GEOGRAPHIC INFORMATION ANALYSIS: AN ECOLOGICAL APPROACH FOR THE MANAGEMENT OF WILDLIFE ON THE FOREST LANDSCAPE

FINAL TECHNICAL REPORT

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ABSTRACT

This paper is a summary of our project which was funded by NASA grant NAGW-1460 as part of the Earth Observation Commercialization/Applications Program (EOCAP) directed by NASA's Earth Science and Applications Division. The goal of our project was to work with several agencies to focus on forest structure and landscape characterizations for wildlife habitat applications. We used new analysis techniques in remote sensing and landscape ecology with geographic information systems (GIS). The development of GIS and the emergence of the discipline of landscape ecology provided us with an opportunity to study forest and wildlife habitat resources from a new prospective. We developed new techniques to measure forest structure across scales from the canopy to the regional level. This paper describes our project team, technical advances, and finally the technology adoption process that we used with our cooperating end-user organizations. Reprints of refereed journal articles relating to this grant are located in the Appendix.

THE PROJECT TEAM

Scientists from the Environmental Remote Sensing Applications Laboratory (ERSAL) in Forest Resources at Oregon State University submitted a proposal with collaborators in response to NASA Research Announcement NRA-87-OSSA-6.*

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*ERSAL was organized in 1972 under the auspices of NASA to 1) establish and maintain communication between the scientific community engaged in research of remote sensing technology and potential users of modern remote sensing capability and 2) engage in projects that would demonstrate the usefulness of the technology and serve to close the gap between the scientific community and the potential beneficiaries. Through the years ERSAL has developed numerous applications for Landsat data and has amassed a substantial body of experience in the monitoring of natural resources.

William J. Ripple served as principal investigator for this project and was responsible for direct supervision of the project work and coordinated the efforts involving the interactions among the team members and their corresponding organizations. Other ERSAL staff members involved in this project were Dennis L. Isaacson, project coinvestigator, along with Gay A. Bradshaw and Maria Fiorella, both graduate research assistants. Three end-user organizations participated in the project including the Pacific Northwest Research Station of the United States Forest Service (USFS), the Oregon Department of Fish and Wildlife (ODFW), and the Oregon Cooperative Wildlife Research unit which is located at Oregon State University and is administered by the United States Fish and Wildlife Service (USFW). Dr. Thomas A. Spies served as a coinvestigator from the U. S. Forest Service. He is a research forester and works in an applied fashion to help bring current forest research results into management practices. In 1989, soon after our project started, he hired a full time remote sensing specialist, Dr. Warren Cohen. The project coinvestigator from ODFW was Mr. Donavin A. Leckenby an experienced wildlife biologist with a longtime interest in remote sensing. Others from ODFW involved in the project were Larry Bright, research administrator, along with Tom O'Nell, and Pricilla Coe, wildlife biologists. Our main cooperator from the USFW was Dr. E. Charles Meslow, a wildlife biologist specializing in the ecology of the spotted owl. Others involved from his unit included Gary Miller, Dave Johnson, and Kit Hershey.

TECHNICAL ADVANCES

Our team has made a number of technical advances in measuring forest structure and habitat across a variety of scales. We have published numerous articles relating to these advances. We have described canopy structure, stand structure, and landscape structure for both small and large landscapes and at the regional scale. New information was developed on the relationships of forest landscape patterns with spotted owl habitat and measures of biodiversity such as vertebrate species richness.

At the canopy level, the USFS digitized aerial videography to evaluate the canopy structure of Douglas-fir forests (Cohen et al., 1990). Forest age and stand structure were estimated from both SPOT textural variables and Thematic Mapper spectral variables using new and innovative satellite data analysis techniques (Cohen and Spies, 1992). Ripple and others (1991c) compared Thematic Mapper to SPOT data for estimating forest volume. Fiorella and Ripple (1993a) found that by using a "structural index" with Thematic Mapper data, old growth forests could be distinguished from mature and young forests. We also demonstrated how Landsat Thematic Mapper data can be used to determine conifer reforestation success (Fiorella and Ripple, 1993b).

At the landscape scale, techniques were developed to measure forest landscape patterns important to wildlife species. A GIS fragmentation index (GISfrag) provided a new automated method for determining the extent of forest fragmentation (Ripple et al. 1991a). Rates and patterns of change of conifer forests between 1972 and 1988 were characterized using landsat imagery in Western Oregon (Spies et al., 1994). Measures of forest fragmentation (Ripple, 1994) were extended to the regional scale for planning a Pacific Northwest biological reserve system for Congress using AVHRR satellite data (Forest Ecosystem Management Assessment Team 1993). We also investigated how the amount of old-growth and mature forest influences the selection of nest sites by the northern spotted owl (Ripple et al., 1991b). In a biodiversity application, we developed a predictive model of wildlife species richness using a GIS (Fiorella 1992). As a spinoff, we were able to improve our understanding digital elevation models (DEMs) by comparing the results from both 7.5 minute and 1 degree DEMs (Isaacson and Ripple, 1990). A Ph.D thesis by Gay A. Bradshaw (1991) and a M.S. thesis by Maria Fiorella (1992) were completed during the course of our grant.

TECHNOLOGY ADOPTION PROCESS

A traditional view for adoption of technology is typically viewed as a series of stages through which adopters progress. A general model was presented by Rodgers and Shoemaker (1971):

1) Awareness - the potential adopter learns of a new idea;

2) Interest - the potential adopter develops an interest in the idea and gathers information;
3) Evaluation - the potential adopter mentally applies the innovation to his situation;
4) Trial - the potential adopter applies the idea in order to determine its utility; and
5) Adoption - the idea is put into regular use.

Our plan for transfer of technology in this project was adopted from the foregoing model. Since we have three different end-user organizations, the way in which adoption occurred varied.

In the case of the USFS, awareness, interest, and evaluation were already in place when the project started. The USFS had a great need to determine forest structural characteristics, landscape patterns, and old growth acreages because of issues concerning old-growth forest management and the spotted owl was about to be listed as a threatened species. The trial sage was short and adoption was initiated in 1989 with the hiring of Dr. Warren Cohen as a remote sensing specialist. The USFS established image processing facilities and hired both Gay Bradshaw and Maria Fiorella after they finished their graduate programs at ERSAL. The USFS has ongoing remote sensing projects and seems committed to this technology as it appears to be the only approach suitable for answering questions about these Northwest forests at the landscape scale.

