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Rates and patterns of landscape change between 1972 and 1988 in the Changbai Mountain area of China and North Korea

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Abstract

Satellite imagery was used to quantify rates and patterns of landscape change between 1972 and 1988 in the Changbai Mountain Reserve and its adjacent areas in the People's Republic of China and North Korea. The 190,000 ha Reserve was established as an International Biosphere Reserve by The United Nations Educational, Scientific and Cultural Organization (UNESCO) in 1979. It is the most important natural landscape remaining in China's temperate/boreal climate. The images used in this research cover a total area of 967,847 ha, about three-fourths of which is in China. Imagery from 1972 and 1988 was classified into 2 broad cover types (forest and non-forest). Overall, forests covered 84.4% of the study area in 1972 and 74.5% in 1988. Changes in forest cover within the Reserve were minimal. The loss of forest cover outside the Reserve appears to be strongly associated with timber harvesting at lower elevations. Landscape patterns in 1988 were more complex, more irregular, and more fragmented than in 1972. This is one of the few studies to assess landscape changes across two countries. The rates and patterns of forest-cover loss were different in China and North Korea. In North Korea, extensive cutting appears to have occurred prior to 1972 and this has continued through 1988 while in China, most cutting appears to have occurred since 1972.

Introduction

Detecting rates and patterns of landscape change is considered an important issue in ecological research for several reasons (Lubchenco *et al.* 1991). First, land use plays an important role in the global carbon budget and the potential for significant climate change as a result of increases in atmospheric CO_2 concentrations has been the subject of great concern and debate (Keeling 1973; Bolin 1977; Woodwell *et al.* 1978; Brown and Lugo 1980, 1982; Houghton *et al.* 1983; Adams *et al.* 1990; Tans *et al.* 1990). In many areas change in forest land has resulted in a net C flux to the atmosphere (Harmon *et al.* 1990; Houghton and Skole 1990; Dixon *et al.* 1994; Flint and Richards 1994; Wallin *et al.* 1996; Cohen *et al.* 1996). Second, land clearing usually results in a significant forest fragmentation (Franklin and Forman 1987; Ripple *et al.* 1991; Li *et al.* 1992; Skole and Tucker 1993; Wallin *et al.* 1994) and this may have a significant impact on biological diversity (Harris 1984; Rosenberg and Raphael 1986; Pulliam 1988; Lehmkuhl *et al.* 1991; Carey *et al.* 1992). Third, implementation of conceptual models for landscape change and conservation area design begins within a description of current landscape patterns and trends (Franklin and Forman

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1987; Harris 1984). Analysis of the recent history and present patterns of forests offers a present day baseline for assessing future landscape patterns and their consequences.

Since landscape types are consistently changing, studies of landscape dynamics at large spatial extent would be difficult without the development of remote sensing techniques during the last two decades. Such studies, in combination with the increasing availability of remotely sensed data and new methods in spatial modeling and GIS (geographic information system), have increased the extent and accuracy of assessing rates, patterns, and direction of regional change (Hall *et al.* 1991; Spies *et al.* 1994; Cohen *et al.* 1995).

Applications of remotely sensed data to illustrate changes in forests over time have been reported by many investigators (Nelson *et al.* 1987; Sader and Joyce 1988; Iverson *et al.* 1989; Green and Sussman 1990). These studies have focused on rates and patterns of land conversion. Some studies have documented changes in landscape characteristics in terms of patch size and edge effect (Skole and Tucker 1993; Spies *et al.* 1994; Turner *et al.* 1994). Most studies of landscape change are focused on tropical forests (Myers 1980; Tucker *et al.* 1984; Mares 1986; Jenkins 1987; Nelson *et al.* 1987; Sader and Joyce 1988; Fearnside *et al.* 1990; Green and Sussman 1990; Bjorndalen 1992; Skole and Tucker 1993). Relatively few studies have been reported for temperate forests (Hall *et al.* 1991; Ripple *et al.* 1991; Fiorella and Ripple 1993; Luque *et al.* 1994; Spies *et al.* 1994; Turner *et al.* 1994).

This study represents the first systematic analysis of landscape change using remote sensing data in the Changbai Mountain area. The Changbai Mountain rises steeply from the surrounding plain. It is one of the few places in the world where such a large vertical zonation can be found within such a short horizontal distance and with only minimal disturbance by human activities until the 1970s. Substantial timber harvesting in the Chinese portion of the study area has been carried out by the state-run forest bureau since the mid-1970s. Little is known about forest management practices in the North Korean portion of the study area.

The objectives of this research were to: 1) quantify rates of change in forest cover for the 16-yr period between 1972 and 1988; 2) characterize changes in landscape pattern resulting from timber harvesting; 3) contrast the patterns and dynamics in the Chinese and Korean portions of the study area with special emphasis on a comparison of areas inside and outside the Changbai Reserve; and 4) compare the results from this study with results from other regions worldwide.

Study area

The Changbai Mountain range stretches along the boundary between China and North Korea (Fig. 1). The images used in this research cover a total of 967,847 ha. Three-fourth of the image is in China including the 190,000 ha Changbai Mountain Reserve. The study area extends from 41.3 to 42.4 degrees north latitude and 127.4 to 128.7 degrees east longitude. Elevations range from about 410 to 2740 m above the sea level with slope gradients between 0 and 73 degrees.

The climate of Changbai Mountain is characterized by cold weather during the long winter, and short, cool summers. The annual mean temperatures in the study area range from –7°C to 3°C. The annual average precipitation ranges from 700 to 1400 mm.

