_{j=1}^{n} \left( \frac{2 \ln(p_{ij})}{\ln(a_{ij})} \right) \left( \frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}} \right)
\]

where \( a_{ij} = \text{area (m}^2\text{) of class i, patch j, and } p_{ij} = \text{perimeter (m) of class i, patch j.} \)

The fractal dimension (and thus AWMPFD) ranges from 1 to 2, approaching 1 for shapes with simple perimeters such as squares or circles, and increasing towards 2 for shapes with highly convoluted perimeters (Mandelbrot 1977, 1982). The other two shape indices are based on the shape index proposed by Patton (1975) which measures the complexity of a patch shape compared to a standard shape, usually a circle or square. In the raster version of FRAGSTATS the shape index for each patch is evaluated based on a square standard. The second index, area-weighted mean shape index (AWMSI), is the average shape index of patches of a cover class weighted by patch area:

\[
AWMSI = \sum_{j=1}^{n} \left( \frac{0.25 p_{ij}}{\sqrt{a_{ij}}} \right) \left( \frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}} \right)
\]
The AWMSI = 1 when all patches of the corresponding class are square and increases as the patch shapes become more irregular. The third index, landscape shape index (LSI), applies the shape index concept to the landscape as a whole for each class, treating all of the class area and edges in the landscape as one large patch:

\[
LSI = \frac{0.25 \sum_{k=1}^{m} e_{ik}}{\sqrt{A}}
\]

where A = total landscape area, e = total length of edge in landscape between classes i and k, and m = the number of classes. The LSI = 1 when the landscape consists of a single square patch of the corresponding class, and increases as landscape shape becomes more irregular or the length of edge of that class type increases.

We also calculated two indices of overall landscape pattern using FRAGSTATS. The contagion index (C) (O’Neill et al. 1988, Li and Reynolds 1993) measures the degree to which landscape elements are aggregated or clumped. Higher contagion values are characteristic of landscapes with a few large continuous patches and lower values of contagion indicate a more fragmented landscape of many smaller patches. Contagion is calculated as:

\[
C = 1 + \frac{\sum_{i=1}^{m} \sum_{k=1}^{m} \left( P_i \frac{g_{ik}}{ \sum_{k=1}^{m} g_{ik} } \right) \ln(P_i) \left( \frac{g_{ik}}{ \sum_{k=1}^{m} g_{ik} } \right)}{2 \ln(m)} (100)
\]

where \( P_i \) = the proportion of the landscape in class i, \( g_{ik} \) = the number of adjacencies between pixels of classes i and k, and m = the number of classes. Contagion ranges from 0-100 and is expressed as the percentage of the maximum possible contagion, given the number of cover classes.

The interspersion and juxtaposition index (IJI) measures the interspersion of cover classes by patches. Unlike contagion, which is calculated on a pixel by pixel basis, IJI only considers patch edges. Higher values of IJI indicate landscapes in which cover
class types are well interspersed, or equally adjacent to each other. Lower values of IJI characterize landscapes with poorly interspersed patch types. IJI is calculated as:

$$\text{IJI} = \frac{\sum_{i=1}^{m} \sum_{k=1}^{m} \left( \frac{e_{ik}}{E} \right) \ln \left( \frac{e_{ik}}{E} \right)}{\ln(0.5[m(m-1)])}$$

where $E$ = the total length of edge in the landscape, and $e_{ik}$ and $m$ are as defined in equations [3] and [4]. Like contagion, the IJI ranges from 0 - 100 and is expressed as the percentage of maximum possible interspersion for the given number of cover classes.

To compare the landscape statistics for the 1975 and 1992 maps we were forced to compute the statistics using the simpler cover class scheme from the 1975 map. We also computed statistics for the 1992 map using the more detailed 1992 class structure to examine the effects of combining the mature and old conifer classes.

Due to the enormous size of the study area, the FRAGSTATS program was incapable of computing statistics on the whole area at 25m resolution. We selected a smaller area of the map and resampled it to a 50m cell size using nearest neighbor rules. To test the effect of resampling, the subset area was analyzed at both 25m and 50m resolutions using FRAGSTATS and the results showed no appreciable difference in any of the calculated statistics. We then resampled the entire 1992 and 1975 maps to a 50m resolution with nearest neighbor rules and were able to use FRAGSTATS to calculate landscape statistics on the entire area.

Results

Accuracy Assessment

Our overall classification accuracy was 79 percent for the 1992 map (Table 2.2). The non-vegetated classes were accurately identified with only minor confusion between snow/ice, clouds, and open pixels. These errors all occurred at high-elevation areas of the map. Among the vegetated classes, approximately 20 percent of the mature and old
conifer pixels were confused with each other, but rarely with other classes. The young conifer class was least accurate, with over half the pixels misclassified as one of the other closed tree classes. In addition, we were able to detect major vegetation disturbances quite accurately. Of the 80 pixels examined, only one was misclassified, leading to an overall classification accuracy of 99 percent. The lone error was due to slight mis-registration of an avalanche track between the 1975 and 1992 images. Our image-based estimate of the total area disturbed in the 17-year period fell approximately midway between the two estimates from the inventory data (Table 2.3). Comparing estimates of disturbance by BEC zone was problematic for several reasons. First, in the inventory any cut block straddling two or more BEC zones was probably assigned to the zone with the largest area. On the satellite based map, disturbed areas were merged to a 10 ha minimum mapping unit when possible so that larger cut blocks could be split between BEC zones only if the pieces were larger than 10 ha. Secondly, there was some error induced by slight image mis-registration. Most of this was removed in image smoothing to 2 ha, but linear features such as avalanche tracks and cut boundaries would have been most subject to this sort of error. Despite these limitations, image-based and inventory estimates of disturbed area by BEC zone were in general agreement except that the imagery-based estimates of the total disturbed area in the ESSF and IDF zones were higher and lower, respectively than those from the inventory. Differences were due in part to a number of smaller disturbances such as new avalanche tracks and small fires that did not appear as part of the forest inventory as well as the inevitable image registration and zone boundary problems.
Table 2.2. Error matrix for 1992 TM land cover map. Kappa statistic was 77 percent.

<table>
<thead>
<tr>
<th>Observed</th>
<th>Water</th>
<th>Snow/ice</th>
<th>Clouds</th>
<th>Open</th>
<th>Old conifer</th>
<th>Mature conifer</th>
<th>Young conifer</th>
<th>Deciduous</th>
<th>Closed</th>
<th>Semi-open conifer</th>
<th>Low decid. veg.</th>
<th>Grass/ meadow</th>
<th>Percentage correct</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>10</td>
<td>12</td>
<td>2</td>
<td>1</td>
<td>1</td>
<td>9</td>
<td>41</td>
<td>15</td>
<td>5</td>
<td>4</td>
<td>2</td>
<td>14</td>
<td>100</td>
</tr>
<tr>
<td>Snow/ice</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>86</td>
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<td>Clouds</td>
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<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>100</td>
</tr>
<tr>
<td>Open</td>
<td></td>
<td>1</td>
<td>1</td>
<td>14</td>
<td>31</td>
<td>9</td>
<td>1</td>
<td>14</td>
<td>15</td>
<td>8</td>
<td>2</td>
<td></td>
<td>88</td>
</tr>
<tr>
<td>Old conifer</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>76</td>
</tr>
<tr>
<td>Mature conifer</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td>80</td>
</tr>
<tr>
<td>Young conifer</td>
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<td></td>
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<td></td>
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<td></td>
<td></td>
<td></td>
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<td>44</td>
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<tr>
<td>Deciduous</td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>82</td>
</tr>
<tr>
<td>Mixed</td>
<td></td>
<td>2</td>
<td>1</td>
<td>14</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>82</td>
</tr>
<tr>
<td>Semi-open conifer</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>72</td>
</tr>
<tr>
<td>Low deciduous veg.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>100</td>
</tr>
<tr>
<td>Grass / meadow</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>89</td>
</tr>
<tr>
<td>Percentage correct</td>
<td>100</td>
<td>92</td>
<td>88</td>
<td>100</td>
<td>66</td>
<td>71</td>
<td>88</td>
<td>78</td>
<td>48</td>
<td>100</td>
<td>100</td>
<td>100</td>
<td>100</td>
</tr>
</tbody>
</table>

Percentage correct 100 92 88 100 66 71 88 78 48 100 100 100 79
Table 2.3. Comparison of estimates of disturbed areas by BEC zone from imagery and forest inventory.