The ODFW has had a long history evaluating Landsat satellite data for elk habitat analysis. In the past, ODFW contracted out for satellite remote sensing services. During this project, team members assisted ODFW in acquiring two ERDAS image processing software systems for use on their existing microcomputers. ERSAL staff conducted several demonstration projects, numerous informal information sessions, and two short courses for ODFW staff on using ERDAS for classifying deer, elk, and spotted owl habitat. ODFW staff are currently in the adoption phase and they continue to contact ERSAL team members for advice and technical support.

The initial trail period for the USFW included a landscape assessment of old growth habitat near spotted owl nest sites. After this initial work was complete, USFW adoption took the form of funding a multiyear (1992-1997) cooperative agreement with ERSAL to study landscape patterns associated with spotted owl nests using remote sensing and GIS. The USFW does not have in-house capabilities but is committed to the adoption of technology through providing funding and ongoing collaboration. The laboratory facilities in ERSAL are currently being used for this project.

In summary, we attributed the success of the technology adoption process to the following factors:

1) The end users were very aware of and interested in the technology.
2) The end users consisted of highly motivated productive individuals that consistently mentally applied the innovations to their own situations.
3) The end users had a major need for the technology and the resulting spatial information. This need grew out of issues relating to old growth and spotted owl management.
4) Economic resources were available for adoption of the technology.

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Appendix

Project Technical Articles

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Comparison of 7.5-Minute and 1-Degree Digital Elevation Models

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ABSTRACT: We compared two digital elevation models (DEMs) for the Echo Mountain SE quadrangle in the Cascade Mountains of Oregon. Comparisons were made between 7.5-minute (1:24,000-scale) and 1-degree (1:250,000-scale) images using the variables of elevation, slope aspect, and slope gradient. Both visual and statistical differences are presented.

INTRODUCTION

THE METHODS REQUIRED to assess the extent and distribution of natural resources often require that topographic factors be taken into account. This may be because these factors are directly influencing the resources of interest (vegetation patterns), or because topographic factors influence the data from Earth resources satellites.

In forested environments, for example, slope gradient and slope aspect are known to influence distribution of various plant species. The shading resulting from various slope and aspect combinations along with solar angle of incidence also influences the appearance of forest types on remotely sensed imagery. By accounting for topographic variables such as slope gradient, slope aspect, and elevation, we should be able to improve our capacity to conduct inventories of natural resources. Eby's (1987) study of the effect of sun incidence angle data, calculated from digital elevation data, is an example in the use of digital Landsat data. Stoszek (1977) and Miller and Heller (1978) both determined that topographic variables—including elevation, aspect and locations near ridgetops—could be used to predict infestations by the Douglas-fir tussock moth.

With Geographic Information Systems (GIS), topographic variables such as these can be developed from Digital Elevation Models (DEM) available in digital formats from the United States Geological Survey (USGS). Greenlee (1987) developed a method to extract drainage networks and other rasterized linework from a DEM for input into a vector GIS. Jenson and Dominique (1988) reported that DEMs can be useful for hydrologic applications such as delineating watersheds and developing overland flow path models. Their results indicated that the accuracy and detail of the hydrologic information was dependent on the resolution and quality of the DEM.

Two commonly used DEM formats are 1:250,000-scale (1-degree) Defense Mapping Agency (DMA) data and 1:24,000-scale (7.5-minute) DEM data. The DMA data are available for the whole of the conterminous United States, but DEM data are available only for selected areas where the USGS has collected data for developing and updating their 7.5-minute quadrangle sheets (U.S. Geological Survey, 1987). The 7.5-minute DEM data are more highly resolved (30 by 30 m per cell) than the 1-degree DMA data, at three arc-seconds (about 65 by 92 m for this mid-latitude area), but its limited availability invites the question of whether the resolution and accuracy of the DMA data, which is available over the entire United States, is adequate for inclusion with large area surveys incorporating satellite-acquired data.

We compared these two data forms on a USGS quadrangle-sized area in the Oregon Cascade Mountains. The study area chosen was the area covered by the Echo Mountain SE quadrangle (also referred to as Tamolitch Falls) within the McKenzie River drainage east of Eugene, Oregon. The western two-thirds of the quadrangle is steep and deeply dissected, and the eastern portion is flatter. This area is typical of many forested mountains of the Pacific Northwest. The objective was to quantify the differences between slope gradient, slope aspect, and elevation images derived from the two digital elevation data forms.

METHODS

The 7.5-minute data for the Echo Mountain SE quadrangle were read as elevations representing distance above mean sea level (MSL) in one-metre intervals with a minimum of 581 m and a maximum of 1747 m. The data were displayed as a level-sliced color image, with red colors as higher elevations and magenta for lower elevations (Plate 1).

Derived images of slope gradient and slope aspect were computed on the original 16-bit data (L.N. Brantley, personal comm.; see the Appendix for algorithms used). Eight sector aspects and a flat category were computed, and slopes were calculated in percent. Level-slicing was used to develop a color scheme for image representation of these two data sets.

The 1-degree data, a full scene of DMA data covering the area within 122 and 123 degrees west longitude and 44 and 45 north latitude, were read from tape to a disk file. A 151- by 151-pixel subset of the data covering the same area as in the 7.5-minute data was selected from the full scene with a 601-m minimum and a 1707-m maximum elevation.

Eighty observations, one from each of the two different elevation data sets, were obtained from a systematic sampling of the study area at points with a 1000-m spacing. A simple linear regression analysis was conducted to determine the relationship between the 80 elevations from the 7.5-minute data set and the 1-degree data set.

For the 1-degree data, slope gradient and slope aspect images were derived, using the same methods described above *before* they were rescaled, rectified, and resampled to 30- by 30-m pixels. The resampling was conducted using a nearest neighbor routine which used the intensity of the closest pixel to determine the value of the output pixel. Slope gradient and slope aspect images were colored using the same scheme as with the 7.5-minute data.

A difference image was generated by subtracting elevations at each pixel location for the whole image area. Comparisons for both slope aspect and slope gradient variables were accomplished with an overlay analysis to generate a matrix of observations showing frequencies of coincidence of all classes.