The flora and fauna of Changbai Mountain are very diverse. There are 127 plant genera, including 1,477 species of higher plants and 510 species of lower plants. More than 300 vertebrate species have been recorded in the area (Tao 1987). Distinct vertical zonation in Changbai Mountain can be generally described as four natural zones: a) the alpine tundra zone lies in the upper part of the volcanic cone above 2000 m where the main species are *Rhododendron* and *Vaccinium*; b) *Betuli ermanii* (Erman's birth) forms pure stands between 1800 and 2000 m; c) a coniferous forest zone exists between 1100 and 1800 m, where the main species are *Picea jezoensis*, *Abies nephrolepis*, and *Larix olgensis*; and d) a mixed forest of broadleaved and coniferous trees is found between 500 and 1100 m, where the main species are *Pinus koraiensis*, *Tilia amurensis*, *Quercus mongolica*, *Acer mono*, *Fraxinus mandshurica*, *Ulmus propinqua*, and *Abies holophylla* (Yang *et al.* 1987).

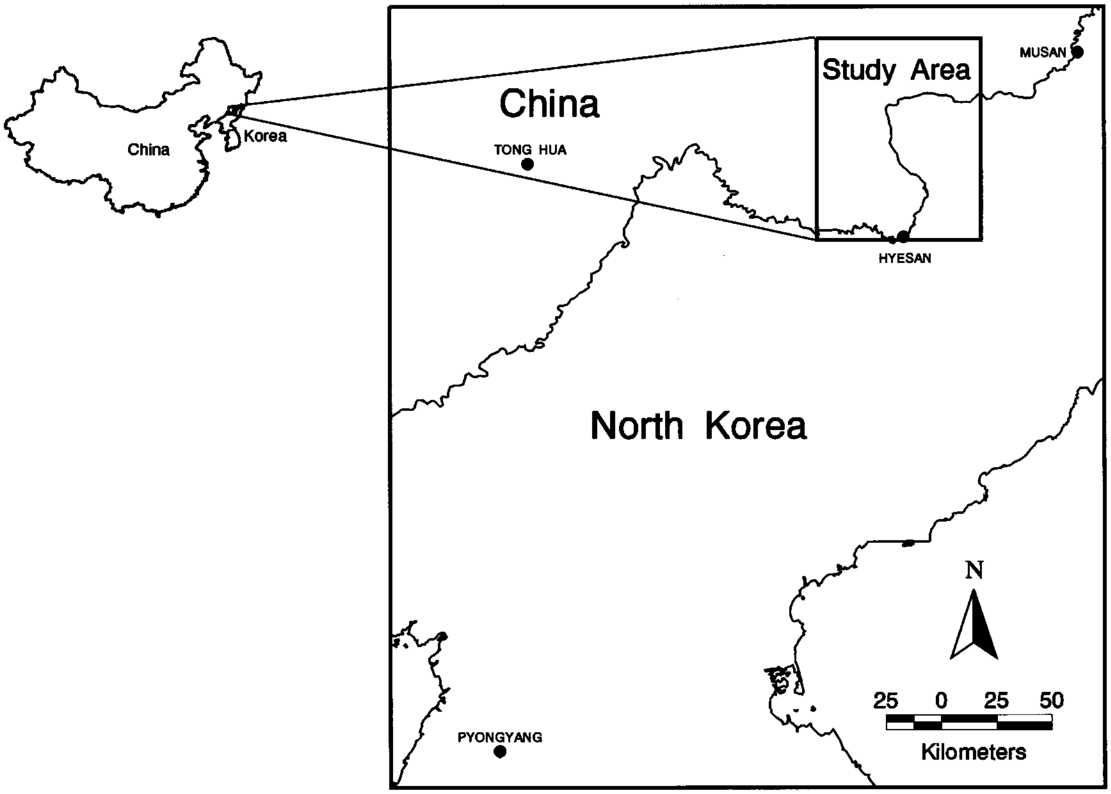


Fig. 1. Orientation map for the study area.

Methods

Data acquisition and processing

A Landsat 5 Thematic Mapper (TM) image (20 September 1988) and a Landsat 1 Multispectral Scanner (MSS) image (31 October 1972) for The Changbai area were used in this study. Prior to analysis, the TM image was rectified to a universal transverse mercator (UTM) projection with a pixel resolution of 30 m using nearest neighbor rules provided by ERDAS (Earth Resource Data Analysis System). To perform a pixel-by-pixel comparison of the two data sets through time, the MSS data was also rectified to the UTM projection and resampled to 30-m pixel resolution. This approach is commonly used when lower resolution images are compared to higher resolution image (Iverson and Risser 1987; Sader 1987; Walker and Zenone 1988; Jakubauskas *et al.* 1990; Price *et al.* 1992). The reported error (root mean squares, RMS) for the registration between the 2 images was < 0.5 pixels. The raw digital satellite image for 1988 was transformed into the brightness, greenness, and wetness axes of the TM Tasseled Cap (Crist *et al.* 1986) and these axes were used as input data for classification of land-cover map in 1988. The Tasseled Cap is a guided principal components analysis that results in a standard or fixed transformation (Freiberger 1960). Since Landsat 1 and Landsat 5 carry different sensors, the calculation of brightness and greenness values for the MSS scene was performed using a different equation provided by Kauth and Thomas (1976), however, the brightness and greenness values calculated from the two equations are equivalent and comparable.