<table>
<thead>
<tr>
<th>BEC Zone</th>
<th>Total Area (ha)</th>
<th>Imagery</th>
<th>Inventory Low</th>
<th>Inventory High</th>
</tr>
</thead>
<tbody>
<tr>
<td>ESSF</td>
<td>1286467</td>
<td>8030</td>
<td>57408</td>
<td>73575</td>
</tr>
<tr>
<td>ICH</td>
<td>866577</td>
<td>49699</td>
<td>49521</td>
<td>55767</td>
</tr>
<tr>
<td>IDF</td>
<td>467859</td>
<td>13827</td>
<td>17470</td>
<td>36534</td>
</tr>
<tr>
<td>MS</td>
<td>113784</td>
<td>11099</td>
<td>11484</td>
<td>12754</td>
</tr>
<tr>
<td>PP</td>
<td>5971</td>
<td>201</td>
<td>21</td>
<td>96</td>
</tr>
<tr>
<td>SBPS</td>
<td>213972</td>
<td>12957</td>
<td>13258</td>
<td>16074</td>
</tr>
<tr>
<td>SBS</td>
<td>342386</td>
<td>23765</td>
<td>19112</td>
<td>21882</td>
</tr>
<tr>
<td>AT</td>
<td>858664</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Totals</td>
<td>4155679</td>
<td>192378</td>
<td>168273</td>
<td>216683</td>
</tr>
</tbody>
</table>

1 See Table 2.1 for definition of BEC zones.

Land Cover

Forests covered 64.3% of the total study area in 1992, and mature and older conifer forests accounted for 41.9% of the total area (Table 2.4). Areas disturbed between 1975 and 1992 covered 4.6% of the total area. Examination of the cover map (Figure 2.2), and the forest inventory comparison (Table 2.3) indicate the great majority of these disturbances are from cutting. Water, snow and ice covered 12.8% of the total area, with the majority of this cover type in the northern half. Major vegetation patterns are evident, with the majority of old growth conifer areas located in the northern half of the image, and the southern half dominated by mature conifer stands. This is reasonable since the northern portion of the study area is more mountainous, receives considerably more precipitation, and fires are less frequent. The semi-open conifer stands are found bordering snow and open areas at high elevation, and in the drier areas of the IDF and SBPS zones in the southwest portion of the area. Deciduous and mixed stands are generally confined to major river valleys and lower elevations along the major lakes in the southern half of the area.
Major provincial parks cover 19.2% of the study area (Table 2.5) and are generally in high elevation areas as evidenced by the higher percentage of snow and open areas (predominantly alpine) than in the landscape outside park boundaries. The total forested area in parks is 51.7% compared with 67.2% outside parks. Almost all disturbances occurred outside of park boundaries. If we assume that only forested pixels were harvested, the disturbed area ranges from 6.6 - 9.8 percent of the total forest in the landscape depending on the definition of forest (Table 2.6). More importantly, if only the area outside of the parks is considered, disturbances accounted for 8.4 - 11.4 percent of the forested area. In this study area the great majority of timber harvesting occurs in coniferous forests, and it is reasonable to assume that only mature and older stands are harvested. A GIS overlay of the 1992 and 1975 maps showed 95 percent of the disturbed pixels were classified as mature + old conifer forest in 1975. Therefore, the estimate of 11.4 percent of the forested area disturbed in the 17-year period is reasonable.

Table 2.4. Results of the 1992 classification.

<table>
<thead>
<tr>
<th>Class</th>
<th>Number of hectares</th>
<th>Percentage of total</th>
<th>Percentage of forest</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water</td>
<td>142796</td>
<td>3.4</td>
<td></td>
</tr>
<tr>
<td>Clouds</td>
<td>167508</td>
<td>3.9</td>
<td></td>
</tr>
<tr>
<td>Snow</td>
<td>397081</td>
<td>9.4</td>
<td></td>
</tr>
<tr>
<td>Open</td>
<td>460153</td>
<td>10.8</td>
<td></td>
</tr>
<tr>
<td>Low deciduous vegetation</td>
<td>71706</td>
<td>1.7</td>
<td></td>
</tr>
<tr>
<td>Grass and meadows</td>
<td>83640</td>
<td>2.0</td>
<td></td>
</tr>
<tr>
<td>Disturbances</td>
<td>193946</td>
<td>4.6</td>
<td></td>
</tr>
<tr>
<td>Semi-open conifer forest</td>
<td>451715</td>
<td>10.6</td>
<td>16.6</td>
</tr>
<tr>
<td>Closed deciduous forest</td>
<td>148634</td>
<td>3.5</td>
<td>5.5</td>
</tr>
<tr>
<td>Closed mixed forest</td>
<td>272005</td>
<td>6.4</td>
<td>10.0</td>
</tr>
<tr>
<td>Closed young conifer</td>
<td>76621</td>
<td>1.8</td>
<td>2.8</td>
</tr>
<tr>
<td>Closed mature conifer</td>
<td>993719</td>
<td>23.4</td>
<td>36.4</td>
</tr>
<tr>
<td>Closed old conifer</td>
<td>783996</td>
<td>18.5</td>
<td>28.8</td>
</tr>
<tr>
<td>Total</td>
<td>4243520</td>
<td>100.0</td>
<td>100.0</td>
</tr>
</tbody>
</table>
Figure 2.2. A June 1992 land cover map derived from classification of TM imagery. The disturbances were mapped using a combination of 1992 TM and 1975 MSS imagery (see text).
Table 2.5. Comparison of land cover in 1992 inside vs. outside parks.

<table>
<thead>
<tr>
<th>Class</th>
<th>Percent of total area</th>
<th>Inside Parks</th>
<th>Outside Parks</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Area (ha)</td>
<td>Area (%)</td>
<td>Area (ha)</td>
</tr>
<tr>
<td>Water</td>
<td>3.3</td>
<td>29161.7</td>
<td>3.6</td>
</tr>
<tr>
<td>Clouds</td>
<td>3.9</td>
<td>64061.4</td>
<td>7.9</td>
</tr>
<tr>
<td>Snow</td>
<td>9.3</td>
<td>135528.7</td>
<td>16.6</td>
</tr>
<tr>
<td>Open</td>
<td>12.6</td>
<td>139625.4</td>
<td>17.1</td>
</tr>
<tr>
<td>Closed old conifer</td>
<td>17.0</td>
<td>188862.6</td>
<td>23.2</td>
</tr>
<tr>
<td>Closed med conifer</td>
<td>23.3</td>
<td>97396.7</td>
<td>11.9</td>
</tr>
<tr>
<td>Closed young conifer</td>
<td>1.8</td>
<td>11820.4</td>
<td>1.5</td>
</tr>
<tr>
<td>Hardwoods</td>
<td>3.5</td>
<td>14331.1</td>
<td>1.8</td>
</tr>
<tr>
<td>Mixed</td>
<td>6.4</td>
<td>32166.4</td>
<td>3.9</td>
</tr>
<tr>
<td>Semi-open conifer</td>
<td>10.6</td>
<td>77437.8</td>
<td>9.5</td>
</tr>
<tr>
<td>Low deciduous vegetation</td>
<td>1.7</td>
<td>12980.6</td>
<td>1.6</td>
</tr>
<tr>
<td>Grass and meadows</td>
<td>2.0</td>
<td>10492.1</td>
<td>1.3</td>
</tr>
<tr>
<td>Disturbed</td>
<td>4.5</td>
<td>1564.6</td>
<td>0.2</td>
</tr>
<tr>
<td>Total</td>
<td>100.0</td>
<td>815429.3</td>
<td>100.0</td>
</tr>
</tbody>
</table>

Percentage of total area in parks: 19.2

Table 2.6. Disturbance as percentage of forested area.

<table>
<thead>
<tr>
<th>Method of Calculation</th>
<th>Inside Parks</th>
<th>Outside Parks</th>
<th>Total Landscape</th>
</tr>
</thead>
<tbody>
<tr>
<td>Disturbed as percentage of total forest</td>
<td>0.4</td>
<td>8.4</td>
<td>6.6</td>
</tr>
<tr>
<td>Disturbed as percentage of total conifer forest</td>
<td>0.4</td>
<td>9.1</td>
<td>7.8</td>
</tr>
<tr>
<td>Disturbed as percentage of closed conifer forest</td>
<td>0.5</td>
<td>11.0</td>
<td>9.5</td>
</tr>
<tr>
<td>Disturbed as percentage of mature + old conifer forest</td>
<td>0.5</td>
<td>11.4</td>
<td>9.8</td>
</tr>
</tbody>
</table>

Changes in Land Cover Between 1975 and 1992 Maps

A direct comparison of land cover in 1975 and 1992 showed the major change in land cover was a decrease of about 217,000 ha in the area covered by mature + old conifers. The majority of this was accounted for by detection of 193,000 ha of disturbances, almost exclusively outside park boundaries (Figure 2.3). There was a slight increase in open area during the period, and a decrease in snow covered area. This effect was greatest inside the parks, which contain a higher percentage of high elevation areas. We were conservative during classification of the disturbed areas, some of the increase in open areas could be disturbances, and some due to mis-registration of the imagery, but most is likely due to variation in snow cover between the two dates. These
differences are of minor importance because we only used the 1975 land cover map to examine changes in the landscape pattern of conifer and disturbed patches.

**Landscape Pattern**

Our analyses showed that the area inside parks changed very little during the 17-year period studied as only 0.19% of the area was disturbed (Table 2.5), therefore landscape change statistics are presented only for the areas outside parks. The average conifer patch size decreased in all but the IDF and PP zones, and the size of the largest patch decreased in all zones (Figure 2.4a, b). The interior area of conifer patches decreased in all but the IDF and PP zones. The disturbed areas were consistently much smaller than conifer patches in all zones, and had a lower percentage of interior area (Figure 2.4c). The mean patch size of disturbed areas was very similar across zones, ranging from 20 - 50 ha, except in the PP zone (9.2 ha). There was little difference in the complexity of conifer patch shapes in 1975 and 1992, but the disturbed areas had much simpler shapes as measured by fractal dimension, and the mean and landscape shape indices (Figure 5a, b, c). Conifer patch shape complexity varied between zones, but the shapes of disturbed areas were similar in all zones.