RESULTS AND DISCUSSION

The relationship between the elevations sampled from the 7.5-minute and the 1-degree data sets (before resampling) is

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0099-1112/90/5611-1523$03.00/0

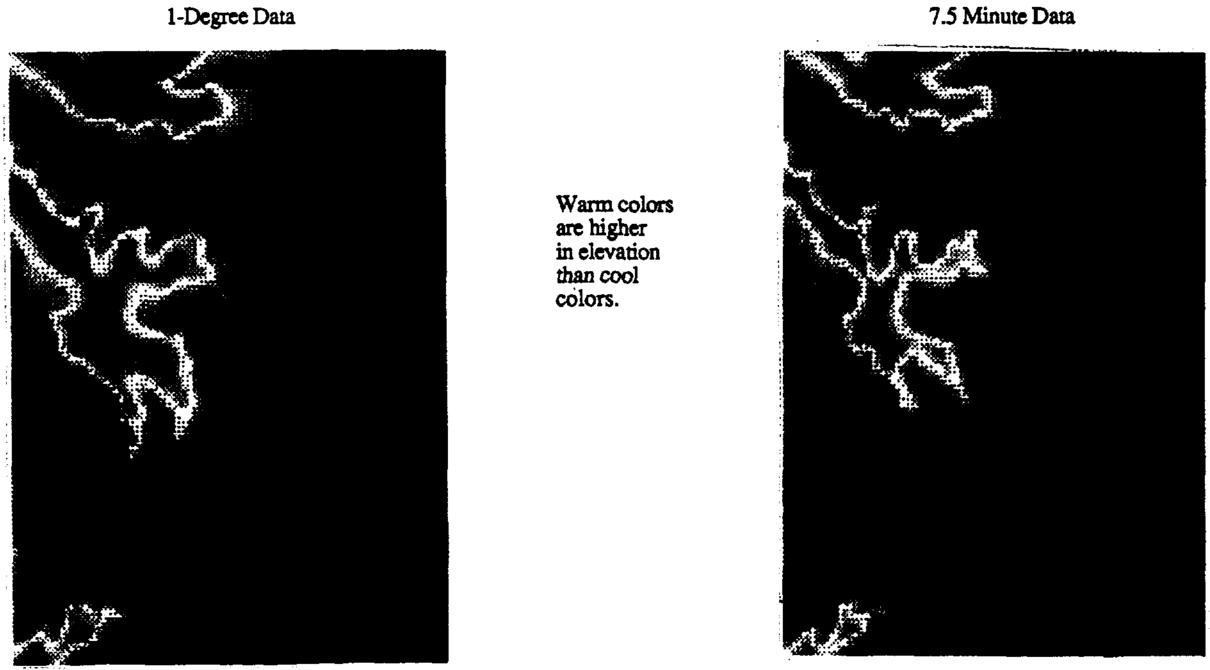

PLATE 1. Comparison images of 7.5-minute and 1-degree digital elevation models for the variable of elevation for the Echo Mountain SE Oregon Quadrangle.

shown in Figure 1. The relationship between these variables was found to be linear with a coefficient of determination of 84.3 percent. The slope, 0.99, and intercept, 36.9, were not significantly different from 1.0 and 0.0, respectively ($P<0.01$), indicating that there was no significant bias. The standard error was 30.6 m, and the residuals from the least-squares regression line appeared to be distributed randomly. By overlaying the elevation layers from the two different data sources, the mean elevation difference, on a pixel by pixel basis for the entire area, was calculated to be 31.0 m, with a standard deviation of 6.99 m. Most of the higher differences appeared to be associated with steeper slopes. A visual comparison of the 1-degree and 7.5-minute elevation maps suggest considerable correspondence of general patterns, but a greater amount of detail in the 7.5-minute image (Plate 1).

The visual correspondence of general pattern between the 1-degree and the 7.5-minute data sets was also apparent for both slope aspect and slope gradient images (Plates 2 and 3). Pattern heterogeneity was higher in the 7.5-minute data sets, probably due to the finer original pixel resolution. While the modes for aspect class comparisons occurred as expected (East with East, Southeast with Southeast, etc.) only 36 percent of pixels were in the same class in both images (Table 1). Sixty-seven percent of pixels were in the same or adjacent (East with Northeast, East and Southeast, e.g.) classes. In the slope gradient images, 29 percent of all pixels were in the same class in both images (Table 2), and gradients calculated from the 1-degree data were lower than those from the 7.5-minute data. The differences apparent in the images seemed to derive from a greater level of detail in the 7.5-minute than in the 1-degree data.

While we have noted anomalies in 1-degree data in other areas, of low relief, which create obvious artifacts in derived topographic images, no instance of this was found in the data used in this comparison in areas of mountainous terrain. This is true also of other 1-degree data sets we have used. For applications in mountainous terrain, 1-degree data seem an appropriate match for Landsat Multispectral Scanner data and for databases where data storage limitations may be of concern.

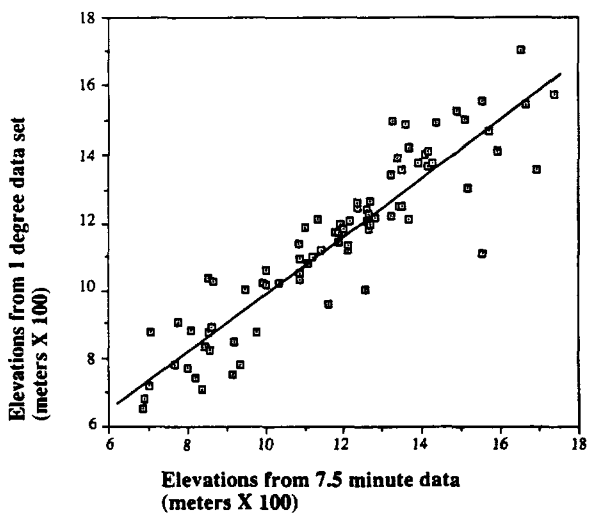

FIG. 1. Scatter diagram of elevations obtained from a 7.5-minute digital elevation model (x axis) and a 1-degree digital terrain model (y axis) on the Echo Mountain SE quadrangle area of the central Oregon Cascade mountains.