Classifying land-cover maps in 1988 and 1972

Major procedures to obtain land-cover maps in 1988 and 1972 include: 1) generating Tasseled Cap image from the 1988 TM scene; 2) classifying land-cover types for 1988, based on the Tasseled Cap image; 3) calculating differences in both the brightness and greenness channels between 1972 and 1988 images to obtain two change maps; and 4) inference of land-cover in 1972, using the classified 1988 land-cover map and the two change maps (Fig. 2). Each of these steps (except step 1) are reviewed below.

Land-cover map in 1988

The transformed Tasseled Cap data set was used as input to generate a land-cover map for 1988. We used ISODATA and MAXCLAS unsupervised classification methods (ERDAS 1993). The ISODATA method uses the minimum spectral distance formula to form clusters. MAXCLAS is the primary multispectral classification program. This program classifies the input image using one of three decision rules. We used the maximum likelihood rule for image processing. The outcome clusters were then grouped into two broad categories: forest and non-forest. The forest cover may include Erman's birch, coniferous forests, mixed deciduous and conifer forests, and birch. The non-forest cover may include recent clearcuts, permanent open areas, residential areas, agricultural lands, early stages of secondary growth, grasslands, and alpine tundra.

Change maps between 1972 and 1988

The differences between 1972 and 1988 values in both the brightness and greenness channels, defined as $value_{1972}$ minus $value_{1988}$, were calculated and used to generate two change maps. For the brightness channel, a high negative number indicates a change from forest to non-forest and a high positive number indicates a change from non-forest to forest. For the greenness channel, the reverse applies, a high negative number illustrates a change from non-forest to forest and a high positive number illustrates a change from forest to non-forest category. For the purpose of image analysis, the calculated differences in both channels were rescaled to eliminate negative numbers. The rescaled values on the change maps were then

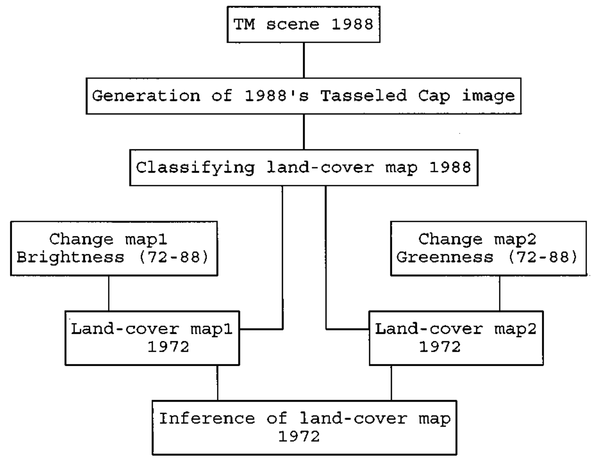


Fig. 2. Flow chart illustrating the development of the 1972 land-cover map based on the classified 1988's land-cover map and two 1972–1988 change maps.

grouped into 3 categories: a) disturbance (forest to non-forest), b) unchanged, and c) regeneration (non-forest to forest) (Fig. 3). Segmenting these two change maps into these three classes involves the selection of appropriate thresholds; areas that have undergone either disturbance or regeneration are readily apparent when comparing the unclassified 1972 and 1988 scenes. Thresholds were selected to produce change maps that were consistent with these readily identifiable changes. In this study, we selected thresholds that represent cumulative frequencies of 20% and 90% in the brightness difference (Fig. 3a) and 10% and 80% in the greenness difference (Fig. 3b) based on the fact that a negative relationship exists between the differences of brightness and the differences of greenness in this region (data not shown). Two change maps were developed; change map1 was based on the brightness difference and change map2 was based on the greenness difference.

Inference of 1972 land-cover map

Two preliminary versions of a land-cover map for 1972 were produced. Land-cover map1 was inferred according to the 1988 classified land-cover map (2 categories) and the change map1 (3 categories), while land-cover map2 was inferred using the 1988 classified land-cover and the

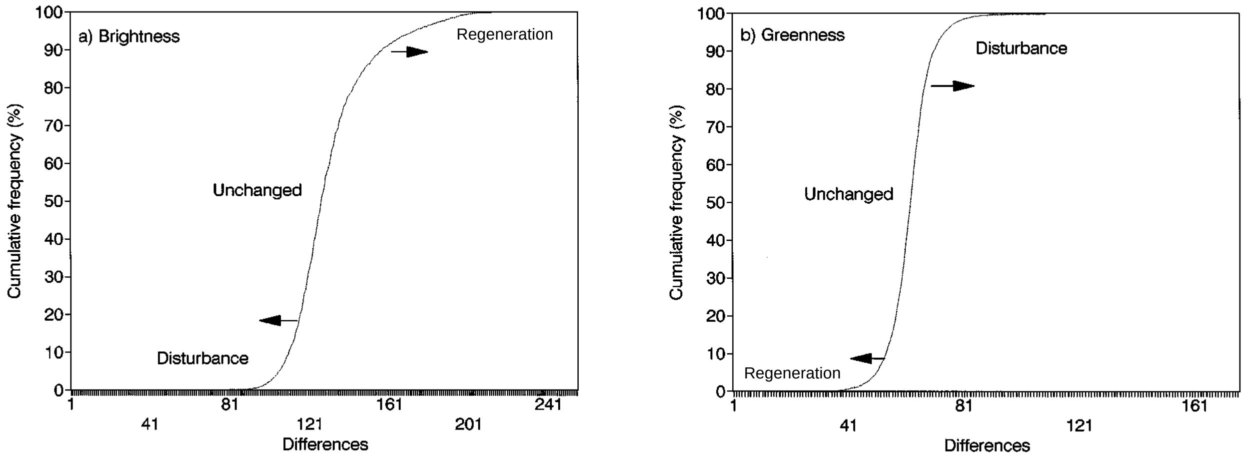


Fig. 3. Cumulative frequency distribution of differences (value$_{1972}$–value$_{1988}$) between: a) brightness; and b) greenness. These differences were rescaled to values between 0 and 255 to eliminate negative numbers. Thresholds used to identify disturbance and regeneration are indicated by arrows (see text for details on determination of these thresholds).