Separating the old and mature conifer patches affected mean patch size, largest patch size and interior area the most (Figure 2.6a, b, c). Mean patch sizes for both conifer classes remained larger than disturbances in the ESSFwet and ICH zones, but average old growth patches in the SBS were actually smaller than disturbances. The largest patches of old and mature conifer stands were greater than largest disturbed patches in all three zones. The amount of core area was similar for all three patch types in all zones. Both mature and old conifer patches had consistently more complex shapes than the disturbed patches although the differences in the three shape indices were not as pronounced as when the two conifer classes were combined (Figure 2.6d, e, f).

There was very little change in either of the two overall landscape statistics during the 17-year period. Contagion changed from 55.5 in 1975 to 56.0 in 1992, and IJI
increased from 60.8 to 61.7 during the period. Apparently not enough of the entire landscape was disturbed to significantly affect either of these indices.

Figure 2.3. Change in cover classes during the period 1975 - 1992.
Figure 2.4. (a) Mean patch size, (b) area of largest patch, and (c) percentage of interior area based on a 100 m edge for disturbed areas, and conifer areas in 1975 and 1992. Error bars in (a) represent one standard error of the mean.
Figure 2.5. Patch complexity as measured by (a) AWMPFD, (b) AWMSI, and (c) LSI (see text) for disturbed areas, and conifer areas in 1975 and 1992.
Figure 2.6. Comparison of (a) mean patch size, (b) size of largest patch, (c) percentage of interior area based on a 100 m edge, (d) AWMPFD, (e) AWMSI, and (f) LSI for disturbed, old, and mature conifer patches in 1992.
Discussion

Satellite imagery has been used to estimate disturbance rates in many forest ecosystems worldwide. Nelson et al. (1987) estimated deforestation rates of 0.4% year\(^{-1}\) for 1981-1984 in Mato Grosso, Brazil. Annual deforestation rates as high as 7.7% were detected for the period 1977 - 1983 in Costa Rica (Sader and Joyce 1988). In the Brazilian Amazon, deforestation rate calculated from satellite imagery averaged 0.4% year\(^{-1}\) from 1978 - 1988 (Skole and Tucker 1993). On Madagascar, rain forest was lost at an annual rate of 1.5% during 1950 - 1985 (Green and Sussman 1990). However, as Spies et al. (1994) point out, deforestation in many tropical landscapes is the direct result of agricultural expansion. Landscape change in British Columbia is best compared with similar studies in other temperate forests where cutting is mainly for wood production.

The disturbance rate in our study area is lower than most remote sensing based estimates from other temperate forests. In B.C. total forest land outside the parks decreased 8.4% in 17 years (0.49% year\(^{-1}\)). Cohen et al. (submitted) reported 15.5% of the total forest land in a 921,000 ha area of western Oregon was harvested between 1972 and 1991. If we assume that there was no cutting in the 8.6% of their study area that is in wilderness areas, then 16.9% of all the other forest land was cut (0.89% year\(^{-1}\)). In a 259,000 ha subset of the Oregon area studied by Cohen et al. (submitted), Spies et al. (1994) reported disturbance rates of 1.2% year\(^{-1}\) on public, nonwilderness forest land, 3.9% year\(^{-1}\) for private forest land, and 0.2% year\(^{-1}\) for wilderness areas for a 16-year period. Tuner et al. (in press) examined relationships between land ownership and land cover change in two watersheds on the Olympic Peninsula, Washington. They reported average annual disturbance rates of 1.41% on private, and 0.28% on public lands. In an area of northern Minnesota, Hall et al. (1991) reported annual rates of major disturbance in mixed and conifer types of 2.7% and 1.8%, respectively on nonwilderness lands. They reported lower annual disturbance rates in the neighboring wilderness, 1.86% and 0.72% for mixed and conifer forests. Our estimate of annual disturbance rates in parks (0.02%) is considerably lower than those reported for wilderness areas in Oregon and Minnesota. However, Zheng et al. (in press) reported a similar low disturbance rate (0.04% year\(^{-1}\)) for the Changbai Mountain International Biosphere Reserve in northeast...
China during the period 1972-1988. They estimated an average rate of disturbance of 1.12 % year\(^{-1}\) for the adjacent area outside the Changbai Reserve.

Clearly the disturbance rate measured in B.C. is at the low end of the range for a variety of forests throughout the world. However, forest growth rates in this portion of B.C. are also substantially lower than those from forests in most of the studies mentioned. This leads to slower reforestation rates and longer rotations for our study area, especially the large portion that is at higher elevations. For example, in western Oregon, many private landowners are planning for rotations of 55 years and the current planned rotation length on public lands is 80 years (Spies et al. 1994). Rotation ages for both conifer and deciduous forests are probably less than 100 years in Minnesota. In our study area rotation lengths vary by BEC zone and species, but generally are planned to approximate the culmination of mean annual increment (CMAI). We calculated the mean site index for each BEC zone in our study area from the inventory data, and then used a managed stand yield simulator, WinTIPSY (B.C. Ministry of Forests 1995), to estimate CMAI for several of the major conifer species. In the ESSF zone, CMAI for planted stands of white spruce is 120 years, and for lodgepole pine CMAI is 80 years. In the ICH zone, CMAI for Douglas-fir is 140 years. The actual rotation ages will probably be longer, especially for spruce in the ESSFwet zone, because we assumed no regeneration delay when running the WinTIPSY model.

Our results indicate the very early stages of fragmentation of the conifer matrix outside the parks during the 17-year period of record. We detected decreases in the mean and largest conifer patch size and percentage of interior area in most BEC zones. These results are consistent with those reported for other areas where forest fragmentation has occurred. In western Oregon, the percentage of edge of conifer patches increased from 9.5 to 13 with increased harvests from 1972 - 1988, with a corresponding drop in the percentage of interior habitat (Spies et al. 1994). In another study in the Oregon Cascades, Ripple et al. (1991) documented decreases in mean patch size and interior area with harvesting over a 15-year period. They also measured increases in patch shape complexity as measured by patch fractal dimension and a diversity index similar to the shape index we used. In an area bordered by China and Korea, forest harvesting caused an increase in the total area, mean size, edge and shape complexity of non-forest patches.
over a 16-year period (Zheng et al. in press). Given that the landscape was classified into only forest and non-forest patches, the results indicate a decrease in forest patch size, and increases in edge and shape complexity of the forest patches.

That we found little difference in the shape complexity of conifer patches during the study period is indication that the conifer matrix in our study area is still relatively intact. Moreover, we are well below the disturbance thresholds postulated by Franklin and Forman (1987). Other studies have documented major changes in patch shape associated with disturbance in forested landscapes over time or in comparisons of adjacent disturbed and undisturbed landscapes (Ripple et al. 1991, Zheng et al. in press, Mladenoff et al. 1993; Skole and Tucker 1993): The fact that we could detect no change in the two landscape-level indices, III and C, is further evidence that our landscape is only in the early stages of fragmentation. However, it is clear that the cut patches imposed by logging during the study period are much smaller and simpler in shape than the patches making up the conifer matrix (Figures 2.4, 2.5, 2.6). Based on results from other forests worldwide, if the rate and pattern of disturbance in our study area continues major changes in the conifer forest matrix will result. In western Oregon, conifer edge was greatest when 40% of the landscape was cut, and declined with increased cutting (Spies et al. 1994). In our study area 64.5% of total forest was still in mature and old conifer stands and recent disturbances made up less than 5% of the landscape, suggesting that the conifer matrix is still dominant. Percolation theory suggests that patch characteristics change very rapidly near the critical probability, \( p_c \), which is the probability at which the largest patch spans the landscape entirely (\( p_c = 0.5928 \) for very large arrays) (Stauffer 1985, Gardner et al. 1987, Turner et al. 1989). From a landscape perspective, this suggests that when a patch type, in this case conifer forests, covers much more than 60% of a landscape, there will be little change in the structure of the landscape with disturbance. Edge should increase, but patch shape is relatively unchanged. As disturbance increases and coverage of conifer forests decreases below 60% of the landscape, changes in the shape of conifer patches will be rapid, approaching that of the disturbed patches as the total area of conifer patches drops below 50%. This eventually leads to a landscape composed of smaller and simpler patches. For example, in Minnesota a managed landscape had more numerous, smaller and simpler patches
than the adjacent old-growth wilderness area (Mladenoff et al. 1993). Our data indicate that coverage of conifer forests is still well above $p_c$ in the study area due to the relatively low disturbance rate and short disturbance history.

There is also evidence that a similar size and shape of forest harvest unit has been applied uniformly across most of the BEC zones in the study area during the period of record. The size and shape of conifer patches varies broadly between BEC zones, yet the variation in cut block size and shape by zone is much narrower (Figures 2.4, 2.5). There is very little data on the natural disturbance regime in this part of British Columbia, but if we assume that the size and shape of conifer patches are to some extent a legacy of varying disturbance regimes in each zone then it is obvious that the current harvesting pattern is radically different from the native disturbance regime in at least some BEC zones. For example, in a variant of the SBS BEC zone in the Prince George Region, northwest of our study area, during the four 20-year time periods before effective fire suppression $50 - 75\%$ of the total area burned was comprised of wildfires larger than 500 ha (DeLong and Tanner in press). Current harvest practices in the area limit clearcuts to 80 ha. Detailed mapping of nine individual fires showed that islands of undisturbed forest ranging from 1 - 73 ha in size accounted for 3 - 15\% of the total area inside the fire perimeter, whereas cutting removes virtually all trees. The shape index of fire perimeters was also consistently higher for wildfires than for harvest units. In the portion of our study area in the SBS zone, mean conifer patch size was over 4 times greater than mean cut block size and conifer patches had much more complex shapes (Figures 2.4, 2.5).