One final note has to do with the order of processing for digital terrain data sets. Resampling to adjust pixel size, rectification of images, and rescaling to different intervals resulted in different final image values depending on the order of processing. In some cases the differences were great. In this set of analyses, for example, we retained the 16-bit format as far along in the chain of processing as possible, and final images for slope and aspect yielded very different quantitative estimates of flat terrain. Because the original data are in one-metre intervals, and the algorithm for slope aspect determines a non-flat aspect for *any* unequal elevation, very few slope aspect "flat"

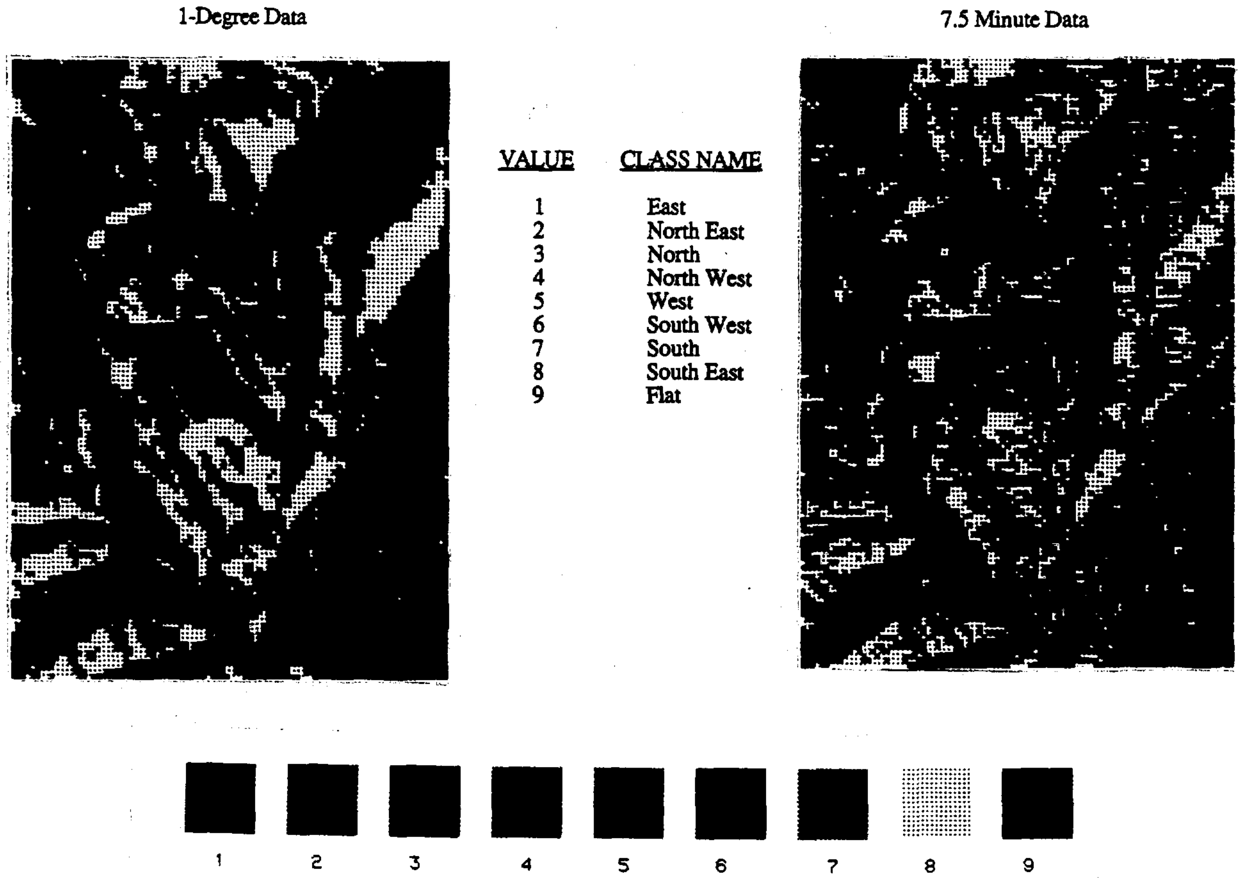

PLATE 2. Comparison images of 7.5-minute and 1-degree digital elevation models for the variable of slope aspect for Echo Mountain SE Oregon quadrangle.