change map2 (Table 1). These two preliminary maps were merged into one final land-cover map for 1972 in such a way that the non-forest class was assigned for a given pixel only if that pixel had been classified as non-forest on both land-cover map1 and map2; otherwise, the forest class was assigned. The lands located above 1800 m were excluded from the change detection because climatic limitations make it impossible for coniferous forests to exist above that elevation.

Accuracy assessment

Unfortunately, we did not have access to aerial photographs or ground data that could be used to conduct an accuracy assessment of our 1972 and 1988 land-cover maps. Instead, the accuracy of the maps was assessed by visually interpreting the unclassified satellite images. A systematic sample of 345 points was obtained for each scene across the entire study area. Identification of land-cover types (non-forest or forest) of the sampling points for each scene was determined based on image interpretation and was then recorded for comparison with the results obtained from the classified maps.

Table 1. Logic used to infer land-cover map in 1972 based on the classified 1988 land-cover map and the change maps between 1972 and 1988.

Classes in 1988 map	Classes in change map	Inferred classes in 1972 map
Non-forest	disturbance	forest
	unchanged	non-forest
	regeneration	non-forest
Forest	disturbance	forest
	unchanged	forest
	regeneration	non-forest

Landscape pattern analysis

An elevation map (1:1,000,000, Defense Mapping Agency Aerospace Center, St Louis Air Force Station, MO, USA) for the study area was digitized using ARC/INFO GIS software (Environmental Systems Research Institute, Inc., Redlands, CA, USA). The vector file was then converted to a raster file for image analysis using TOPOGRID. Four elevation zones, defined as: 1) <= 1,000 m, 2) 1,001–1,500 m, 3) 1,501–2,000 m, and 4) > 2,000 m, were used to examine the relationship between net annual loss rate of forest cover and elevation in the study area.

Boundary layers for the Changbai Mountain Reserve and the border between China and North

Korea were digitized based on a land-use map of that region with a scale of 1:500,000 (G.H. Jing 1995, personal communication).

Patch structures for both non-forest and forest covers were examined using FRAGSTATS – a spatial pattern analysis program for quantifying landscape structure (McGarigal and Marks 1995). Landscape patterns were evaluated in terms of three landscape indices: 1) edge density (ED, m/ha) where an edge is defined as the interface between a patch of forest and non-forest, and ED equals the sum of the lengths (m) of all edge segments divided by the total landscape area (ha); 2) patch density (PD, number per 100 ha); and 3) mean patch size (MPS, ha). Changes for these indices between 1972 and 1988 were calculated as the values in 1988 relative to the value of the index in 1972. Patch structure was also analyzed for interior forest, defined as the amount of forest remaining after designating a 100-m edge zone. The width of the edge zone is adequate for landscape study in the Changbai Mountain area because the average height of the dominant species (Korean Pine) in the area is about 30 m. Only those patches with size >= 1 ha were used in patch structure analyses after a rule-based merging algorithm was performed to eliminate the ‘salt and pepper’ effect (Ma 1995).

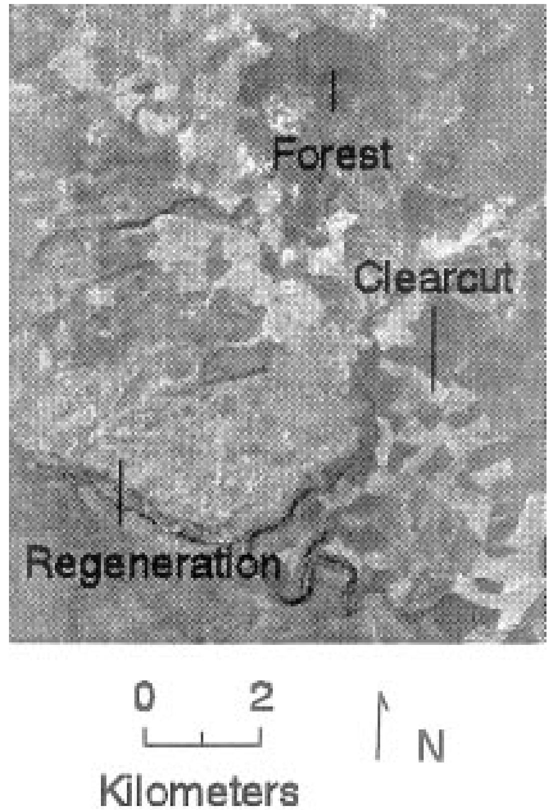


Fig. 4. Unclassified 1988 raw satellite imagery for the same area indicated in Fig. 5d. The red, green and blue colors were assigned to channels 4, 3, 2, respectively.

Results and discussion

Accuracy of classification

Based on the comparison with 345 visually interpreted points from each image, the overall classification accuracies were estimated to be 81.9% and 91.8% on the 1988 and 1972 land-cover maps, respectively. Standard errors obtained from the accuracy assessment performed at 345 sampling points were 1.3% for forest cover and 5.6% for non-forest cover in 1972; and 2.0% and 6.5% in 1988, respectively. Comparison of a subarea of the unclassified imagery (Fig. 4) to the classified map (Fig. 5) shows good agreement.

