Detecting landscape changes in the interior of British Columbia from 1975 to 1992 using satellite imagery

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Abstract: To consider the regional scale effects of forest management requires complete and consistent data over large areas. We used Landsat Thematic Mapper and Multispectral Scanner (TM and MSS) imagery to map forest cover and detect major disturbances between 1975 and 1992 for a 4.2×10^6 ha area of interior British Columbia. Forested pixels were mapped into closed conifer, semiopen conifer, deciduous, and mixed forest classes, with further subdivision of the closed conifer type into three age-classes. The image-based estimate of harvested area was similar to an independent estimate from forest inventory data. Changes in landscape pattern from 1975 to 1992 were examined by calculating indices that describe overall landscape pattern and that of conifer and harvested patches in each biogeoclimatic zone. Harvesting affected 8.4% of the forest area outside provincial parks during the 17-year period. Harvested areas were consistently much smaller than conifer patches in all biogeoclimatic zones and had a lower percentage of interior area and perimeter/area ratio. Conifer patch-shape complexity varied between zones; harvested patches had simpler shapes and were similar in all zones. Results indicate that this landscape is only in the early stages of fragmentation, but a similar harvest pattern has been imposed on differing ecological zones.

Résumé : Afin de considérer les effets de l'aménagement forestier à l'échelle régionale, des données complètes et cohérentes sur de grandes superficies sont requises. Nous avons utilisé des images Landsat « Thematic Mapper and Multispectral Scanner » (TM et MSS) afin de cartographier le recouvrement forestier et de détecter les perturbations majeures qui se sont produites entre 1975 et 1992 dans une aire de $4{,}2 \times 10^6$ ha de l'intérieur de la Colombie-Britannique. Les pixels occupés par la forêt étaient répartis parmi les classes suivantes : forêt fermée de conifères, semi-ouvertes de conifères, décidues et mixtes. La classe des forêts fermées de conifères a été subdivisée en trois classes d'âge. L'estimation des superficies récoltées à partir des images était similaire à une estimation indépendante basée sur des données d'inventaire forestier. Les changements des patrons de paysages de 1975 à 1992 étaient examinés par le calcul d'indices décrivant le patron général des paysages, ainsi que le patron des blocs de conifères et des blocs récoltés dans chaque zone biogéoclimatique. La récolte a affecté 8,4% de la superficie forestière à l'extérieur des parcs provinciaux au cours de la période de 17 ans. Les surfaces récoltées étaient régulièrement beaucoup plus petites que les blocs de conifères dans toutes les zones biogéoclimatiques et leur pourcentage de superficie interne ainsi que leur ratio périmètre–superficie étaient inférieurs. La complexité des formes des blocs de conifères variait entre les zones; celle des blocs récoltés était plus simple et demeurait similaire dans toutes les zones. Les résultats indiquent que ce paysage est à un stade de fragmentation peu avancé, mais que des patrons similaires de récolte ont été imposés dans différentes zones écologiques.
[Traduit par la Rédaction]

Introduction

Forest management has traditionally focused on forest stands ranging in size from a few hectares to perhaps a square kilometre. It has become increasingly apparent, however, that stands cannot be managed wisely and efficiently in isolation. Forest management must also take into account the larger "landscape" in which stands occur because management actions on smaller areas can have aggregated effects at regional or global scales. For example, forest harvesting can fundamentally alter landscape patterns (Franklin and Forman 1987; Ripple et al. 1991; Spies et al. 1994) with potential impacts on biological diversity (Harris 1984; Rosenberg and Raphael 1986; Lehmkuhl et al. 1991). Management activities can have effects on regional climate and hydrology (Bonan et al. 1992; Jones and Grant 1996; Bruijnzeel 1996). 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; Dixon et al. 1994; Cohen et al. 1996). Clearly, at least some aspects of forest management must be considered at very large scales, and many have argued for a shift to ecosystem or landscape management of all forest lands (Forest Ecosystem Management Team (FEMAT) 1993; Franklin 1993; Galindo-Leal and Bunnell 1995; Christensen et al. 1996).

To manage very large areas requires data about them. Moreover, the data must be consistent and complete. By consistent we mean that there must be no spatial pattern to the random and systematic error in the data. Alternatively, spatially dependent systematic and random error may be present, but the

Received March 29, 1997. Accepted October 2, 1997.

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Fig. 1. Location of study area in British Columbia.

pattern of its spatial dependence must be quantifiable. By complete we mean simply that the data cover the entire area rather than provide a sampling of it, as would be the case with plot-based methods. Recent advances in satellite remote sensing and geographic information system technologies have made it possible to gather consistent and complete data for very large areas, and such techniques have been used increasingly in forested landscapes (Iverson et al. 1989). In the Pacific Northwest region of the United States, 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), 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. 1996).

In this study we develop a consistent and complete data set on vegetation cover for a large region in south central British Columbia. We also calculate the area harvested during a 17-year period for which satellite imagery was available and compare the remote-sensing-based estimate with an independent estimate based on forest inventory records for the same area. Finally, we document the resulting changes in landscape pattern caused by forest harvesting and discuss potential management implications of these changes as they relate to the concept of large-scale landscape management.

Methods

Study area

This study examined an area of approximately 4.2×10^6 ha in south central British Columbia ranging from Barriere in the south to McBride in the north (Fig. 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 (Table 1). The forests of the ESSF zone were further subdivided by separating the small percentage of the forests found in the drier subzones (Lloyd et al. 1990) in the southern portion of the study area from the wetter ESSF forests dominating the northern half of the area (Table 1). 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.

Table 1. Biogeoclimatic (BEC) zones found in study area.