Management Implications

We have documented a relatively low annual rate of disturbance over a vast area in the interior of British Columbia. Spatial analysis of the cutting pattern showed that the conifer forest matrix is in the very early stages of fragmentation and has not yet reached any of the hypothesized empirical thresholds (Franklin and Forman 1987, Turner et al. 1989). However, a similar type of disturbance pattern is being imposed on most BEC zones. It is important to remember that these results are averages over a 4.2 million
ha area. Examination of the 1992 land cover map (Figure 2.2) clearly shows that some watersheds have been severely impacted by cutting while others have remained untouched. Given that some level of timber harvest will continue on a large portion of the landbase outside of the provincial parks, it is inevitable that the landscape pattern will be changed. We have the ability to control this change, but need to determine the sort of pattern that should be created.

Concerns over habitat fragmentation (Harris, 1984, Franklin and Forman 1987) and protection of endangered species coupled with the continued need for resource extraction have led to the development of the concept of ecosystem management (Johnson and Agee 1988, FEMAT 1993, Grumbine 1994, Galindo-Leal and Bunnell 1995). Current thinking is that the spatial and temporal scale of management should fall within the range of natural variability in disturbance regimes (Hunter 1993, Swanson et al. 1993), and be based on the spatial requirements of organisms (Galindo-Leal and Bunnell 1995) although these two objectives may not be compatible in all cases. This will require management at various scales including stands, landscapes, and entire regions (Franklin 1993, Pojar et al. 1994).

Ecosystem management at the landscape scale, will require knowledge of natural disturbance regimes and their interaction with managed landscapes. There has been a good deal of theoretical work on developing indices of landscape pattern (O’Neill et al. 1988, McGarigal and Marks 1995, Baskent and Jordan 1995), and a number of studies have quantified landscape structure (e.g., Spies et al. 1994, Ripple et al. 1991, Turner et al. in press, Baskent and Jordan 1995, this study), but there has been relatively little empirical examination of the relationship between landscape structure and disturbance (Baker 1992, 1993), although theoretical work exists (Franklin and Forman 1987, Turner et al. 1989, Li et al. 1993, Wallin et al. 1994). Locally, we have some information about individual disturbance agents. There have been several studies of fire history in the region (DeLong and Tanner in press, Tande 1979, Masters 1990, Johnson et al. 1990), and there is organism-specific and stand-level information about root disease pathogens (Morrison et al. 1991) and individual insects (e.g., Wood and Van Sickle 1992).

However, as elsewhere, there is very little information about the interaction of natural disturbance agents and forest management to create landscape pattern. The development
of spatially explicit landscape models that include both natural disturbance agents and harvesting will be critical to predicting the structure of future landscapes under different management scenarios (e.g., Mladenoff et al. in press).

Ecosystem management is also based on the hypothesis that landscape structure plays an important role in regulating wildlife populations. However, much support for this idea has come from studies of responses of vertebrate populations to fragmentation in forest remnants where forest fragmentation was caused by agricultural or urban expansion (Saunders et al. 1991). It is unclear how results apply to fragmentation of forested landscapes caused by intensive forest management (McGarigal and McComb 1995). There have been few experimental studies relating population responses of wildlife species to landscape structure in commercial forest areas. In California, bird and amphibian richness increased with increasing landscape fragmentation (Rosenberg and Raphael 1986). Similar results for bird species were reported by Lehmkuhl et al. (1991) for the Washington Cascades, but mammal richness and abundance showed little relationship to landscape pattern. In the most extensive investigation to date, McGarigal and McComb (1995) examined the relationship between the relative abundance of breeding birds and landscape pattern in 30 landscapes (250 - 300 ha) in the Oregon Coast Range. Landscape structure typically explained less than 50% of the variation in species' abundance among the landscapes. As in other studies, individual bird species were generally more abundant in the more fragmented habitats.

Given our incomplete knowledge of the interaction of natural disturbance regimes and forest management to determine landscape pattern, and the resultant response of wildlife populations, it seems prudent to manage within the spatial and temporal variability of the natural disturbance regime (Hunter 1993, Swanson et al. 1993). It is certainly conceivable that alternative landscape patterns could maintain biodiversity and maximize the flow of wood products, but in the absence of empirical data and reliable predictive tools, it seems best to err on the side of conservatism and assume that species are best adapted to the range of landscape patterns they have evolved with. The pattern in the current undisturbed forest has already been influenced by fire suppression during this century. Clearly, by applying a single disturbance pattern across most BEC zones we risk creating a landscape pattern outside the historical range in at
least some zones. For example, in two watersheds in the Oregon Cascades there is evidence that fire suppression and the pattern of harvesting over 50 years created a landscape pattern outside the range of conditions experienced in the previous 500 years (Wallin et al. in press).

There is a unique opportunity in the interior of British Columbia to practice ecosystem management on these lands for several reasons. First and foremost, as this study indicates, the conifer matrix is still relatively intact, so we are less constrained by a legacy of cutting than in other parts of the world. Additionally, the large areas of relatively undisturbed forest support functional populations of large carnivores and herbivores, and the bulk of forest land (97%) is under a single public ownership making planning and the implementation of policy much easier than in multi-ownership landscapes (Galindo-Leal and Bunnell 1995). We also have a well developed ecosystem classification system that is currently in use for regional planning (Lewis et al. 1994, Pojar et al. 1994), including a strategy for increasing the protected areas of the province from the current 6.5% to 12% of the land base by the year 2000. Implementing a system of protected areas is critical, but managing the remaining landscape matrix is as important due to its role in maintaining diversity and controlling landscape connectivity (Franklin 1993). Simulations by Wallin et al. (1994) have indicated that an established landscape pattern created by dispersed cutting will persist on the landscape and be difficult to erase, even with a substantial reduction in the harvest rate. Clearly we must carefully plan the pattern of our disturbances. Altering the spatial pattern over a large landscape appears to take no more than one rotation. Erasing that pattern will take much longer.

Literature Cited


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Introduction

It is widely believed that increasing atmospheric concentrations of CO$_2$ and other radiatively-active trace gases may lead to global climate change (IPCC 1996). The most recent global estimates (IPCC 1994) show atmospheric CO$_2$ content is currently increasing at approximately 3.3 ± 0.2 Pg of carbon each year. The two major sources of atmospheric CO$_2$ are combustion of fossil fuels and changes in land use, which release 5.5 ± 0.5 Pg and 1.6 ± 1.0 Pg of carbon per year, respectively. Oceans are estimated to be a net sink for 2.0 ± 0.8 Pg C year$^{-1}$, leaving approximately 1.3 ± 1.5 Pg carbon accumulating in the atmosphere annually that can not be accounted for. Recent work by Dixon et al. (1994) has reduced the landuse emissions estimate to 0.9 ± 0.4 Pg C year$^{-1}$ reducing the imbalance in the global carbon budget to 1.1 ± 1.0 Pg C year$^{-1}$. However, these results do not agree with recent estimates based on atmospheric CO$_2$ and $^{13}$CO$_2$ indicating that the terrestrial biosphere is currently a net carbon sink, perhaps sequestering as much as 2.6 Pg C during 1992-93. The same estimates indicate the tropics are a net carbon source, implying the mid- to high latitude terrestrial biosphere is a strong carbon sink (Melillo et al. 1996).

Given the inability to balance the global carbon budget there has been a large effort over recent years to more accurately estimate the carbon flux from terrestrial ecosystems. Global estimates (e.g., Houghton et al. 1983, Dixon et al. 1994) rely heavily on national (e.g., Kauppi et al. 1992, Kurz et al. 1992, Turner et al. 1995) and regional (e.g., Harmon et al. 1990) estimates. Unfortunately, the large scale national estimates are often based on very coarse scale inventory data that are rarely spatially explicit. Improved regional estimates are needed to narrow the uncertainty in the global terrestrial carbon budget. To overcome the shortcomings of inventory data or to provide estimates
where inventory data are lacking, one possible solution is to use remotely sensed data for regional carbon budgets (Cohen et al. in press). To test this methodology it would be useful to compare carbon budget estimates from remotely sensed data to those derived from a fine-scale inventory in the same area. In B.C. we are fortunate to have good spatially explicit forest inventory data readily available for such a comparison. We have used this forest inventory data to demonstrate that we can accurately map forest cover and disturbances in the Southern Interior of B.C. using LANDSAT imagery (Chapter 2).

We focus on the regional carbon budget for Engelmann spruce (*Picea engelmannii*) - Subalpine fir (*Abies lasiocarpa*) forests in British Columbia. These forests, placed in the ESSF zone in the British Columbia Ministry of Forest’s biogeoclimatic ecosystem classification (Meidinger and Pojar 1991), cover approximately 14% of the land area of British Columbia. As more accessible lower-elevation forest types have been logged, a significant and increasing amount of the annual allowable cut from forests on Crown Lands in British Columbia is coming from old-growth ESSF stands. It is estimated that logging of old-growth Douglas-fir forests in the Pacific Northwest has released as much as 1.3% of the global total land-use related carbon flux, although these lands represent only 0.25% of the earth’s forests (Cohen et al. in press). Therefore it is important to understand the impact of the increasing harvest of ESSF forests on the carbon budget of the region. In this study we compare inventory- and satellite-based estimates of carbon storage in ESSF forests in the Southern Interior of B.C. for a 17-year period ending in 1992, and project future trends using a modeling approach that tracks live and detrital carbon pools on the site, and the fate of carbon in harvested biomass.