Application of both the brightness and greenness differences provides a more reliable approach for identifying areas that have undergone change. Although the brightness axis accounted for almost 60% of the original spectral variation in TM data (Cohen *et al.* 1995), it was sensitive to texture and moisture of soils, as well as soil color (Crist *et al.* 1986). Greenness is a contrast between near-infrared and visible reflectance, and is thus a measure of the density of green vegetation. For example, when fire is used for site preparation following a timber harvest, brightness differences may be minimal yet greenness differences will be substantial.

The purpose of this study was to detect major changes in land cover over a 16-year period rather than to develop a detailed classification of the vegetation. Our ability to detect these major changes is not likely to have been significantly affected by the relatively minor differences in spectral reflectance caused by atmospheric effects, differences in the sensors used to obtain the images, and differences in vegetation seasonality among years. Each

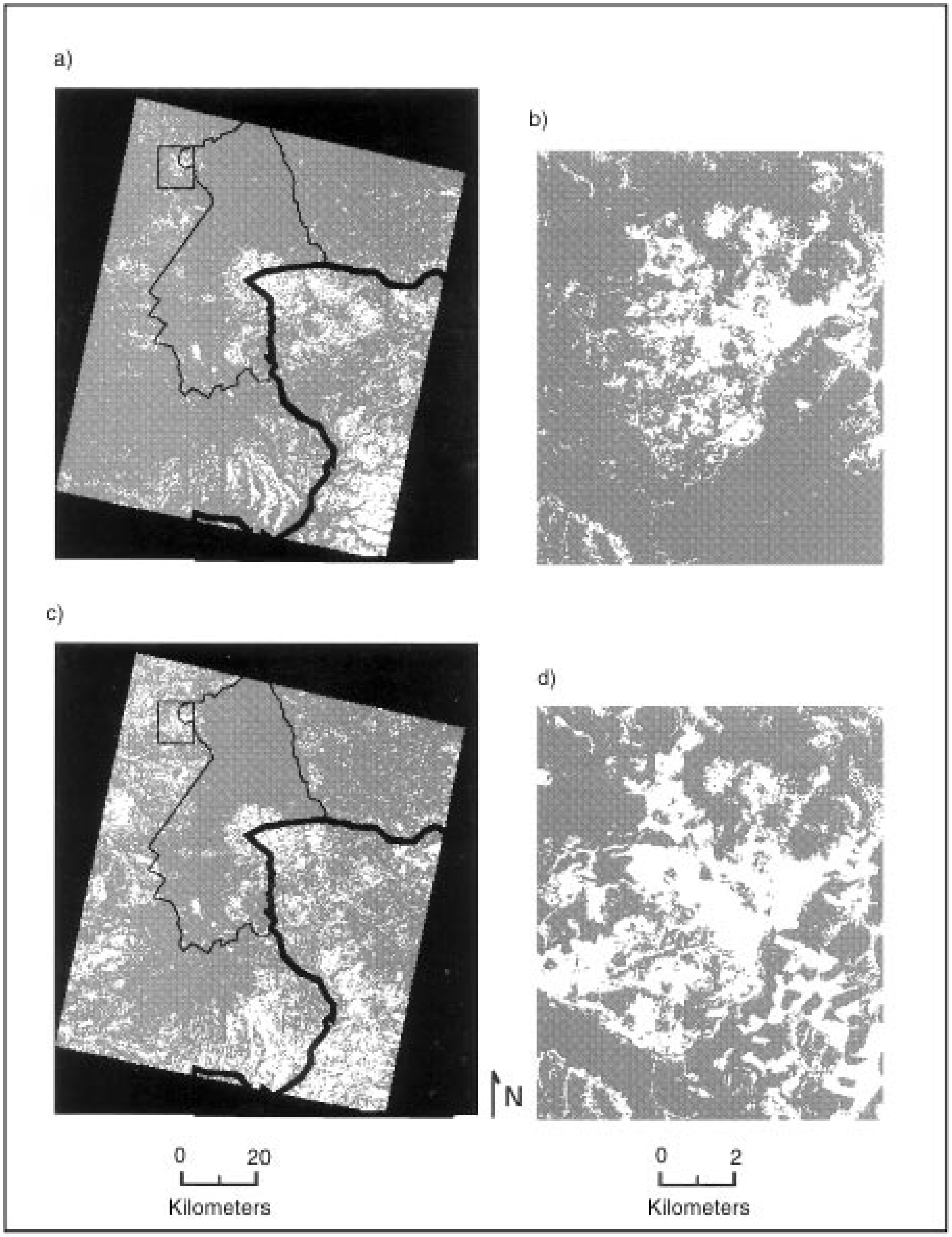


Fig. 5. Forest (in gray) and non-forest (in white) cover maps developed from the 1972 (panels a and b) and 1988 (panels c and d) satellite imagery. Panels b and d show a blow-up of a subset of panels a and c (rectangular line). International and Reserve boundaries are shown in thick and thin lines, respectively.

of these factors become increasingly important sources of error as the number of vegetation class identified in the classification increases. In our study, we distinguish only two land cover types: forest and non-forest. Furthermore, differences in seasonality between years is more likely to be a significant source of error in areas dominated by deciduous vegetation and 84% of our study area is above 1100 m in elevation, a zone in which vegetation is dominated by coniferous forests (Yang *et al.* 1987). Hall *et al.* (1991) used satellite imagery in 1973 and 1983 to quantify changes in vegetation cover in northern Minnesota, USA. They describe an approach to radiometric rectification that can be used to minimize the effects of atmospheric conditions, seasonality and sensor differences. Their study area is dominated by deciduous vegetation and their classification identified six different cover types. Given the more modest objective of our classification and the dominance by coniferous vegetation, we felt that this detailed radiometric rectification was not necessary in our case. Our methods are very similar to those used successfully to detect changes in land cover between 1972 and 1988 in the U.S. Pacific Northwest (W.B. Cohen 1995, personal communication).