BEC zone	Leading species	% of study area
ESSFdry	*Picea engelmannii* Parry ex Engelm., *Abies lasiocarpa* (Hook.) Nutt.	3
ESSFwet	*Picea engelmannii*, *Abies lasiocarpa*	28
ICH	*Thuja plicata* Donn ex. D. Don, *Tsuga heterophylla* (Raf.) Sarg., *Picea glauca* (Moench) Voss, *Picea engelmannii*, *Abies lasiocarpa*, *Larix occidentalis* Nutt., *Pseudotsuga menziesii* (Mirb.) Franco, *Pinus monticola* Dougl. ex D. Don, *Populus trichocarpa* Torr. & A. Gray, *Populus tremuloides* Michx., *Betula papyrifera* Marsh.	21
IDF	*Pseudotsuga menziesii*, *Pinus contorta* Dougl. ex Loud., *Pinus ponderosa* Dougl. ex P. & C. Laws.	11
MS	*Picea engelmannii* × *glauca*, *Pinus contorta*, *Abies lasiocarpa*	3
PP	*Pinus ponderosa*, *Pseudotsuga menziesii*	<1
SBPS	*Pinus contorta*, *Picea glauca*, *Populus tremuloides*	5
SBS	*Picea engelmannii* × *glauca*, *Pinus contorta*, *Populus tremuloides*, *Betula papyrifera*, *Pseudotsuga menziesii*	8
AT	Alpine tundra: no trees except krummholz forms	21

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 (25-m cell size) and a portion of the image discarded before we received it. Both TM scenes were joined by georeferencing to B.C. Ministry of Forests (BCMOF) 1 : 20 000 forest cover maps (UTM NAD 27) and resampling TM image 46/23 to a 25-m cell size using nearest neighbor rules. The two MSS images were georectified to the 1992 TM images and resampled to a 50-m cell size using nearest neighbor rules and a series of 60 control points. All georeferencing was done using third-order transformations with a root mean square (RMS) error of less than one pixel. The study area is quite

Table 2. Initial training classes and final class structure used for creation of the 1992 land cover map.

Class	Subclass	Description
Initial training classes		
Closed forest		>25% crown cover
Conifer forest		≥80% of crown cover is coniferous; ages ranging from 10 to >200 years
Deciduous forest		≥80% of crown cover is deciduous; ages ranging from 10 to 100 years
Mixed forest		>20% and <80% of crown cover is deciduous; ages ranging from 10 to 140 years
Semiopen conifer forest	Alpine parkland stands Dry, open forests Selectively logged forests	<25% crown cover, but overstory present
Shrubs		
Grasslands		
Alpine meadows		
Final class structure		
Closed forest		>25% crown cover
Conifer forest	Young	≥80% of crown cover is coniferous <40 years if *Pinus* sp. or *Pseudotsuga menziesii* <60 years if *Tsuga heterophylla*, *Thuja plicata*, *Picea* sp., or *Abies lasiocarpa*
	Mature	41–140 years if *Pinus* sp. or *Pseudotsuga menziesii* 61–140 years if *Tsuga heterophylla*, *Thuja plicata*, *Picea* sp., or *Abies lasiocarpa*
	Old	141+ years
Deciduous forest		≥80% of crown cover is deciduous
Mixed forest		>20% and <80% of crown cover is deciduous
Semiopen conifer forest		≤25% crown cover, but overstory still present
Shrubs		
Grasslands and alpine meadows		

mountainous, and a third-order transformation provided the lowest RMS error and the best visual fit of the MSS to the TM images. Because the two 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.

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. 1998) in the Pacific Northwest. Additionally, a 1 : 250 000 digital elevation model of the area was coregistered to the TM imagery and a cosine incidence angle image was created (Smith et al. 1980).

A 1992 land cover map was produced using the Tasseled Cap TM images and a combination of unsupervised and supervised classification techniques. First, the unsupervised classification algorithm ISODATA (ERDAS, Inc. 1994) was used to separate pixels containing vegetation from nonvegetated pixels. Cover classes associated with spectral classes were assessed using a combination of extensive ground knowledge, air photos, and forest cover maps. Some spectral clusters were confused among more than one land cover class. Image pixels representing these spectral classes were then reclassified using ISODATA. After several iterations the nonvegetated pixels were stratified 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 Institute Inc. 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). A series of 268 training polygons were selected and digitized on the TM Tasseled Cap image using air photos and forest cover maps as references. Training polygons included closed conifer, deciduous, and mixed forest stands of various ages and three types of semiopen 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. Stands were considered closed if they had >25% crown cover (crown class 3 or higher on BCMOF cover maps) and semiopen if crown cover was ≤25% but an overstory was still present. Stands were considered to be coniferous or deciduous if conifers or deciduous species composed at least 80% of the crown cover, respectively. All other closed-canopy stands were considered to be mixed (Table 2). The remaining training polygons represented shrub-covered areas (avalanche tracks and recent brush-covered clearcuts), alpine meadows, and grasslands. For each training polygon the mean values for brightness, greenness, wetness, and cosine incidence angle were extracted from the imagery for use as independent variables in the discriminant functions analysis. In addition, for each of the forest stands, data on the species mix, disturbance date (if any), age-class, and crown cover class were entered from the forest cover maps.