**Study Area**

This study examined an area of approximately 4.2 million ha located in south central British Columbia ranging from Barriere in the south to McBride in the north (Figure 2.1). Most of the area is Crown Land and is managed by the Kamloops, Cariboo and Prince George Forest Regions of the B.C. Ministry of Forests. Also included are all
of Wells Gray and parts of Bowron Lakes and Mt. Robson Provincial Parks. The landscape is quite varied and encompasses seven zones of the biogeoclimatic ecosystem classification. For this study we were concerned with the 1.3 million ha of the Engelmann Spruce Subalpine-fir (ESSF) zone within the total study area. The ESSF zone is the highest forested zone in the area, occurring just below alpine tundra in mountainous areas. The dominant climax tree species are Engelmann spruce and subalpine-fir, with lodgepole pine (Pinus contorta) frequently occurring as a seral species. Growing seasons are cool and short, and winters are long and cold with mean monthly temperatures below 0°C for 5 - 7 months. Precipitation ranges from 400 - 2200 mm annually, with over 50% falling as snow (Meidinger and Pojar 1991).

**Methods**

There were three major steps to this project. First, a stand-level model of carbon storage was calibrated using data gathered from a chronosequence of ESSF stands, stand yield tables, and data available in the literature. Second, we calculated the age-class distribution of ESSF forest stands and the rate of harvest in the study area using both B.C. Ministry of Forests (MOF) inventory data and satellite imagery. Finally, we combined stand-level estimates of C stores from the model with the age class and harvest data and used a model of the fate of carbon in forest products to estimate the annual net carbon flux from these forests.

**Field Sampling**

We sampled sixteen sites ranging in age from 20 - 235 years. At each site we determined the total biomass of trees, forest floor, and fine and coarse woody debris. Tree diameters and heights had been measured previously at five of these sites by Jull (1990) and we only sampled the detrital pools at these sites. To determine above ground tree biomass at the other 11 sites we established two parallel transects 100 m apart from a random starting point and located three 0.05 ha circular plots at 100 m intervals along
each transect for a total of six plots in each stand. In each plot we measured the diameter of every tree or snag greater than 2 m tall and 5 cm in diameter, and the height of at least two dominant trees of each species. Several dominant and codominant trees were cored to estimate stand age. Stand biomass was calculated with equations developed by Feller and Hamilton (1994) on sites near the geographical center of our study area. We sampled the L, F, and H layers with a 4.7 cm diameter soil core at the plot center and 5 m outside the plot edge in all four cardinal directions. Each sample was a composite of three replicate cores. Samples were dried at 70° C for 72 hours and oven dry weight determined. Fine woody debris (FWD) mass (< 10 cm diameter) was estimated using a line intercept method and a triangular array of 25 m transects at each plot (Trowbridge et al. 1986). To estimate coarse woody debris (CWD) mass we used three 100 m transects in a triangular array at each study site. We measured the diameter of all logs (> 10 cm diameter) intersecting each transect, and assigned each log to one of 5 decay classes (Triska and Cromack 1980; Sollins 1982). The volume of CWD was calculated using line intercept statistics (Pickford and Hazard 1978). We sampled 35 logs from various stands to estimate the density of logs of each decay class. Three cross sections were cut from each sample log and their volume and weights determined in the field. Decay class 5 logs were excavated with a hand trowel. All cross sections were oven dried at 70° C for 72 hours and weighed. Wood density for each cross section was calculated from oven dry weight and cross sectional area. We calculated the mean density for each log weighted by cross sectional area, and then an overall mean density for each decay class. Density estimates were combined with the line-intercept volume estimates to calculate CWD mass at each site. We estimated carbon storage by assuming all biomass was 50 percent carbon.

The Forest Stand Carbon Model

We used the STANDCARB model (Harmon et al. 1996) to simulate the dynamics of live and dead carbon pools in forest stands. The model is a hybrid between a gap simulation model (e.g., Botkin et al. 1972, Aber et al. 1982, Urban 1993) and an
ecological process model. STANDCARB simulates a stand as a series of cells, each representing the area occupied by a single, mature tree. Each cell can have up to four layers of vegetation, six detrital pools, and a stable soil carbon pool. The four vegetation pools in each cell are represented as an upper tree, lower tree, herb, and a shrub layer. The live biomass pools track accumulation of heartwood, sapwood, branches, foliage, fine and coarse roots. Vegetation layers can compete for light and soil moisture. For example, a shade tolerant tree species in the lower tree layer can eventually replace an intolerant species as the upper tree, and trees can shade out herb and shrub layers. As with other gap models, the dynamics of multiple plots are simulated in each model run, and the results are averaged to predict stand level responses over time.

As a first step in model calibration we used the VDYP yield simulator (BC MOF) to produce stand yield tables for spruce on high, medium, and low quality sites and then adjusted the STANDCARB model to approximate these growth rates for boles. Next we calibrated the model so that the growth of the other tree components fell within the range of data from our field plots and the sites measured by Feller and Hamilton (1994) (Table 3.1). Our field sites encompass the range of variability of ESSF stands in our study region. We assumed that the biomass accumulation rates for stands on low, medium and high sites would fall in the lower, middle and higher parts of the range of available data (Figure 3.1a). We used the same technique to calibrate STANDCARB for the coarse woody debris carbon pools, setting the decomposition rates for CWD to approximate our input data (Figure 3.1b). This technique proved problematic for other dead carbon pools because the simulated detrital pools in STANDCARB do not correspond directly to those we measured in the field or that have been reported in the literature. For example, some portions of all detrital pools in STANDCARB enter the stable soil pool, which presumably includes some of the forest floor H layer as well as mineral soil carbon. Our field measurements of litter include the L, F and H layers, and similar data is reported by Feller and Hamilton (1994), who also reported the carbon content of the top 20 cm of mineral soil on their sites. We assigned the average mineral soil carbon content for their sub-hygric and mesic sites to our good and medium sites respectively, and decreased the mesic site value slightly for low sites. We then set the stable soil pool decomposition rate in STANDCARB to maintain a constant pool size, reasoning that this pool is usually
unaffected by standard forest management practices (Johnson 1992). Assigning mean soil carbon values to the field site data allowed a comparison of all dead pools from the model, excluding fine and coarse roots, with the total dead carbon pool as measured in the field, by summing all measured field dead carbon pools (CWD+LFH+FWD+SOIL) (Figure 3.1c) as a final check on the validity of the simulation.

Table 3.1. Field and literature data used in calibration of STANDCARB model. All units are Mg C ha\(^{-1}\) and do not include roots.

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<td>105.59</td>
<td>358.10</td>
<td>463.69</td>
</tr>
<tr>
<td>250</td>
<td>Feller et al. 1994</td>
<td>223.90</td>
<td>31.14</td>
<td>25.50</td>
<td>56.64</td>
<td>76.20</td>
<td>280.54</td>
<td>356.74</td>
</tr>
<tr>
<td>250</td>
<td>Feller et al. 1994</td>
<td>134.15</td>
<td>29.07</td>
<td>21.50</td>
<td>50.57</td>
<td>33.15</td>
<td>184.72</td>
<td>217.87</td>
</tr>
</tbody>
</table>
Figure 3A. Comparison of estimates of a) total live, b) coarse woody debris, and c) total dead carbon pools from field and literature data (points) and predictions from the STANDCARB model (lines) for high, medium, and low productivity sites.
Forest Products Modeling

We used the FORPROD model (Harmon et al. 1996) to track the fate of harvested material in short-term structures, long-term structures, paper supplies, mulch, open dumps and sanitary landfills. Short-term structures are wood products such as pallets with a life-span of less than 20-years, whereas long-term structures include buildings and other products with life spans exceeding 20 years. We assume that 95% of short-term structures are replaced in 30 years, and 95% of long-term structures are replaced in 300 years. FORPROD also tracks the amount of waste material from the disposal of paper, long- and short-term products that is recycled.

We used data from Kurz et al. (1992) to define the fate of biomass harvested from ESSF forests and efficiencies of production of forest products in British Columbia. We assume that 82.2 % of stems are harvested as saw timber, and the remaining 17.8 % are pulp logs. Of the total sawlog mass, 50.5% becomes construction lumber and other lumber products, 25% is chipped for pulp, and the remaining 24.5% is burned as waste. The pulp logs are converted to chips with an efficiency of 85%, and the residue is burned for fuel. The efficiency of the conversion of chips to pulp is dependent on the pulping process used. We assume an overall pulping efficiency for B.C. of 52.3%, computed as the average of the four pulping processes used in B.C., weighted by the percentage of pulp processed with each method.

We examined carbon storage under 3 different scenarios. Our first scenario used the assumptions described above, and a gradual transition from disposal of paper waste in open dumps or by incineration to greater use of landfills and higher paper recycling rates based on data from Oregon and Washington (Harmon et al. 1996). For 1975, we assumed that 14% of paper waste is incinerated, 24% is recycled, 20% goes to open dumps, and 42% ends up in landfills. By the year 2000, we assumed only 10% will be incinerated, recycling reaches 50%, open dumps will no longer be used, and the remaining 40% will be in landfills. In the second scenario we used the constant paper waste disposal rates from the Canadian carbon budget Kurz et al. (1992). In this case only 4.5% of waste is incinerated, 5% is recycled, there is no disposal in open dumps, and the remaining 90.5% of paper waste ends up in landfills. Our third scenario assumed
that mill production efficiencies would increase drastically, with sawlog conversion to lumber increasing to 60% efficiency by the year 2000, and pulping efficiency rising to 85%. The rest of the assumptions were identical to scenario one. To initialize the forest products pools we ran all FORPROD simulations starting with the year 1970, and assumed a constant harvest rate from 1970-1974 equal to the average harvest rate from 1975-1980.