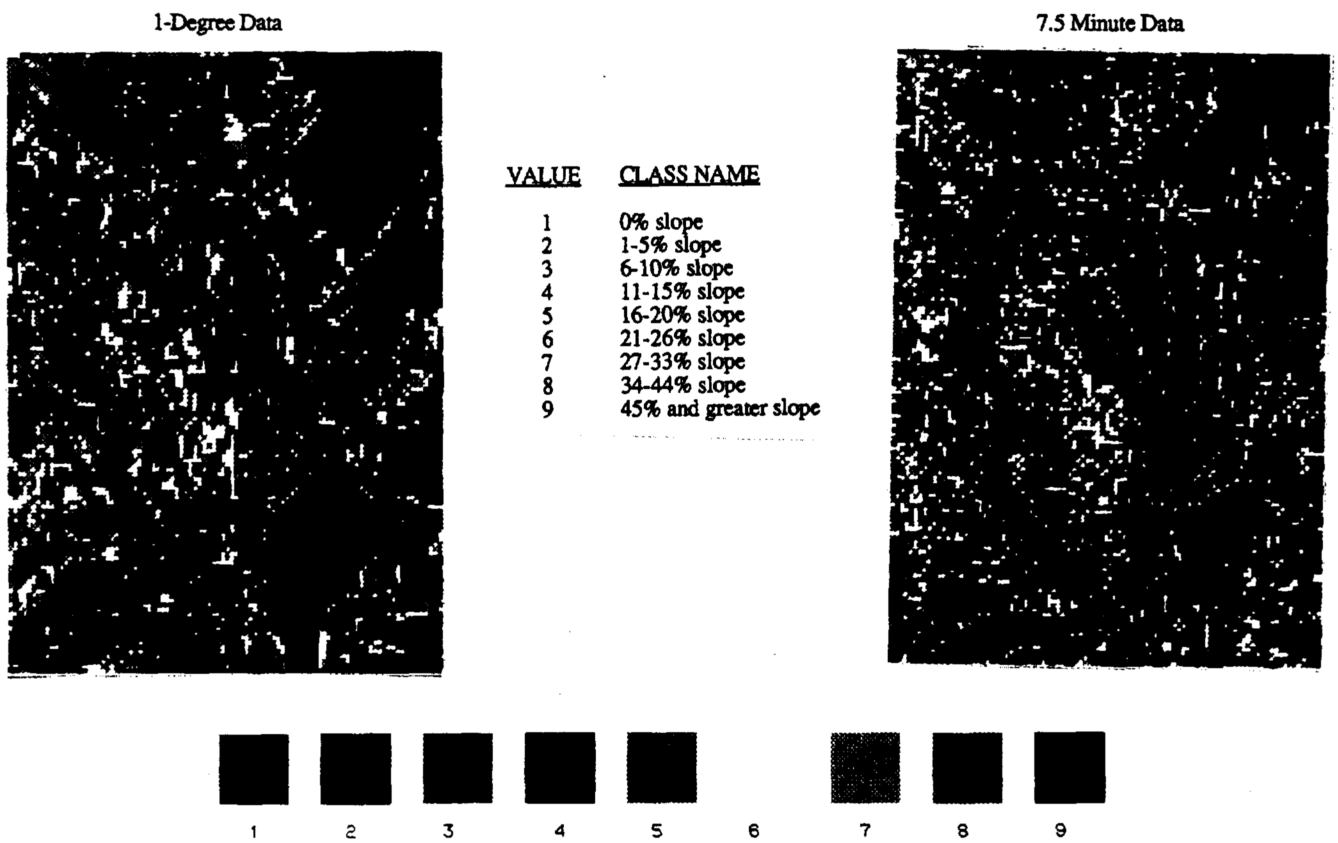

PLATE 3. Comparison images of 7.5-minute and 1-degree digital elevation models for the variable of slope gradient for Echo Mountain SE Oregon quadrangle.

TABLE 1. A COMPARISON OF SLOPE ASPECTS DERIVED FROM 7.5-MINUTE AND 1-DEGREE DIGITAL ELEVATION MODELS (DEM). NUMBERS ARE COUNTS OF COINCIDENT OCCURRENCE FOR THE ENTIRE ECHO MOUNTAIN SE QUADRANGLE.

1-degree DEM Aspects	7.5-minute DEM Aspects									
	E	NE	N	NW	W	SW	S	SE	Flat	TOTAL
E	8030	2689	686	497	524	651	888	2777	10	17403
NE	4886	7667	3191	808	744	637	556	1375	25	20526
N	1516	2905	7285	2557	927	470	462	799	17	17408
NW	576	569	1805	4921	2980	833	689	498	35	13739
W	470	241	728	2573	6383	2710	1236	525	254	17730
SW	1026	565	392	1027	4561	9924	3130	1131	28	31708
S	1518	384	188	437	1378	4821	6635	3337	4	23523
SE	5697	1087	426	480	788	1337	3340	9567	12	24071
FLAT	630	463	398	340	422	361	615	531	40	4161
TOTAL	24349	16570	15099	13640	18607	21744	17551	20540	425	170269
AGREEMENT	33%	46%	48%	36%	34%	46%	38%	47%	9%	36%

TABLE 2. A COMPARISON OF SLOPE GRADIENT CLASSES DERIVED FROM 7.5-MINUTE AND 1-DEGREE DIGITAL ELEVATION MODELS (DEM). NUMBERS ARE COUNTS OF COINCIDENT OCCURRENCE OVER THE ENTIRE ECHO MOUNTAIN SE QUADRANGLE

1-degree DEM (percent)	7.5-minute DEM slopes (in percent)									
	0	1-5	6-10	11-15	16-20	21-26	27-33	34-44	>44	TOTAL
0	2314	2313	1798	1225	810	630	440	507	678	10715
1-5	565	3026	2967	1922	1201	808	708	845	1459	13501
6-10	398	2538	3775	2842	1686	1206	911	912	1346	15614
11-15	297	1545	2310	2184	1746	1612	1219	1079	1330	13322
16-20	217	609	1100	1381	1448	1724	1502	1299	1244	10524
21-26	296	479	901	1301	1683	2581	2670	2400	2564	14875
27-33	240	320	729	1221	1550	2629	3229	3314	3479	16711
34-44	250	243	570	996	1457	2440	3688	5245	6714	21603
>44	222	384	808	1240	1607	2459	3576	8015	21866	40177
TOTAL	4799	11457	14958	14312	13188	16089	17943	23616	40680	157042
AGREEMENT	48%	76%	25%	15%	11%	16%	18%	22%	54%	29%

pixels resulted. In slope gradient determinations, rounding can result in zero percent calls without absolute equality in the elevations used in calculations, and the zero percent classes were greater: 10,715 zero percent slopes *versus* 4161 "flat" aspects in the 1-degree data, and 4,799 zero percent slopes *versus* 425 "flat" aspects in the 7.5-minute data. Users of digital terrain data should carefully consider and check each step in their processing to ensure meaningful results. The methods of production can effect the kinds of artifacts that occur and the accuracy of the results.

We have concluded that, in our applications in wildlife habitat analysis, 1-degree DEM data are preferable to our developing our own digital elevation data for the extensive areas for which 7.5-minute data are not available. We suggest, however, that additional comparative research be conducted involving both types of DEMs for a variety of terrain types. The results reported in this paper may or may not be indicative of DEM differences found in other areas, but the comparison techniques in this paper can be applied to other areas with different topographic profiles.

ACKNOWLEDGMENTS

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APPENDIX

Computations of slope gradient and slopect aspect (source, L.N. Brantley, personal comm.):

consider the 3- by 3-pixel matrix of elevations around the point (x,y) –

	$x-1$	x	$x+1$
$y-1$	a	b	c
y	d	e	f
$y+1$	g	h	i

where a,b,c, etc., are the elevations at the indicated points.