In making a 1992 cover map 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 the separation of forested areas into pure conifer, pure deciduous, and mixed stands (containing both deciduous trees and conifers) was desirable, with further divisions based on stand age or crown cover, if possible. As a first step discriminant functions were calculated with the training polygon data organized into the initial training classes (Table 2), and the classification accuracy was checked using the cross-validation option in SAS (SAS Institute Inc. 1994). This technique minimizes bias by classifying each observation using a discriminant function computed from all other points in the data set, excluding the observation being classified. It was impossible to discriminate reliably between grasslands and alpine meadows or between the three types of open conifer stands, and these classes were combined into the final grassland – alpine meadow and semiopen conifer classes (Table 2). Next, different groupings of the training data were tried to subdivide the closed forest classes based on age-class, species mix, and crown cover class. At each attempt the percentage of correctly classified polygons in each class was examined as well as the way that polygons were misclassified. After numerous iterations it was apparent that it was impossible to further divide the mixed and deciduous classes by age or cover class with any reliability, but that closed conifer stands could be classified into three groups based on age-class. The age of division between the young and mature age-class stands varied depending on the predominant species (Table 2). For stands dominated by *Tsuga heterophylla*, *Thuja plicata*, *Picea* spp., or *Abies lasiocarpa*, the break between young and mature stands occurred at age 60 (age-class 4 on the forest cover maps). For all other stands, which were dominated by *Pinus contorta* or *Pseudotsuga menziesii*, the age break came at 40 years (age-class 3). This makes ecological sense in that the mature 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 *Pinus contorta* and *Pseudotsuga menziesii*. The best break between mature and old conifer stands was at age 140 years in all test stands. This is at least partially an artifact of the age classification scheme used on BCMOF cover maps. Age-classes 1–7 use 20-year intervals, but age-class 8 stretches from 141 to 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 the final discriminant functions were applied to all of the unclassified pixels containing vegetation to produce a classified vegetation map.

Major areas of vegetation change between 1975 and 1992 were identified using a combined five-band image containing the Tasseled Cap axes from the 1975 MSS and 1992 TM images. Although the TM and MSS images differ radiometrically, and in their spectral and spatial resolutions, classification of a combined image works well when the goal is to detect gross vegetation changes. Additionally, the classification of combined images has been shown to be as accurate as image differencing for change detection in Pacific Northwest conifer forests (Cohen and Fiorella 1998). The ISODATA algorithm was used to perform an unsupervised classification on the combined five-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 class differences had been maximized. The disturbed class consisted of many large patches that appeared to match the obvious cut block boundaries on the original 1992 cover map. However, areas in the regrowth class often consisted of many small patches interspersed with areas of no change inside older cut blocks. We decided to combine the regrowth and no change classes because it would have been difficult to use the regrowth information in any landscape pattern analysis, and we were most interested in analyzing the pattern of cutting. This left a map of those areas that had been disturbed between 1975 and 1992. This map was merged with the original 1992 cover map in a GIS overlay operation to produce a 1992 cover map that included a new disturbance class.

The 1992 TM-based map was used in conjunction with the 1975 MSS Tasseled Cap image to make a cover map for 1975. All pixels classified as unchanged from the previous analysis were used 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. Lacking accurate 1975 ground data it was impossible to use a supervised approach. Because of 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, a combined deciduous and mixed forest class and a combined mature and old closed conifer class were used in the 1975 map.

The boundaries of the three major parks in the area were digitized and overlaid on the 1975 and 1992 maps to examine changes in land cover and landscape pattern in actively managed areas separately from those in parks. A digital copy of the B.C. biogeoclimatic zone map was registered to the 1992 TM imagery and used as an overlay to examine landscape pattern and changes by biogeoclimatic zone. As a final step a rule-based merging algorithm was used 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. Only pixels in the disturbed class were allowed to merge in this step. This operation kept large individual cut blocks from being subdivided between BEC zones if only a small fragment of the area (<10 ha) was in a different BEC zone from the majority of the cut.

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, Inc. 1994). Each point was identified on forest cover maps or air photos, its true cover type determined, and a contingency table produced. Accuracy of the disturbed class was assessed in two ways. First, a series of 80 random points on the disturbance image were selected, by 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.

The BCMoF 1995 Provincial Forest Inventory was used for a second, independent assessment of the accuracy of detecting disturbance. The inventory is organized by 1 : 20 000 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 the resulting data file was sorted 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 of these stands from the original estimate to produce a low estimate of the area disturbed. The true inventory estimate should lie between the low and high estimates as stands with only slight overstory removal would probably appear as unchanged in the satellite image classification if the overstory were still dominated by tree canopies.

We could not assess the accuracy of the 1975 land cover map because of a lack of 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. The 1975 map was used solely to analyze the pattern of conifer and disturbed patches.

Landscape pattern analysis

The land cover pattern was analyzed by calculating indices that describe the overall landscape pattern and that of each cover class using the FRAGSTATS computer program (McGarigal and Marks 1995). Numerous patch and landscape indices have been proposed in the literature; however, a recent factor analysis indicated that the majority of the variation in 85 maps of land use and cover could be explained by five univariate metrics, including average perimeter/area ratio, contagion, standardized patch shape, patch perimeter–area scaling (e.g., fractal dimension), and number of attribute classes (Ritters et al. 1995). The first four of these indices were used in our study as well as mean patch size and percentage of a patch type in the landscape that was interior area based on a 100 m edge width. Interior area was included because declines in the extent of large patches of interior forest area have raised concerns about the viability of populations of interior dwelling species, particularly those that rely on old growth such as the northern spotted owl (*Strix occidentalis*) in the Pacific Northwest (Thomas et al. 1990; FEMAT 1993). In the Cascade Range of Oregon and Washington, Chen et al. (1992) demonstrated that the majority of the changes in stocking density, growth, mortality, and regeneration of three conifer species occurred within 120 m of clearcut edges. The tree species in the interior of British Columbia do not reach the heights of those in the Cascades, so a value of 100 m was chosen for edge width in this study.