Rates and patterns of landscape change

Forest *vs.* non-forest landscape changes

The percentage of forest cover in the entire landscape decreased from 84.4 in 1972 to 74.5 in 1988 – equivalent to an area of about 95,828 ha. Amounts and changes in the properties of the two classes differed strongly among locations. Within the Korean portion of the study area, percentage of forest cover decreased slightly from 65.0% in 1972 to 61.8% in 1988 (Table 2), most of the deforestation occurred along the Yalu River (lower-right portion of the image, Fig. 5c), where this river serves as the international boundary. The smallest decrease in forest cover between 1972 and 1988 occurred inside the Reserve (Table 2). The largest loss of forest cover occurred outside the Reserve in China, from 91.4 to 75.0% (Table 2). Most of these losses were concentrated in the western and southeastern portions of the study area (Fig. 5c). The net annual rate of loss of forest cover averaged 0.73% over the entire area during the study period. The rates varied substantially for different subregions, ranging from 0.04% to 1.12%. The annual loss rate of forest cover was 2.7 times greater in China than in Korea and it was 3.6 times greater outside the Reserve in China than in Korea (Table 2). Furthermore, the net annual loss rate was faster at lower elevations where the terrain was less complex and more easily accessed. The net annual loss rates were 2.0% for the elevation zone below 1000 m, 0.5% for the elevation zone between 1000 to 1500 m, and almost no change for the elevation zone above 1500 (data not shown).

Considerable fragmentation of the landscape has occurred during the 16-year study period. Over the entire study area, the number of patches increased 72% for forest patches and 37% for non-forest patches. Within the Reserve, the number of forest patches increased 6% while it increased 21% for non-forest patches (Table 3). Outside the Reserve, the number of patches increased 310% for forest and 56% for non-forest (Table 3). Within the Korean portion, the trends were the same; the number of patches increased 20% for forest patches and 14% for non-forest patches (Table 3). Within the Reserve, mean forest patch size decreased 6.2% from 729 ha in 1972 to 684 ha in 1988. Outside the Reserve, there was a 5-fold decrease in mean forest patch size from 1888 ha in 1972 to 378 ha in 1988, resulting from a rapid increase in the number of new isolated forest patches and a significant decrease in forest-cover during the 16-year period. In Korea, mean forest patch size decreased 21% from 183 ha to 145 ha (Table 3). The same trends existed for non-forest patches inside the Reserve and in Korea. Within the Reserve, mean non-forest patch size decreased 12% from 30 ha to 26 ha while decreased 3% from 48 ha to 47 ha in the Korean portion. The trends were reversed outside the Reserve; mean non-forest patch size doubled while the number of non-forest patches increased. Although statistical analyses showed that none of these decreases in mean patch size were significant at the level of 0.05, the ecological significance caused by these changes should not be underestimated.

Table 2. Percentage of area in non-forest (NF) and forest (F) in 1972 and 1988 and net change in F for different parts of the study area. Right 3 columns indicate area (ha) undergoing disturbance (F-NF) and regeneration (NF-F) and the ratio.

	Total	% in 1972		% in 1988		Net change in F			Total ha changed		
Parts	Area (ha)	NF	F	NF	F	ha	%	annu. %	F-NF	NF-F	Ratio
Entire scene	967847	15.6	84.4	25.5	74.5	–95828	–11.7	–0.73	121444	25616	4.74
Korea	245767	35.0	65.0	38.2	61.8	–7890	–4.9	–0.31	28311	20421	1.39
China	722080	9.0	91.0	21.2	78.8	–87938	–13.4	–0.84	93133	5195	17.93
China, Reserve	191208	9.9	90.1	10.6	89.4	–1206	–0.7	–0.04	3463	2256	1.54
Non-Reserve	530872	8.6	91.4	25.0	75.0	–86731	–17.9	–1.12	89670	2939	30.51

Table 3. Number of patches, average patch size for different subregions in the study area.

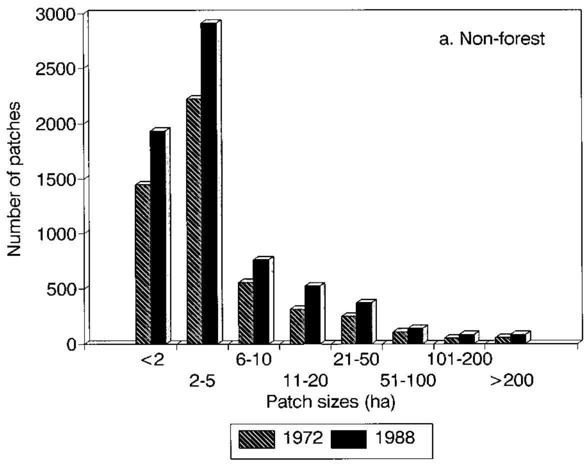
	Number of patches						Average patch size (ha)					
	Inside Res.		Outside Res.		Korea		Inside Res.		Outside Res.		Korea	
	1972	1988	1972	1988	1972	1988	1972	1988	1972	1988	1972	1988
Forest	238	252	260	1066	884	1058	728.8	683.5	1888	377.5	182.7	144.7
Nonforest	626	755	2703	4214	1739	1975	29.6	26.2	14.9	30.5	48.3	46.8