Computation of average x and y elevation changes:

$\Delta x_1 = c - a$ $\quad \Delta y_1 = a - g$

$\Delta x_2 = f - d$ $\quad \Delta y_2 = b - h$

$\Delta x_3 = i - g$ $\quad \Delta y_3 = c - i$

$$\Delta x = \frac{\Delta x_1 + \Delta x_2 + \Delta x_3}{3} \qquad \Delta y = \frac{\Delta y_1 + \Delta y_2 + \Delta y_3}{3}$$

(1) Computation of slope gradient:

let s = cell size

$\Delta e = \sqrt{\Delta x^2 + \Delta y^2}$

if $\Delta e \leq s$, percent slope = $\Delta e / s * 100$

if $\Delta e > s$, percent slope = $100 + [(1 - s/\Delta e) * 100]$

(2) Computation of slope aspect:

if $\Delta x = \emptyset$ and $\Delta y = \emptyset$, aspect = FLAT,

otherwise

$$\theta = \tan^{-1}\left[\frac{\Delta y}{\Delta x}\right]$$

$\frac{-\pi}{8} < \theta < \frac{\pi}{8}$ aspect = E

$\frac{\pi}{8} < \theta < \frac{3\pi}{8}$ aspect = NE

$\frac{3\pi}{8} < \theta < \frac{5\pi}{8}$ aspect = N

$\frac{5\pi}{8} < \theta < \frac{7\pi}{8}$ aspect = NW

$\frac{7\pi}{8} < \theta < \pi$ aspect = W

or $-\pi < \theta < \frac{-7\pi}{8}$ aspect = W

$\frac{-7\pi}{8} < \theta < \frac{-5\pi}{8}$ aspect = SW

$\frac{-5\pi}{8} < \theta < \frac{-3\pi}{8}$ aspect = S

$\frac{-3\pi}{8} < \theta < \frac{-\pi}{8}$ aspect = SE

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Dedicated to the improvement of the quality of life on earth through better use of geographic information.

The International Geographic Information Foundation was originally founded in 1979 by the American Society of Photogrammetry to administer the funding of educational scholarship and awards programs. In 1986, its scope was expanded in response to the need by other professional societies concerned with geographic information for assistance and expertise in raising and administering funding for scholarship, awards, and research programs.

The IGI Foundation exists to:

- Conduct programs in the field of the mapping sciences which provide an avenue for practitioners of the geographic sciences to make tangible contributions from their own knowledge and experience to the advancement and practices of the sciences.
- Extend to other organizations the opportunity to exchange information and mapping experience, and the practical solutions of mapping sciences problems.
- Encourage improvements in the geographic information sciences and their applications by making grants for the conduct of research in the fields of the mapping sciences and establishing awards programs to recognize significant contributions to humanity through the sciences
- Improve public awareness of the benefits available from the geographic information sciences and of the contributions they make to society.

The Foundation aids the societies by assisting in raising funds through charitable contributions and endowments for worthwhile educational, research, and awards programs, and providing expertise in funds management. The Foundation solicits charitable contributions from individuals, corporations, interested organizations, and other foundations. Each society served by the Foundation develops its own programs and performs its own internal selection of scholarship, awards, and research grant recipients.

A variety of contribution options are available:

- Established Fund Awards – Named Awards Programs may be established and maintained perpetually with a minimum donation of $15,000, which may be contributed over a 3-year period, or the fund will be disbursed as a one-time award.
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Measuring Forest Landscape Patterns in the Cascade Range of Oregon, USA

52-43

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ABSTRACT

This paper describes the use of a set of spatial statistics to quantify the landscape pattern caused by the patchwork of clearcuts made over a 15-year period in the western Cascades of Oregon. Fifteen areas were selected at random to represent a diversity of landscape fragmentation patterns. Managed forest stands (patches) were digitized and analysed to produce both tabular and mapped information describing patch size, shape, abundance and spacing, and matrix characteristics of a given area. In addition, a GIS fragmentation index was developed which was found to be sensitive to patch abundance and to the spatial distribution of patches. Use of the GIS-derived index provides an automated method of determining the level of forest fragmentation and can be used to facilitate spatial analysis of the landscape for later coordination with field and remotely sensed data. A comparison of the spatial statistics calculated for the two years indicates an increase in forest fragmentation as characterized by an increase in mean patch abundance and a decrease in interpatch distance, amount of interior natural forest habitat, and

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the GIS fragmentation index. Such statistics capable of quantifying patch shape and spatial distribution may prove important in the evaluation of the changing character of interior and edge habitats for wildlife.

INTRODUCTION

The Douglas-fir *Pseudotsuga menziesii* (Mirb.) Franco forests of western Oregon have been cut extensively during the past 40 years. In these forests, clearcuts, new plantations, and second-growth stands now exist on the landscape formerly dominated by extensive old-growth forests and younger forests resulting from fire disturbance (Spies & Franklin, 1988). Consequently, the landscape has become more spatially heterogeneous. Some of the effects of this newly created landscape on the forest ecosystem are immediately apparent. For example, the amount of old-growth forest habitat for interior species such as the northern spotted owl *Strix occidentalis caurina* is altered with forest harvesting practices. Less well understood are the long-term and more subtle interactive effects on ecosystem processes (e.g. changes in species diversity and abundance, nutrient cycling and primary forest productivity), which may occur as a result of the changed forest mosaic. It is generally accepted that wildlife ecology and behavior may be strongly dependent on the nature and pattern of landscape elements (Forman & Godron, 1986), but few precise measurements relating these changes to landscape spatial alteration have been made. Properties of forested landscapes such as patch size, the amount of edge, the distance between habitat areas, and the connectedness of habitat patches have a direct influence on the flora and fauna (Thomas, 1979; Harris, 1984; Franklin & Forman, 1987; Ripple & Luther, 1987). For these reasons, models and monitoring schemes are urgently needed for prescribing the location, size, and shape of future harvest units and old-growth habitat patches. With proper design, these forest landscapes should be able to achieve desired habitat values and maintain biological diversity (Noss, 1983; Harris, 1984; Franklin & Forman, 1987).

As a first step to studying forest landscape pattern, the spatial character of the landscape must be quantified to relate ecological processes to landscape configuration. Numerous methods and indices have been proposed for these purposes (e.g. Forman & Godron, 1986; Milne, 1988; O'Neill *et al.*, 1988). Landscape pattern can be quantified using statistics in terms of the landscape unit itself (e.g. patch size, shape, abundance, and spacing) as well as the spatial relationship of the patches and matrix comprising the landscape (e.g. nearest-neighbor distance and amount of contiguous matrix). A selection of these measures can therefore describe the several aspects of fragmentation which occur as the result of forest harvesting practices.