Three 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:

$$[1] \qquad \mathrm{AWMPFD} = \sum_{j=1}^{n}\left[\left(\frac{2\ln(0.25p_{ij})}{\ln a_{ij}}\right)\left(\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}}\right)\right]$$

where a_{ij} is area (m^2) of class i, patch j, and p_{ij} is 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 with a standard shape, a square in the raster version of FRAGSTATS. The second index, area-weighted mean shape index (AWMSI), is the average shape index of patches of a cover class weighted by patch area:

$$[2] \qquad \mathrm{AWMSI} = \sum_{j=1}^{n}\left[\left(\frac{0.25p_{ij}}{(a_{ij})^{1/2}}\right)\left(\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}}\right)\right]$$

The AWMSI has a value of 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:

$$[3] \qquad \mathrm{LSI} = \frac{0.25\sum_{k=1}^{m} e_{ik}}{A^{1/2}}$$

where A is total landscape area, e is total length of edge in landscape between classes i and k, and m is the number of classes. The LSI has a value of 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.

Two indices of overall landscape pattern were also calculated. 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

Table 3. Error matrix for 1992 TM land cover map.

	Predicted												
Observed	Water	Snow–ice	Clouds	Open	Old conifer	Mature conifer	Young conifer	Deciduous	Closed mixed	Semiopen conifer	Shrubs	Grass–meadow	% correct
Water	10												100
Snow–ice	12	2											86
Clouds			22										100
Open	1	1	14										88
Old conifer					31	9	1						76
Mature conifer					9	41			1	80			
Young conifer				5	4	15	2	8					44
Deciduous								14	3				82
Mixed					2		1		14				82
Semiopen conifer						4		1	3	21			72
Shrubs											18		100
Grass–meadow								1				8	89
% correct	100	92	88	100	66	71	88	78	48	100	100	100	79

Note: κ statistic was 77%.

Table 4. Comparison of estimates of disturbed areas by BEC zone from imagery and forest inventory.

		Area disturbed (ha)		
BEC zone[a]	Total area (ha)	Imagery	Inventory low	Inventory high
ESSF	1 286 467	80 830	57 408	73 575
ICH	866 577	49 699	49 521	55 767
IDF	467 859	13 827	17 470	36 534
MS	113 784	11 099	11 484	12 754
PP	5 971	201	21	96
SBPS	213 972	12 957	13 258	16 074
SBS	342 386	23 765	19 112	21 882
AT	858 664			
Totals	4 155 679	192 378	168 273	216 683

[a]See Table 1 for definition of BEC zones.

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

$$[4] \qquad C = \left[1 + \frac{\displaystyle\sum_{i=1}^{m}\sum_{k=1}^{m}\left[(P_i)\frac{g_{ik}}{\displaystyle\sum_{k=1}^{m} g_{ik}}\right]\left[\ln(P_i)\left(\frac{g_{ik}}{\displaystyle\sum_{k=1}^{m} g_{ik}}\right)\right]}{2\ln(m)} \right](100)$$

where P_i is the proportion of the landscape in class i, g_{ik} is the number of adjacencies between pixels of classes i and k, and m is the number of classes. Contagion ranges from 0 to 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 extent to which patch types are interspered. 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

$$[5] \qquad \mathrm{IJI} = \frac{-\displaystyle\sum_{i=1}^{m}\sum_{k=1+1}^{m}\left[\left(\frac{e_{ik}}{E}\right)\ln\left(\frac{e_{ik}}{E}\right)\right]}{\ln(0.5[m(m-1)])}(100)$$

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

It was necessary to use the simpler cover class scheme from the 1975 map to compare the landscape statistics for the 1975 and 1992 maps. Statistics for the 1992 map were also computed using the more detailed 1992 class structure to examine the effects of combining the mature and old conifer classes. Only the ESSFwet, ICH, and SBS zones had adequate numbers of both old and mature conifer patches for this separate analysis.

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

Results

Accuracy assessment

An overall classification accuracy of 79% was achieved for the 1992 map (Table 3). The nonvegetated classes were accurately identified with only minor misclassification of snow–ice, clouds, and open pixels. These errors all occurred at high-elevation areas of the map. Among the vegetated classes, approximately 20% 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 canopy tree classes.

Table 5. Comparison of land cover in 1992 inside versus outside of parks.

Class	% of total area	% of forest	Inside parks Area (ha)	Inside parks Area (%)	Outside parks Area (ha)	Outside parks Area (%)
Water	3.4		29 161.7	3.6	113 634.1	3.3
Clouds	3.9		64 061.4	7.9	103 446.9	3.0
Snow	9.3		135 528.7	16.6	261 552.6	7.6
Open	12.6		139 625.4	17.1	320 527.4	9.4
Closed old conifer	17.0	28.8	188 862.6	23.2	595 133.8	17.4
Closed mature conifer	23.3	36.4	97 396.7	11.9	896 321.9	26.2
Closed young conifer	1.8	2.8	11 820.4	1.5	64 800.5	1.9
Closed deciduous	3.5	5.5	14 331.1	1.8	134 302.7	3.9
Closed mixed	6.4	10.0	32 166.4	3.9	239 838.8	7.0
Semiopen conifer	10.6	16.6	77 437.8	9.5	374 276.8	10.9
Shrubs	1.7		12 980.6	1.6	58 725.3	1.7
Grass and meadows	2.0		10 492.1	1.3	73 147.9	2.1
Disturbed	4.5		1 564.6	0.2	192 381.5	5.6
Total	100.0		815 429.3	100.0	3 428 090.1	100.0
% of total area in parks	19.2					

Major vegetation disturbances were detected quite accurately. Of the 80 pixels examined, only one was misclassified, leading to an overall classification accuracy of 99% for disturbances. The lone error was due to slight misregistration of an avalanche track between the 1975 and 1992 images.

The 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 4). Comparing estimates of disturbance by BEC zone was problematic for several reasons. First, in the inventory data 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 when the pieces were larger than 10 ha. Second, there was some error induced by slight image misregistration and the differences in spatial and spectral resolution between TM and MSS imagery. 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.