**Carbon Storage at the Stand Level**

Combining output from STANDCARB with FORPROD we can examine the carbon dynamics for a single stand through the course of a rotation (Figure 3.2). During the final 50 years of the previous rotation carbon in the live and dead pools is approximately constant. At harvest there is an immediate transfer of carbon to the forest products pool. The live carbon pool drops to zero, and takes over 200 years to recover to pre-harvest levels. Total dead carbon decreases rapidly after the initial input of logging slash as there is little input in litterfall, and reaches a minimum at 60 years following harvest. Total dead carbon increases after age 60 and reaches pre-harvest levels after 150 years. Carbon in the forest products pool has decayed to less than 50% of the initial mass within 15 years, and only 10% remains after 270 years. Total carbon (live + dead + forest products) is a minimum at 40 years following harvest and takes 270 years to reach pre-harvest levels.
Figure 3.2. Change in carbon stores for a medium productivity stand as predicted by the STANDCARB model. Curves illustrate the final 50 years of the previous 300-year rotation followed by a clearcut harvest and another 300-year rotation.

Age Class Structure from Inventory

We obtained a copy of the 1995 Provincial Forest Inventory from the B.C. Ministry of Forests and used SAS software (SAS 1994) to select all map sheet files that covered the study area, and all forest cover polygons that were in the ESSF zone. We combined all map sheet files into one large inventory file for ESSF forests and used the site index information on each polygon record to assign each polygon to high, medium and low site classes. Using a combination of logging dates, burn dates, and establishment dates we determined the age of each inventory polygon and wrote a program that reconstructed the inventory for each year from 1975-1992. We found a total of 1,106,141 ha of ESSF polygons on the inventory. Of these we could determine the age of 774,717 ha (Table 3.2). The remaining polygons represent alpine areas, agricultural lands, and other non-forested areas, which we excluded from the analysis.

One potential problem with our approach was that not all of the areas harvested between
1993 and 1995 were entered on the inventory files (personal communication, G. Johansen, Inventory Branch). Therefore we assumed a constant rate of cutting after 1992 that was equal to the average harvest rate between 1975 and 1992. To calculate the inventory for 1993-1995 we modified our program to tally those polygons already listed as harvested for a given year and then to randomly select polygons greater than 275 years old for cutting until the average harvest was reached for that year. We used this approach to project the inventory until the year 2015 at a constant harvest rate. When creating annual inventory files if a polygon had a stand age, but no logging or burning date, we assumed it originated from a wildfire. Otherwise polygons were classed as originating after logging. We further subdivided logged polygons according to whether they had been slash burned. As a final step we modified our program to reconstruct the inventory from 1975-2015 assuming that there had been no harvesting during the period by continuing to age polygons instead of allowing them to be harvested at their inventory harvest date.

Table 3.2. Comparison of estimates of total, forested, and disturbed areas (ha) in the ESSF zone of the study area based on MOF inventory and satellite imagery.

<table>
<thead>
<tr>
<th>Source of Estimate</th>
<th>Area (ha)</th>
<th>Total</th>
<th>Forested</th>
<th>Disturbed 1975-1992</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inventory (outside parks)</td>
<td>1106141</td>
<td>774718</td>
<td>73575</td>
<td></td>
</tr>
<tr>
<td>Satellite (outside parks)</td>
<td>1009614</td>
<td>778918</td>
<td>79480</td>
<td></td>
</tr>
<tr>
<td>Satellite (in parks)</td>
<td>276853</td>
<td>196975</td>
<td>1004</td>
<td></td>
</tr>
<tr>
<td>Satellite Total</td>
<td>1286467</td>
<td>975893</td>
<td>80484</td>
<td></td>
</tr>
</tbody>
</table>

**Age Class Structure from Satellite Data**

To compare carbon budget estimates based on inventory data with remote sensing based estimates we used a previously classified image (Chapter 2) to estimate the age class distribution for the study area in 1992. We subset those areas of the image that fell within the ESSF zone, and further subdivided the area into lands that fell within
Combining Inventory Data with Carbon Models

We used STANDCARB to simulate the carbon dynamics of ESSF forests growing on low, medium, and high quality sites. For each site class, we simulated three types of stands; those originating after wildfire, and stands originating after logging both with and without slashburning. Because STANDCARB is a stochastic model, we used

provincial parks and those lands outside parks, because the provincial inventory data only applies to lands outside of parks. Because the ESSF zone contains no dominant deciduous tree species, we reasoned that the total area in the image outside of parks covered by all pixels classified as old, mature, young or semi-open conifer plus those identified as disturbed between 1975-1992 should equal the forested area from the inventory. Indeed, the two estimates differ by less than 1% (Table 3.2). To calculate the carbon budget using satellite data, we developed the age class distribution in 1992, assuming that old-growth stands were 250 years old and mature and young conifer stands were equal to the midpoint of their class age range from the image classification (Table 3.3). We also assumed a constant rate of cutting between 1975-1992 by dividing the total area disturbed by the number of years. Again, we used a SAS program to take this 1992 satellite-based inventory estimate and calculate the forest inventory in previous years to 1975 using the average harvest rate and assuming that only old growth conifer stands were harvested. The satellite data provided no estimate of site index, so we applied the distribution of site class from the inventory data to the satellite estimate.

Table 3.3. Age class and area distribution of ESSF conifer stands in the study area based on satellite imagery.

<table>
<thead>
<tr>
<th>Class</th>
<th>Age Range</th>
<th>Assumed Age</th>
<th>Area (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Outside Parks</td>
<td>In Parks</td>
</tr>
<tr>
<td>old conifer</td>
<td>140+</td>
<td>250</td>
<td>299211</td>
</tr>
<tr>
<td>mature conifer</td>
<td>60-140</td>
<td>100</td>
<td>261395</td>
</tr>
<tr>
<td>young conifer</td>
<td>&lt; 60</td>
<td>30</td>
<td>14847</td>
</tr>
</tbody>
</table>

Combining Inventory Data with Carbon Models

We used STANDCARB to simulate the carbon dynamics of ESSF forests growing on low, medium, and high quality sites. For each site class, we simulated three types of stands; those originating after wildfire, and stands originating after logging both with and without slashburning. Because STANDCARB is a stochastic model, we used
the mean of 10 model runs for each type of stand simulated. For the model simulations we assumed a harvest age of 300 years with removal of 85% of boles. We adjusted the model parameters controlling prescribed burning so that the amount of detritus consumed by a burn matched the mean of 6 prescribed burns following clearcutting of mesic ESSF stands (Feller and Hamilton 1994). We produced lookup tables defining carbon storage in live and dead pools for each stand type for every year. For both inventory based and satellite-based age class distributions in any year we multiplied the area of each age class and stand type by the appropriate value from the lookup tables to generate an estimate of total carbon stores, and summed all age classes for an overall estimate for that year. The total harvested area for each stand type was multiplied by harvest estimates for 300 year-old stands, and summed over stand type to generate a total harvested mass for each year. The total harvested mass for each year was input to the FORPROD model for an estimate of total carbon storage in forest products over time. The annual estimates of total live and dead carbon storage from STANDCARB were combined with the forest products estimates and the net annual carbon flux was calculated as the change in the live pool + the change in the dead pool + change in the forest products pool.

Results

Carbon Budget from MOF Inventory

Based on the Ministry of Forests inventory data, the ESSF forests in this area were a net carbon source of 1.80 Tg to the atmosphere during the period from 1975 - 1992. Average annual carbon flux was 0.14 Mg ha⁻¹. Had there been no cutting during the period, these forests would have been a net sink for atmospheric carbon of 5.17 Tg, which translates to an average annual flux of -0.39 Mg ha⁻¹. Extending the analysis to the year 2015 assuming the average rate of cut from the previous 17 years shows these forests would continue to be a net source of 6.53 Tg of carbon to the atmosphere for the entire 40-year period with an average annual carbon flux of 0.21 Mg ha⁻¹. If we assume
no cutting, and no other major disturbance, these forests would continue to be a net carbon sink totaling 12.93 Tg for 1975 - 2015 with an average annual flux of -0.41 Mg ha\(^{-1}\) (Figures 3.3, 3.4). Not surprisingly, the major difference between the cut and no cut scenarios is the change in live carbon storage. Harvesting reduces the live carbon pool by almost 10 Tg in the 40 year period and only about 4 Tg of this harvested mass is retained in the forest products pool (Figure 3.4).

The carbon budgets mentioned above were calculated with our standard forest products scenario. Changing our assumptions about the efficiency of manufacturing and the amount of paper recycling increased the total amount of carbon stored in forest products by about 0.4 Tg during the 1975 - 1992 period, and by as much as 1.1 Tg over the 40-year simulation (Figure 3.5a). As expected, forest products carbon stores were highest when we assumed very high manufacturing efficiencies. Our standard assumptions produced the least total carbon storage in forest products. However, even under the most efficient manufacturing scenario we predict these forests would be a net source of 5.4 Tg of atmospheric carbon from 1975 - 2015. Under any of the three scenarios, the bulk of the carbon storage in forest products is in landfills and long-term structures (Figure 3.5b).