Landscape patterns

The dominant landscape pattern in the area was a continuous matrix of forests with isolated patches of forest and non-forest concentrated outside the Reserve. The modal patch size for both forest and non-forest categories was in the 2–5 ha range (Fig. 6). In 1972, 98.4% of the forest patches were <= 50 ha in size but these patches accounted for only 0.73% of the forest area. A single forest patch accounted for 98.9% of the total forest cover in 1972. By 1988, 1.5% of the forest cover was in patches <= 50 ha while the single largest forest patch accounted for 95.4% of the forest cover. In 1972, more than 73% of the non-forest patches were <= 5 ha in size and these patches accounted for less than 5.5% of the total non-forest area. By 1988, these figures had declined to 70.3% and 4.3%, respectively, suggesting a general trend of coalescence among non-forest patches.

The sizes of clearcut areas varied during the 16-year period within the Chinese portion of the study area. This variation resulted from different management policies. Before 1980, most timber harvests involved the use of clearcuts in which almost everything in a given site was taken away and cutting units were usually greater than 20 ha. After 1980, while clearcuts were still allowed within the study area, there has been a shift towards smaller cutting units (usually < 20 ha). In the mean time, selective cutting was requested by the government. Selective cutting involved the removal, on average, of 40% of the basal area. Selective cutting was conducted for about 30% of the cutting units in the area. These select-cut areas resulted in less dense forests with understory materials left and they may not have been identified as non-forest cover in this study given the broad classification logic we presented above. Most of these timber harvests occurred at the areas close to residential areas, roads, and state-owned forest bureaus (Edit board of Jilin Forests 1988). The area labeled as a clearcut (Fig. 4) represents more recent timber harvests (less than 10 years), while the area labeled as regeneration indicates that the forest was cut in the early 1970s but after 1972.

In our study area, second growth can reach closed canopy condition (> 70% canopy cover) in about 15 years because approximately 75–80% of annual precipitation is concentrated during the growing season (Publishing House of Atlas 1984). Larger non-forest patches were more likely to be permanent open areas and large residential areas.



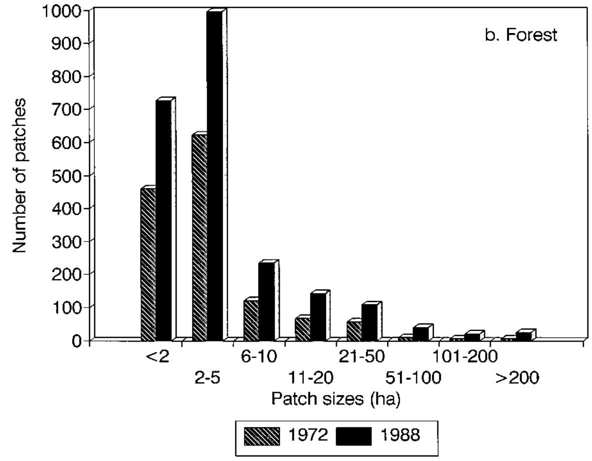


Fig. 6. Patch-size distribution for both forest and non-forest categories between 1972 and 1988.

The coverage of non-forest land increased by 3.2% (35 to 38.2%) in Korea, by 0.7% (9.9 to 10.6%) inside the Reserve, and by 16.4% (8.6 to 25%) outside the Reserve in China (Table 2). Edge density and patch density increased for both non-forest and forest categories during the 16-year period (Fig. 7). The increases mainly resulted from the increasing number of isolated forest and non-forest patches in the areas. The increases in edge density for both categories were similar while the increase in patch density for non-forest cover was less than that of forest cover, indicating coalescence of non-forest patches. This conclusion was also supported by examining the change in mean patch size for both forest and non-forest covers in the study area during the 16-year period. While mean patch size for forest cover decreased 48% in 1988, compared to 1972, mean patch size for non-forest cover increased 24% (Fig. 7). Results from this study quantitatively demonstrate that overall landscape patterns in 1988 were more complex, more irregular, and more fragmented than in 1972. Deforestation activities occurring outside the Reserve was the major cause of these changes. Fragmentation of the forest category was even more evident. These changes will have a significant impact on composition of flora and fauna in the area if the trend continues.

Disturbance *vs.* regeneration

The estimated loss of forest cover between 1972 and 1988 over the entire area (95,828 ha) was lower than the actual disturbance rate (121,444 ha), because 25,616 ha of land (the difference) changed from non-forest to forest cover during the period (Table 2). Disturbance results in sudden changes in the spectral reflectance of a stand and these changes are readily detectable in a series of satellite images. Regeneration ultimately results in a reversal of these spectral reflectance changes as the stand undergoes succession but this occurs relatively slowly.

The ratio of disturbance to succession in different portions of the study area may be used to infer the timing of timber harvesting. In Korea, the relatively low disturbance to regeneration ratio (1.39) suggests that a great deal of timber harvesting took place in the Korean portion of the study area prior to 1972. In contrast, the very high ratio of disturbance to regeneration (30.51) outside the Reserve in China suggests that most timber harvesting has occurred since 1972 (Table 2). It should be pointed out that some of the areas classified as non-forest do not necessarily represent human disturbances but may be caused by natural disturbances. These two disturbances are not distinguished in this study.