Land cover

Forests covered 62.6% of the total study area in 1992, and mature and older conifer forests accounted for 40.3% of the total area (Table 5). Areas disturbed between 1975 and 1992 covered 4.5% of the total area. Examination of the cover map (Fig. 2) and the forest inventory comparison (Table 4) indicate that the great majority of these disturbances were from forest harvesting. Water, snow, and ice covered 12.7% 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 semiopen 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 (Fig. 2).

Major provincial parks cover 19.2% of the study area (Table 5) and are generally in high-elevation areas as evidenced by the higher percentage of snow and open areas (predominantly alpine) than in the area 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. A GIS overlay operation using the 1992 and 1975 maps showed that 95% of the disturbed pixels were classified as mature + old conifer forest in 1975; the other 5% were semiopen, or mixed–deciduous forest in 1975. Thus, the disturbed area represents 6.6% of the total forest in the landscape. More importantly, 8.4% of the forested area outside of the parks was disturbed from 1975 to 1992, whereas disturbance affected only 0.4% of the forest area inside park boundaries.

Changes in land cover between 1975 and 1992 maps

A direct comparison of land cover in 1975 and 1992 showed that 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 consisted of 193 000 ha of disturbances, almost exclusively outside park boundaries (Fig. 3). There was a slight increase in open area during the period and a decrease in snow-covered area. The latter 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 due to disturbances and some to misregistration of the imagery or the differences in spatial and spectral resolution between the TM and MSS images, but most is likely due to variation in snow cover between the two dates. These differences are of minor importance because the 1975 land cover map was only used to examine changes in the landscape pattern of conifer and disturbed patches.

Landscape pattern

The area inside parks changed very little during the 17-year period studied, as only 0.19% of the area was disturbed

Fig. 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). Provincial parks are outlined in yellow.

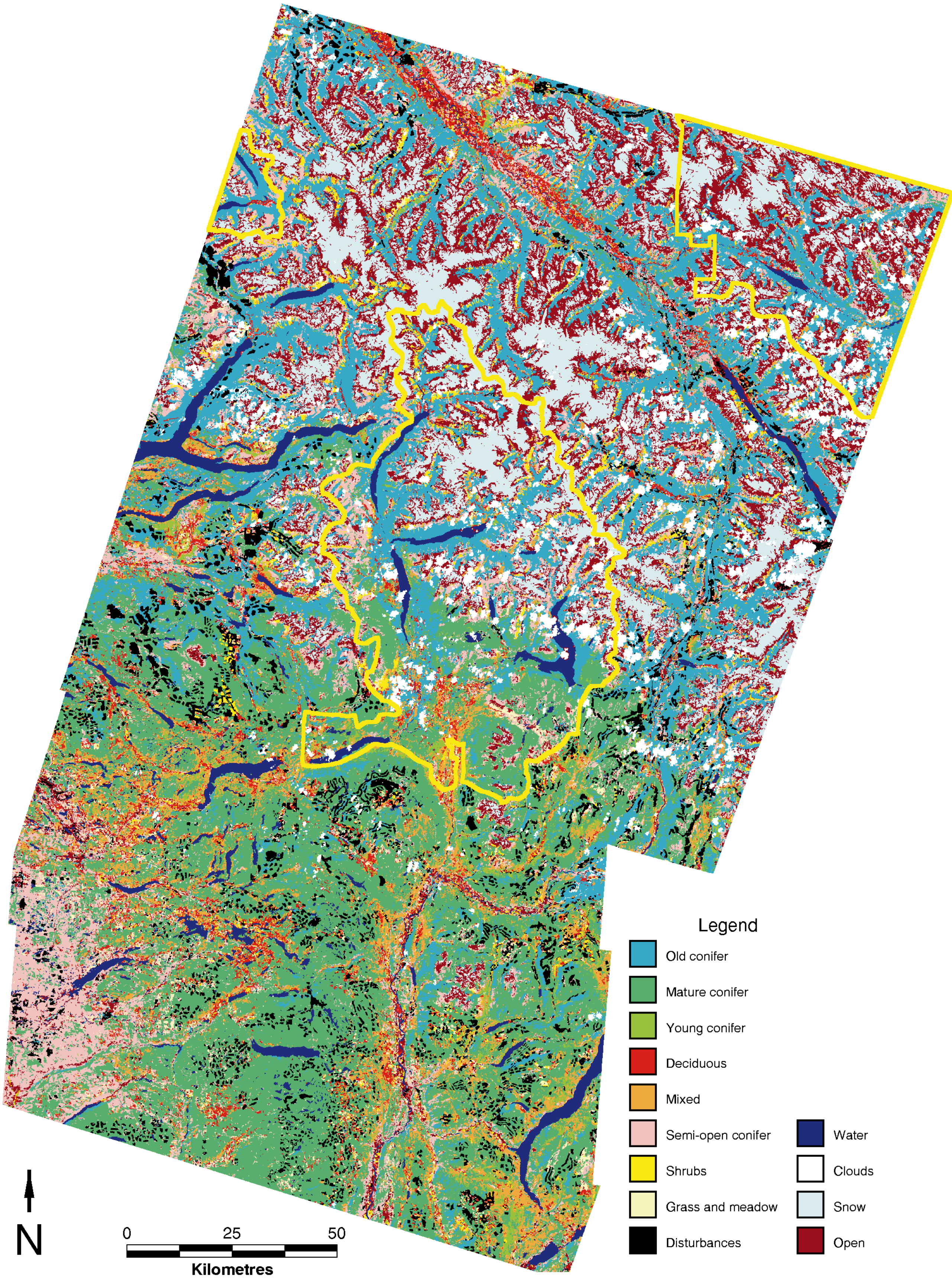
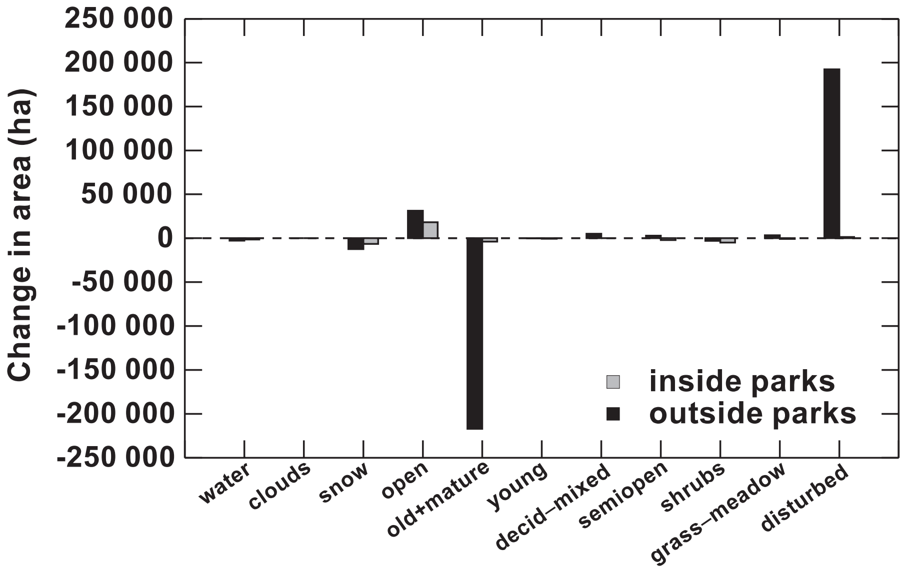