Because spatial analyses are often cumbersome due to large amounts of data, it is desirable that the analysis be automated. Furthermore, it should be amenable for study at various scales in concert with a variety of data types. Automated systems such as geographical information systems (GIS) address such needs (Ripple, 1987, 1989). With the development of GIS, the ability to readily measure spatial characteristics of landscapes in conjunction with field and remotely sensed data has become possible.

The overall goal of the present study was to test the feasibility of measuring forest landscape patterns accurately using a set of spatial statistics and a GIS. Specifically, we set out to apply a set of statistics to a series of forested landscapes in the Cascade Range of western Oregon to (1) assess the sensitivity of these statistics to characterize landscape pattern; (2) develop and test a GIS-derived index to measure fragmentation; and (3) quantify the change and types of forest fragmentation through time in our study area. The results of this pilot study are intended to aid the forester, wildlife biologist, and land manager in assessing change in wildlife habitat and alteration in ecosystem processes as a result of forest fragmentation.

STUDY AREA

The study area consisted of the Blue River and Sweet Home ranger districts of the Willamette National Forest. According to Franklin and Dyrness (1973), these ranger districts lie primarily in the western hemlock *Tsuga heterophylla* and pacific silver fir *Abies amabilis* vegetation zone, with the major forest tree species consisting of Douglas-fir, western hemlock, pacific silver fir, noble fir *Abies procera*, and western redcedar *Thuja plicata*. The climate can be characterized as maritime with wet, mild winters and dry, warm summers. Under natural conditions, Douglas-fir is the seral dominant at elevations below 1000 m, where it typically develops nearly pure, even-aged stands after fire. Large areas are covered by old-growth Douglas-fir/western hemlock forests in which Douglas-firs are over 400 years old. Fires during the past 200 years have created a complex mosaic of relatively even-aged natural stands throughout the study area. Superimposed on this natural mosaic is a second component of pattern complexity resulting from timber harvesting over the last 40 years.

METHODS

Data acquisition

Landscapes were classified in terms of managed and natural elements. Managed forest stands (typically young forest plantations of up to approximately 40 years old established after clearcutting) were defined as internally homogeneous units (or managed patches) embedded in a matrix consisting of natural, uncut forest. This two-phase mosaic is a simplified representation of a more complex system in which many patch and matrix types exist. For the present analysis, however, we chose to classify the elements of the system as either 'managed' or 'natural'. A managed patch consisted of one, or (if adjacent) more than one, unit that was clearcut in the past.

Maps of forest patterns were constructed for the years 1972 and 1987. Managed forest patches for 1972 were mapped using high-altitude infrared images (scale 1:60 000) and were transferred to US Forest Service vegetation maps using a zoom transfer scope. The 1987 data were acquired from vegetation maps produced in 1987 by the Forest Service at a scale of 1:15 840 for the entire Willamette National Forest. Fifteen forested landscapes were chosen at random from the Blue River and Sweet Home ranger districts' vegetation maps for this study. Each sample landscape consisted of a rectangle representing approximately 3·5 × 5·0 km (1750 ha). To qualify for selection, each landscape was required to be forested, lie within US Forest Service land, and contain at least one managed patch by 1972. Other landscapes were rejected during the random selection process. Several of the selected landscapes included roadless areas with much uncut forest land. The locations, sizes, and shapes within the managed forest patches and the natural forest matrix were digitally recorded as polygons for both the 1972 and 1987 data. The total number of digitized patches for 1972 and 1987 were 150 and 298, respectively.

Landscape statistics

The degree of fragmentation sustained by the forest matrix which characterizes a given landscape may be described as a function of the varying size, shape, spatial distribution, and density of clearcut patches (Burgess & Sharpe, 1981). Thus the degree of fragmentation can be measured in a number of ways. Because a single statistic is usually deemed insufficient to capture the entire spatial character of the landscape, a suite of statistics was selected. Five groups of statistics were used to quantify landscape heterogeneity and pattern for each of the fifteen areas for each of the two years: (1) patch size; (2) patch abundance; (3) patch shape; (4) patch spacing; and (5) matrix characteristics. Patch sizes and shapes were only determined for interior patches (i.e. patches that were not truncated by the borders of the landscape study sites).

Patch size for each year and sample area were expressed in terms of the average patch area and average patch perimeter. The area and perimeter of each patch were computed using the digitizing routine mentioned above.

The means of the patch areas and perimeters for each landscape for each year were also calculated. The second set of statistics, 'patch abundance', includes a measure of the patch density (expressed as the number of managed forest patches present per landscape study area) and percent in patches (the percent of the total landscape area occupied by managed patches). Means of these two statistics were calculated for each landscape and year. Because patch shape and patch spacing statistics can involve more than one variable in their calculation, some background discussion of the equations is included.

Patch shape was measured in three different ways: (1) the simple ratio of patch perimeter to patch area; (2) the fractal dimension; and (3) a diversity index. All three indices are a function of the perimeter and area of a given patch. The application of these similar measures on the same data set afforded an opportunity to compare their ability to detect spatial pattern.

The fractal dimension, D, was used to quantify the complexity of the shape of a patch using a perimeter–area relation. Specifically

$$P \approx A^{D/2} \tag{1}$$

where P is the digitized patch perimeter and A is the patch area (Burrough, 1986). The fractal dimension for each sample area and year was estimated by regressing the logarithm of patch area on its corresponding log-transformed perimeter. The appeal of fractal analysis is that it can be applied to spatial features over a wide variety of scales. A fractal dimension greater than 1 indicates a departure from a euclidean geometry, i.e. an increase in shape complexity. As D approaches 2, the patch perimeter becomes 'infilling' (Krummel *et al.*, 1987).

A similar index is the diversity index, DI, which was used to express patch shape as

$$DI = \frac{P}{2\sqrt{\pi A}} \tag{2}$$

where the variables are defined as above in eqn (1) (Patton, 1975). Theoretically, the diversity index increases to 1 as the unit shape approaches a circle, similar to the case of the fractal dimension. However, in contrast, the diversity index increases without limit as patch shape becomes more complex.