![Graph](image-url)

Figure 3.3. Estimates of total annual carbon flux from all ESSF forests of the study area with and without harvesting based on inventory data. Estimates after 1992 are projections based on the average rate of harvesting from 1975 - 1992.
Figure 3.4. Carbon budget for ESSF forests of the study area for 1975 - 2015 with and without forest harvesting. A positive change in total stores indicates a net sink and a negative change in total carbon stores indicates a net source of atmospheric carbon.
Figure 3.5. a) Total accumulation of carbon in forest products under three different sets of assumptions regarding efficiency of manufacturing and disposal of waste (see text). b) Accumulation of carbon in the major forest products pools for the standard scenario (see text).
Comparison of Satellite and Inventory-based Carbon Budgets

Calculating the overall carbon budget based on the age-class distribution from the 1992 satellite map indicates that these forests were a net carbon sink of 2.0 Tg with an annual net carbon flux of -0.15 Mg ha\(^{-1}\) yr\(^{-1}\) (Figure 3.6). The satellite-based estimate of total live carbon storage was approximately 10 Tg higher than that from the inventory, but showed a similar pattern of decrease with cutting (Figure 3.7). However, the satellite-based estimate of dead carbon storage increased slightly over the 17-year period while the inventory-based estimate remained constant (Figure 3.7). This occurred because we failed to account for older cuts prior to 1975 in the satellite-based estimate.

The satellite data also allow us to estimate the carbon budget for provincial parks. These lands experienced very little disturbance during the study period (Chapter 2), therefore they were a net carbon sink of 2.0 Tg, and stored 0.59 Mg ha\(^{-1}\) of carbon annually.

Summing the estimates for inside and outside the parks, we estimate that ESSF forests were a net sink for 4.0 Tg of atmospheric carbon during the period 1975 - 1992 with an annual carbon flux of -0.24 Mg ha\(^{-1}\) yr\(^{-1}\) (Figure 3.6). However, even with the park lands added, annual carbon storage decreased steadily, and these forests had become a net carbon source by 1992.
Figure 3.6. Comparison of estimated annual carbon flux from ESSF forests of the study area based on inventory data, satellite imagery, and satellite imagery including areas in provincial parks.

Figure 3.7. A comparison of inventory- and satellite-based estimates of total live and dead carbon stores in ESSF forests of the study area.
Discussion

Our estimates of ESSF carbon budgets are comparable to other recent estimates for North American forests. In 1986, the Canadian forest sector was estimated to be a net carbon sink of 76.8 Tg (Kurz et al. 1992), which translates to an average storage rate of 0.19 Mg C ha\(^{-1}\) based on the reported total forest land area of 404.2 million ha. The Canadian estimate was based on a national inventory that was subdivided into 10 ecoclimatic regions. Our study area lies predominantly within the Cordilleran ecoclimatic region, which was estimated to be a slight net carbon sink of 0.08 Mg ha\(^{-1}\). More recently, Kurz et al. (in press) have subset the B.C. portion of the national carbon budget and estimated that the B.C. forest sector was a net carbon sink of 0.97 Mg ha\(^{-1}\) yr\(^{-1}\) during the period 1920 - 1989, but that the rate of net carbon storage was decreasing. During 1990, the forests of the conterminous United States were a net carbon sink of 79 Tg, or 0.39 Mg ha\(^{-1}\) based on the reported area estimate (Turner et al. 1995). In a finer scale carbon budget for a 1.2 million ha area in the Oregon Cascades, Cohen et al. (in press) combined remote sensing techniques with models of carbon production and decomposition to estimate that forests were a net carbon source of 1.13 Mg ha\(^{-1}\) yr\(^{-1}\) to the atmosphere during the period 1972 - 1991. Extrapolating their results to the 10.4 million ha of Pacific Northwest forests, they estimate a total net carbon flux of 11.8 Tg yr\(^{-1}\) from the region to the atmosphere.

It is interesting that the broad, national or provincial carbon budget estimates based on coarse scale inventories show very slight net carbon sinks, the fine-scale budget from Oregon shows a net carbon source, and our estimate based on inventory data falls between, showing a slight source. In the national and provincial estimates, carbon storage by areas recovering from previous periods of harvesting or other disturbance can outweigh areas of active harvesting that are currently carbon sources. For example, in Canada, the national inventory showed the age-class structure of the forest is skewed towards the earlier stages of stand development (Kurz et al. 1992). Similarly, in the U.S. broad areas in the northeast are recovering from previous harvesting or are reverting to forests after clearing for agriculture (Turner et al. 1995). The western Oregon budget
examines a currently active timber harvest region. The forests were a net source of atmospheric carbon because early successional forests covered 40% of the study area, the bulk of which were a net carbon source because decay of carbon pools from the previous forest was greater than carbon uptake in the young forest (Cohen et al. in press). Our study shows a similar pattern, but a much smaller carbon source because much less of the area has been harvested. Only 8% of the area was less than 20 years old in 1975. Additionally, in ESSF forests, productivity is lower, with carbon storage per unit area on a medium site about 60% of that in Oregon, so potential carbon release by harvesting is smaller. However, our analysis also shows that if harvesting continues at present rates, the area will continue to be a net source of carbon to the atmosphere for years to come.

Our analysis show that, had there been no harvesting, the forests of the study area would have been a slight net carbon sink, slightly lower, but of the same magnitude as the Canadian and American national estimates when viewed on a unit area basis. However, our estimate should be considered as the maximum potential storage for these forests because we assume no losses from disturbance at the stand or individual tree level. Stand level losses due to fires are difficult to predict, but appeared to be minor in our area as only 0.19% of the mature forest area inside parks was disturbed between 1975 - 1992 (Chapter 2). Assuming there was no active fire suppression in provincial parks, this is an estimate of the background stand level disturbance rate for that period. We make no attempt to quantify individual tree mortality due to insects, disease, and windthrow, all frequent events in old-growth ESSF forests.

Our estimate of the carbon budget using satellite imagery is roughly comparable to the inventory-based estimate in total live and dead carbon pool sizes, but we estimate that forests were a slight net carbon sink during the period. There are two reasons why the satellite- and inventory-based patterns through time are not closer. First, we identified the total disturbed area using only satellite images from the two endpoints of the time period, and therefore had to assume an average rate of harvest. While the total harvested area agrees well with that from the inventory, it is clear that the annual variation in harvest levels causes variation in the carbon flux estimate. When we switched to an average rate of cut in 1993 the annual variation in carbon flux decreased (Figure 3.3). This problem could be overcome by using several intermediate images to
give a better idea of the variation in harvest rate, as done by Cohen et al. (in press). A second problem is that we failed to account for any older, regenerating areas prior to 1975. Most of the annual variation in carbon flux under the no cutting scenario is due to decomposing slash in areas harvested prior to 1975 (Figure 3.3). An analysis of the inventory data showed that forest polygons less than 10 years old accounted for 5.5% of the area in 1975. As an experiment, we included these in the satellite based age-class estimate, assuming that some of the area of our shrub or grass classes were actually regenerating cuts. When we recalculated the carbon budget the dead carbon pool matched the inventory-based estimate almost exactly, and the total estimated change in carbon storage for the area outside of parks dropped from 2.0 Tg to 0.54 Tg.

Conclusions

Satellite data can provide carbon budget estimates that are similar to those from a fine-scale forest inventory. It is important to remember that both inventory methods are estimates. We have estimated the satellite mapping has an overall accuracy of 79% (Chapter 2), but have no way to assess the accuracy of MOF inventory data. However, it is encouraging that both methods give similar carbon budget estimates. Obviously satellite-based inventories are critical for carbon budgets, but have some limitations. Differentiating between recovering disturbances and natural, long-term non-forest areas will always be problematic when considering the time period prior to the beginning of the satellite record. If the satellite record is long enough, and several intermediate images are available, stand development should be apparent, and areas that do not change can be considered to be permanent non-forest. Even so, there is little way to tell if some long-term non-forest areas were originally forests before the first image date and if so, when harvesting or other disturbance occurred. Given that recently disturbed and regenerating forests have the greatest rate of change in total carbon storage (Figure 3.2), a mis-classification of a few percent of the landscape can significantly affect carbon storage estimates.
What currently available remote sensing techniques cannot do is measure the productivity or site index of an area. We simply applied the site class distribution from the MOF inventory data to our satellite-based inventory estimate. For western Oregon, Cohen et al. (in press) used an existing site productivity map, but this data is unavailable for many areas. Ultimately, it would be useful to predict site quality from readily available spatial GIS layers such as slope, aspect, elevation, and climate variables.

Another potential problem is the difficulty of assigning age classes to deciduous and mixed stands. This was not much of a problem in the ESSF where such stands are rare, but would be a serious problem where these stands comprise a significant portion of the landscape such as in the Interior Cedar-Hemlock zone.

Despite these limitations, satellite imagery is critical for estimating carbon fluxes from forests over vast regions where inventory data is unavailable, such as the park lands in our study area. Even where inventory data is available it is often difficult to determine the accuracy of the inventory. In addition, satellite data can give spatially explicit carbon budget estimates (e.g., Cohen et al. in press) that are not possible with most current inventories.