Fig. 3. Change in cover classes during the period 1975–1992. Cover class labels are as follows: old + mature, old + mature conifer; young, young conifer; decid–mixed, deciduous + mixed stands; semiopen, semiopen conifer. The rest of the labels match the text.



(Table 5); therefore landscape change statistics are presented only for the areas outside parks. The bulk of timber harvest in the study area occurred in mature and old conifer stands, thus we report patch statistics for only the mature + old conifer and harvested land types. The average conifer patch size decreased in all but the IDF and PP zones, and the average perimeter/area ratio of conifer patches increased slightly in all zones (Figs. 4*a*, 4*b*). The perimeter/area ratio of disturbed areas was lower than that of conifer patches in all zones (Fig. 4*b*). The percentage of interior area of conifer patches decreased slightly 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 (Fig. 4*c*). The mean patch size of disturbed areas was very similar across zones, ranging from 20 to 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 (Figs. 5*a*, 5*b*, 5*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 and interior area the most (Figs. 6*a*, 6*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 disturbed patches. Old and mature conifer stands had higher perimeter/area ratios than disturbed patches in all three zones. The amount of interior 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 (Figs. 6*d*, 6*e*, 6*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 alter either of these indices.

Discussion

The disturbance rate in our study area is lower than most remote-sensing-based estimates from other temperate forests. In British Columbia, 8.4% of the forest land outside the parks was disturbed in 17 years (0.49%·year^{-1}); the great majority of these disturbances were forest harvesting. There was a decrease of 10.9% (0.64%·year^{-1}) in mature or older conifer forest area. In a 1.2×10^6 ha area of western Oregon, 14.7% of the total forest land was harvested between 1972 and 1993 (0.7%·year^{-1}) (Cohen et al. 1998). In a 259 000 ha subset of the same Oregon area, 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. Turner et al. (1996) 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 lands 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. The estimate of annual disturbance rates inside the park boundaries in the B.C. study area (0.02%) is considerably lower than those reported for wilderness areas in Oregon and Minnesota. However, Zheng et al. (1997) 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 in this portion of British Columbia is at the low end of the range for a variety of temperate forests throughout the world. However, forest growth

Fig. 4. (*a*) Mean patch size, (*b*) patch perimeter/area ratio, and (*c*) percentage of a patch type in the landscape that is interior area based on a 100-m edge for disturbed areas, and conifer areas in 1975 and 1992. Error bars in Figs. 4*a* and 4*b* represent 1 SE of the mean.