Comparison with other landscape change studies

Deforestation is a world-wide issue for many dif-

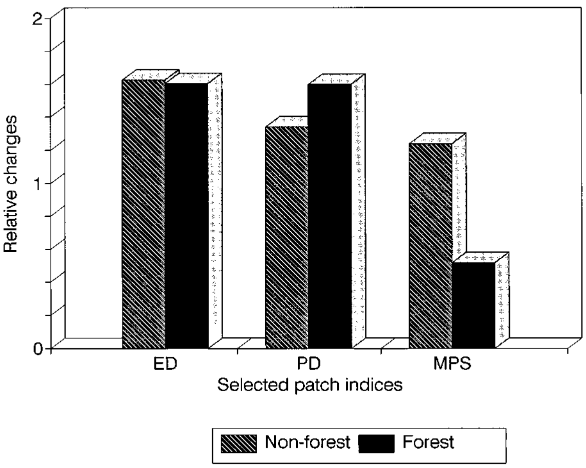


Fig. 7. Percent change in three selected landscape indices for non-forest and forest categories between 1972 and 1988. ED = edge density, PD = patch density, and MPS = mean patch size. Change is calculated relative to the value of the index in 1972 (i.e., 1 indicates no change).

ferent forest ecosystems. Comparable studies of forest cover change have been reported in the other areas. Overall, the change rate of forest cover in our study area was lower than that reported for the Central Cascade Mountains in Oregon, USA, during the same period (Spies *et al.* 1994). The trends in term of annual net loss rates of forest cover were similar between the two studies. In the Oregon study, the rates of change in conifer forest varied from –0.07% per year in wilderness areas to –2.81% on private land with an average of –1.16%. Our results showed that annual loss rates of forest cover were –0.04% inside the Reserve, and –1.12% outside the Reserve during the 16-year period with an average of –0.73% for the entire study area.

The estimated annual disturbance rate of 1.16%, outside the Reserve, was much less than the average cutting rate of 3.1% that has been reported for Jilin Province as a whole for the late 1980s (Tao 1987). The annual rates of major disturbance (change to 'Clearings', 'Regenerating', or 'Broadleaf' types) in conifer and mixed conifer-deciduous types on nonwilderness land in Minnesota, USA were 1.8 and 2.7% (Hall *et al.* 1991). Three watersheds were used to examine relationships between land ownership and land cover change in the Southern Appalachian highlands, NC, USA, and the Olympic Peninsula, WA, USA (Turner *et al.* 1994). Their results showed that net annual loss rates of forest cover on private land in WA (averaging 1.41% for 2 watersheds) was about 5 times faster than that on public land (0.28%) during a 16-year period from 1975 to 1991. In NC, the annual loss rate of forest cover on private land (0.006%) was slightly slower than in public land (0.013%). Other investigators have demonstrated that the average annual deforestation rate in tropical forest area worldwide was 0.06% in the later 1970s (FAO 1981) and 1.42% in the later 1980s (WRI 1990). Rain forests of Madagascar experienced an annual loss rate of 1.80% during 1950–85 (Green and Sussman 1990). Annual rate of forest loss in Mato Grosso, Brazil was 0.25% during 1981–84 (Nelson *et al.* 1987). In the Brazilian Amazon basin, deforestation rates calculated from satellite data were as high as 1.9% in 1978 and 5.6% in 1988 (Skole and Tucker 1993).

Conclusions

Rates and patterns of landscape change from forest cover to non-forest cover during a 16-year period (1972–88) in the Changbai Mountain area can be quantified from satellite images. Differences in land-use history and management policy have resulted in substantial differences in these parameters observed between China and Korea and especially between the Reserve and non-Reserve lands in China. Results from this study provide base line information that can be used to develop better resource management strategies in the region and for determination of future landscape changes.

The results from our simplified analysis may not be adequate for comprehensive ecosystem management because biological and environmental variation was not considered. However, these observations provide considerable information on the recent rates and major patterns of forest landscape dynamics in this area and have several management implications.

First, the minimal changes from forest to non-forest cover within the Reserve indicate that the policy of preventing human disturbance in the

Changbai Reserve has been very successful. However, from the perspective of maintaining biodiversity, decision-makers may need to consider expansion of the Reserve boundary to include more lower elevation areas where substantial loss of forest cover was observed. There are two reasons for doing this: 1) many unique species of animal and plants can be found below an elevation of 1100 m in the study area (Yang *et al.* 1987); 2) the lands with elevations < 1,100 m occupy only 16% of the total area within the current Reserve boundary.

Second, highly fragmented forest areas are observed at low elevation outside the Reserve in the 1988 scene, especially in the western portion of the image. While these highly fragmented forest areas may be considered ecologically less valuable now, many are in highly productive, species-rich sites that have considerable potential for restoration. A restoration strategy may include: 1) increasing the productivity and stability of the ecosystems; and 2) continuing reasonable utilization of forest resources while maintaining the biodiversity of the ecosystems. The strategy would require regrowth of cutover areas, delaying or excluding cutting in some areas and concentrating it in other areas.

Third, large remnant forest areas south of the Reserve and northeast of the Reserve might disappear in the next 20 years if the current rate of disturbance outside the Reserve is not slowed. Another alternative is to speed up the afforestation rate in this area, especially in the Chinese portion outside the Reserve.

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