Perhaps the largest unresolved issue in large scale carbon budget estimates is predicting soil carbon stores spatially or temporally. Current estimates are that soils and peat contain over two-thirds of global forest carbon (Dixon et al. 1994). However, these estimates are based on less than 20,000 points worldwide. Ideally, regional estimates of soil C storage could be made by relating soil C to site characteristics such as precipitation, temperature, or evapotranspiration. Relationships between soil C and site characteristics vary between forested regions (e.g., Spain 1990, Grigal and Ohmann 1992, Arrouays et al. 1975). Unfortunately predictions in mountainous forested regions are difficult. For example, site characteristics could explain no more than 50% of the variation in soil organic C in the mountainous, forested region of western Oregon (Homann et al. 1995). Predictions of temporal soil C trends at the stand level are also subject to great uncertainty. We only consider soil carbon to a depth of 20 cm and assume, as others have (e.g., Harmon et al. 1990, Kurz et al. 1992, Turner et al. 1992), that the soil carbon pool remains in equilibrium over the course of one rotation. This assumption is based on a review by Johnson (1992) who found that the majority of
studies showed no change (± 10%) in mineral soil carbon following harvesting. This is an untested assumption that needs to be addressed. At our sites a 10% change in soil carbon over 50 years would equal a carbon flux of about 0.1 Mg ha⁻¹ year⁻¹, which could affect our results significantly. Until these spatial and temporal issues concerning soil carbon are resolved, we cannot have complete confidence in forest C budgets.

Recent work by Tans et al. (1990) suggested that carbon unaccounted for in the current global budget is accumulating in terrestrial ecosystems of the northern hemisphere. Two recent studies have suggested northern temperate (and boreal) forests may be accumulating carbon at rates high enough to balance global budget (Sedjo 1992, Kauppi et al. 1992). However, Houghton (1993) has pointed out that both of these studies failed to account for all the major carbon pools, and that had they done so, both studies would have observed a small net sink. He also notes that to balance the current global carbon budget, northern temperate forests would have to accumulate, on average, approximately 2.5 Mg C ha⁻¹ yr⁻¹ or at least 1.0 Mg C ha⁻¹ yr⁻¹ if boreal forests are included. Our results indicate that even with no harvesting, the ESSF forests in our area accumulate carbon at a far lower rate. Despite uncertainties in our analyses, even if our estimate was low by a factor of two these stands would still be accumulating less than 1.0 Mg C ha⁻¹ yr⁻¹ and are unlikely to be a major part of any unknown terrestrial carbon sink.

Our projections show that ESSF forests of this area could be an increasing source of atmospheric carbon for many years if they are clearcut at the current rate. This is not surprising given that ESSF forests are relatively uncut, with the majority of the area in older stands. It has been argued that forests managed for optimum yield will accumulate significantly less carbon than mature or old-growth forests (Cooper 1983, Harmon et al. 1990). While forest management may actually increase carbon storage in ecosystems where natural disturbances are frequent (Kurz et al. 1992, Price et al. 1995), there is little doubt that conversion of older forest types with low natural disturbance frequencies to managed stands will release substantial amounts of carbon, especially if the planned rotation age is significantly shorter than the time needed to reach old-growth. Given that the average fire return interval in ESSF forests is 200 - 300 years (Parminter 1992), and planned rotation ages are a maximum of 150 years, ESSF forests in this area will
continue to be a net source of atmospheric carbon for many years. Increased utilization standards will have only a slight effect on the carbon flux. Alternative silvicultural systems such as partial cutting would ultimately release less carbon per unit area, but the balance may be offset if more land area is cut annually to maintain current harvest levels. This possibility should be explored in future simulations. Ultimately, if the majority of stands are to be harvested by clearcutting, rotation ages in excess of 150 years should be considered to reduce long-term carbon release from the landscape.

**Literature Cited**


Harmon, M.E., Marks, B., and Hejeebu, N.R. 1996. A users guide to STANDCARB version 1.0. A model to simulate the carbon stores in forest stands. Dept. of Forest Science, Oregon State University, Corvallis, OR.


Chapter 4. Conclusions

Results and Immediate Research Needs

This work has shown that LANDSAT imagery is useful for mapping and detecting changes over large forested areas in the interior of British Columbia. It was possible to separate conifer, deciduous, and mixed forest stands, and to delineate three age classes of conifer forest with an overall mapping accuracy of 79%. Unfortunately, it was impossible to separate deciduous or mixed forest stands by age class. A comparison with inventory data showed that disturbances could be mapped very accurately. Areas of regrowth could also be detected, but were much more variable and difficult to map than disturbances. The satellite map proved very useful for spatial analysis of landscape level changes over a 17-year period.

A satellite-based carbon budget estimate agreed reasonably well with that from inventory data. The major differences in budget estimates were caused by a failure of the satellite estimate to account for recovering areas cut during the two decades before the first satellite image. A second source of error stemmed from the need to assign an average rate of cut to the period of satellite record because only two image dates were used.

Some of the difficulties with satellite imagery could be overcome by using additional images from intermediate dates. These would allow an assessment of the changes in the cutting rate, and improve the detection of regrowth areas, which could allow automated mapping of older cuts. However, there will always be some uncertainty associated with areas that were ‘brush fields’ before the date of the first satellite image, and show no signs of recovery during the satellite record. If they have been harvested recently, then detrital material is probably releasing carbon to the atmosphere. Certainly the shapes of such areas would give a clue as to their history, but it is difficult to automatically classify areas based on shape, and manual classification of very large images is time-consuming.
The problem of detecting age class structure of hardwood or mixed stands will be more difficult to overcome, but is important for landscape pattern analysis and carbon budget estimates in biogeoclimatic zones such as the Interior Cedar-Hemlock where these stands are a significant portion of the landscape. Recent work in western Oregon has shown that satellite imagery can detect differences in a canopy texture index in hardwood stands (T. Maiersperger, personal communication). This texture index is calculated from stand spacing, height, and crown cover data, and can be related to stand age or biomass. Such a technique could be tried in B.C., but would require ground data from a large number of hardwood and mixed stands.

**Implications for Forest Management**

The results of the landscape pattern analysis from chapter two indicate that conifer forest matrix is in the early stages of fragmentation. The cutting rate has been relatively low compared to other temperate forest regions. However, we appear to be applying, on average, a very similar type of disturbance pattern in all biogeoclimatic zones. The size and shapes of the conifer patches varies broadly between zones, but the cut blocks are of similar size and shape in most zones. Also, the cut blocks are much smaller, with simpler shapes than the patches that make up the conifer matrix. If we assume that the conifer matrix is an indication of the past disturbance regime then it is obvious that forest harvesting is creating a vastly different landscape pattern than the recent natural disturbance regime in many zones. We need to consider if applying the same general size and shape of cut in all ecosystems is sound management practice.

Much of this study area is still relatively undisturbed and, given that harvesting will continue, we still have a chance to determine what sort of pattern to impose on the landscape. In order to make these decisions we will have to develop a better understanding of the effects of landscape pattern on wildlife populations, and the interactions of various other disturbance agents, such as wildfire, insects and pathogens with harvest patterns. Until such time we should probably err on the side of conservatism, and try to create landscape patterns that fall within the range of conditions
created by natural disturbance regimes, with the implicit assumption that species are best adapted to landscapes they have evolved in.

The carbon budget analysis shows that the ESSF forests from this area have been a slight source of atmospheric carbon during 1975 - 1992. The projections indicate that at the current rate of harvesting, these forests will be a continuing source of carbon for many years. Had there been no cutting, the ESSF forests of the area would have been a slight net carbon sink. The analysis does not take into account any other disturbances besides harvesting and therefore overestimates the sink strength in the absence of cutting. However, I assumed 100% regeneration success on all harvested areas. Given the high rate of regeneration failures in the ESSF, the carbon storage estimates by young stands may be optimistic, thus causing an underestimate of the amount of carbon released when harvesting occurs. In light of these results, it would be prudent to plan on rotation lengths of at least 150 years to minimize the carbon release from these areas. Future simulations should examine whether alternative silvicultural systems such as partial cutting can reduce carbon losses from ESSF stands, and the net effect of removing less volume per hectare, but harvesting more area to maintain current harvest levels.

**Long-term Research Directions**

In this thesis work I have shown that satellite imagery can be used to map broad areas of the interior of B.C. and address large scale research questions regarding landscape patterns and carbon budgets. Clearly, our ability to map older disturbances needs to improve, and we must find ways to distinguish broad age or structural classes of hardwood and mixed forest types, but we can currently generate maps that are very useful. The B.C. Ministry of Forests inventory could be used to extend the carbon budget approach to other forest types in the B.C. Interior so that we can calculate a total carbon budget for the entire landscape. Satellite data could be used to map areas where inventory data are lacking.
Ultimately, what is needed is a spatially explicit landscape model that will allow us to predict the effects of different management scenarios on future landscape patterns. Such patterns could then be assessed for their potential effects on wildlife populations. Annual harvests could be tracked to estimate the change in landscape carbon storage. A landscape model must incorporate simulations of natural disturbance agents such as insects, pathogens, and wildfire, and must be capable of simulating spatially explicit harvest patterns. It must also track the age class, species composition, and detrital stores of all forested polygons. Such models do exist (e.g. Mladenoff et al. 1993), and require base maps such as the one created in this project. The application of such a model in B.C. forests needs to be examined.
Bibliography


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