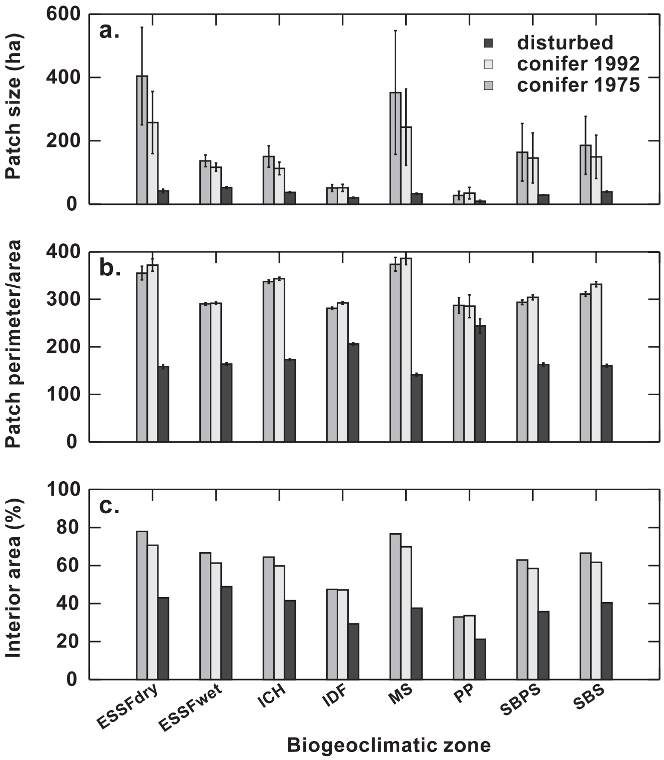
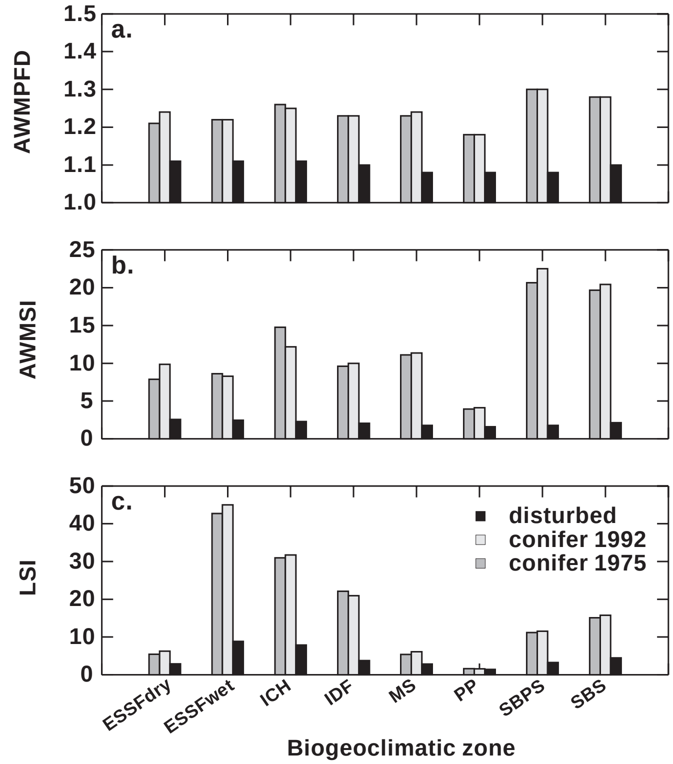


Fig. 5. Patch complexity as measured by (*a*) area-weighted mean patch fractal dimension (AWMPFD), (*b*) area-weighted mean shape index (AWMSI), and (*c*) landscape shape index (LSI) (see text) for disturbed areas, and conifer areas in 1975 and 1992.



rates in the area 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 (Spies et al. 1994). The planned rotation length on public lands was 80 years at the time of that study, but has since been lengthened (FEMAT 1993). 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). To estimate rotation ages, 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 (Mitchell et al. 1995), to estimate CMAI for several of the major conifer species. In the ESSF zone, CMAI for planted stands of white spruce (*Picea glauca*) 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 ESSF zone, because we assumed no regeneration delay when running the WinTIPSY model.

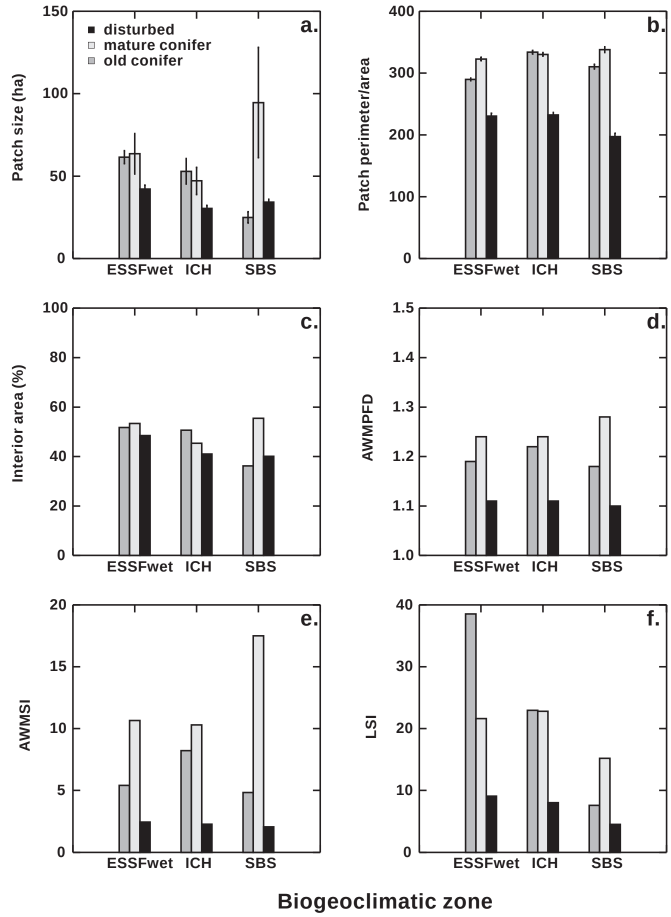
The results of this analysis show the very early stages of fragmentation of the conifer matrix outside the parks during the 17-year period of record. Decreases in the mean conifer patch size and percentage of interior area were detected in most BEC zones. These results are consistent with those reported 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 to 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 nonforest patches over a 16-year period (Zheng et al. 1997). Given that the landscape was classified into only forest and nonforest patches, the results indicate a decrease in forest patch size, and increases in edge and shape complexity of the forest patches.

That the shape complexity of conifer patches changed very little during the study period is indication that the conifer matrix in the study area is still relatively intact. Moreover, this landscape is well below the disturbance thresholds discussed by Franklin and Forman (1987), whose simulations of a checkerboard cutting pattern exhibited rapid declines in forest patch size when 30% of the landscape was cut, and increases in cutover patch size when cutting affected 50% of the landscape. 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; Mladenoff et al. 1993; Skole and Tucker 1993; Zheng et al. 1997). That there was no detectable change in the two landscape-level indices, IJI and C, is further evidence that the B.C. study area is only in the early stages of fragmentation.

Though we documented only minimal fragmentation, it is clear that the patches created by logging during the study period are much smaller and simpler in shape than the mature + old conifer patches making up the conifer matrix (Figs. 4, 5, 6). Based on results from other forests worldwide, if the rate and pattern of disturbance in the study area continues, major changes in the conifer forest matrix will result. For example, in an area on the border of Wisconsin and Michigan a managed landscape had more numerous, smaller, and simpler patches than a nearby old-growth wilderness area (Mladenoff et al. 1993). In western Oregon, conifer edge was greatest when 40% of the landscape was cut, and declined with increased cutting (Spies et al. 1994). In British Columbia, 64.5% of total forest was still in mature and old conifer stands and recent disturbances made up less than 5% of the entire landscape, suggesting that the conifer matrix is still dominant due to the relatively low disturbance rate and short cutting history.

There is also evidence that a similar size and shape of forest harvest unit have been applied uniformly across most of the BEC zones in the study area during the period of record. The sizes and shapes of conifer patches vary broadly between BEC zones, yet the variation in cut block size and shape by zone is much narrower (Figs. 4, 5, 6). 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 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 composed of wildfires larger than 500 ha (DeLong and Tanner 1996). 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 to 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 (Figs. 4, 5).

Fig. 6. Comparison of (*a*) mean patch size, (*b*) mean patch perimeter/area ratio, (*c*) percentage of a patch type in the landscape that is interior area based on a 100-m edge, (*d*) area-weighted mean patch fractal dimension (AWMPFD), (*e*) area-weighted mean shape index (AWMSI), and (*f*) landscape shape index (LSI) for disturbed, old, and mature conifer patches in 1992.



Management implications

This study has documented a relatively low annual rate of disturbance over a large area of interior British Columbia. Spatial analysis of the cutting pattern showed that the conifer forest matrix is only in the very early stages of fragmentation. There is considerable variability in the size and shape of mature + old conifer patches among BEC zones, yet a similar size and shape of harvest patch has been imposed across most BEC zones. Essentially, forest zones that are quite different are being pushed towards a similar landscape pattern by forest harvesting. It is important to remember that these results are averages over a 4.2×10^6 ha area. Examination of the 1992 land cover map (Fig. 2) clearly shows that some watersheds have been severely impacted by cutting, while others have remained untouched. Nonetheless, given that timber harvesting will almost certainly continue on a large portion of the landbase outside of the provincial parks, it is inevitable that the overall landscape pattern will be changed. Forest managers have the opportunity to control this change and the obligation to consider the sort of pattern that should be created (Franklin and Forman 1987; Li et al. 1993; Wallin et al. 1994).

Ignoring the spatial arrangement of forest harvesting can have serious long-term consequences. In the U.S. Pacific Northwest, 50 years of dispersed small clearcuts on federal lands have resulted in a markedly more fragmented landscape (Ripple et al. 1991; Spies et al. 1994), with potential effects on wildlife populations (Harris 1984; Lehmkuhl et al. 1991), most notably the northern spotted owl (Thomas et al. 1990; FEMAT 1993). There will also undoubtedly be interactions between the spatial pattern of harvesting and various natural disturbance agents (Turner et al. 1989). Thus, 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. 1996).

Given an incomplete knowledge of how natural disturbance regimes and forest management interact to determine landscape pattern, and how wildlife populations respond, 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 without reducing 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. Although landscape patterns in many undisturbed portions of our study area have already been influenced by fire suppression, clearly, by imposing a similar disturbance pattern across most BEC zones, there is a risk of 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 well outside the range of conditions experienced in the previous 500 years (Wallin et al. 1996).

In the interior of British Columbia there is a unique opportunity to practise large-scale landscape management for several reasons. First and foremost, as this study indicates, the conifer matrix is still relatively intact, so that forest management is less constrained by a legacy of cutting than in other parts of the world. Additionally, the large areas of relatively undisturbed forest support viable populations of large carnivores and herbivores, and the bulk of forest land is under a single public ownership, making planning and the implementation of policy much easier than in multiownership landscapes (Galindo-Leal and Bunnell 1995). However, large-scale management opportunities are decreased as cutting proceeds. 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. The pattern of forest disturbance must be carefully considered. Altering the spatial pattern of forests over a large landscape takes no more than one rotation. Erasing that new pattern will take much longer.

Regardless of the choice of forest cutting pattern, management at the landscape or regional scale will require complete and consistent data on large areas. We have demonstrated that satellite data can be used to rapidly map and detect changes in vegetation cover and landscape pattern over large areas with reasonable accuracy. More accurate, spatially explicit forest inventory information may eventually become available, but probably never for all areas of the landscape. Also, updating inventories over large areas is expensive and time-consuming, whereas change detection using satellite data can be done rapidly. Detection and monitoring of major changes in landscape pattern are crucial to understanding the cumulative, large-scale effects of past and future forest management decisions.

Acknowledgements

This research was completed as part of a doctoral program in the Forest Science Department at Oregon State University and was funded jointly by the NASA Global Change Fellowship Program, Forest Renewal British Columbia (FRBC Award FR-95/96-37), and the B.C. Ministry of Forests through the Kamloops Forest Region and Research Branch. Inventory files and some satellite images were provided by Inventory Branch, B.C. Ministry of Forests. The authors gratefully acknowledge the assistance of A. Vyse of the Kamloops Forest Region, M. Eng of Research Branch, G. Johansen of Inventory Branch, D. Lousier and M. Jull of the Prince George Forest Region, and D. Salayka of the McBride Forest District. Assistance with the FRAGSTATS program was provided by B. Marks of the Forest Science Department, Oregon State University. We also thank M. Harmon, P. Comeau, E. Jensen, and three anonymous reviewers for commenting on an earlier version of the manuscript.

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