Patch spacing was characterized by measures of the mean nearest-neighbor distance and a measure of dispersion. The mean nearest-neighbor distance was calculated manually with a scale by measuring the distance from the centroid of each patch to the centroid of its nearest neighbor and computing the mean distance for the sample landscape. The centroid of each patch was determined through an ocular estimate procedure. Clark and

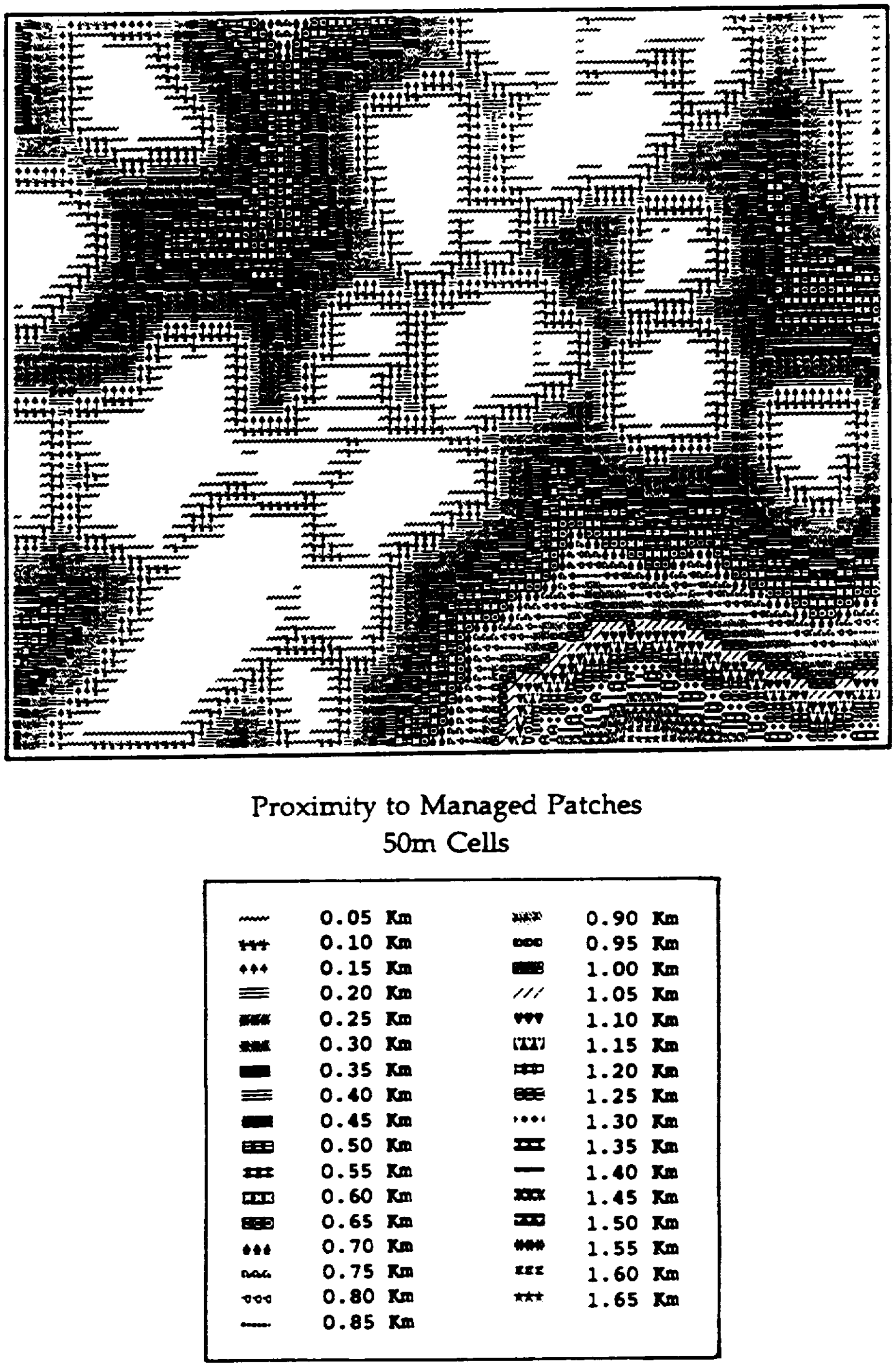


Fig. 1. An example of a proximity map using the spread function in the pMAP geographic information system. Cell values in the matrix were assigned based on their distance to the nearest managed forest patch (areas shown in white). The GIS fragmentation index (GISfrag) was determined by calculating the mean of all grid cell values on the proximity map.

Evans (1954) developed a measure of dispersion of patches using the equation

$$R = 2\rho^{1/2}\bar{r} \tag{3}$$

where R is dispersion, $\bar{r}$ is the mean nearest-neighbor distance, and ρ is the mean patch density (number of patches per unit area; see also Pielou, 1977, p. 155). Dispersion is a measure of the non-randomness of the patch arrangement. In a random population, $R = 1$; R less than 1 indicates aggregation of the patches, while R greater than 1 indicates that the patch population forms a regular dispersed pattern or spacing.

A fifth group of statistics was calculated which predominantly reflects the character of the matrix (in this case, natural forested land) as opposed to managed patch configuration (clearcut areas): namely, a GIS-derived index (christened *GISfrag*), matrix contiguity, interior habitat, and total patch edge. To determine contiguity, a mylar sheet with an 8 × 8 grid consisting of 64 cells each with a size of 27 ha was overlaid on the sample landscape maps. The largest contiguous natural forest area (i.e. the largest number of contiguous grid cells) in the sample landscape was recorded. Using this 8 × 8 grid, the contiguity index could potentially range from 0 (a landscape with no natural forest patches greater than 27 ha, i.e. highly fragmented) to 64 (a landscape with no managed stands, and hence no fragmentation). A 27-ha cell size was chosen because it was considered to be a viable habitat patch size and fits within the structure of the existing landscape.

The GIS fragmentation index (GISfrag) was computed by first producing a proximity map using the SPREAD function in the pMAP GIS software (Spatial Information Systems, 1986). This procedure assigned cell values based on the distance to the managed forest patches (i.e. a distance of one cell away from a managed patch was equal to 50 m, a distance of two cells was equal to 100 m, and so forth; see Fig. 1 for an example of a digital map illustrating all matrix distances to managed patches). A GISfrag was computed as the mean value of all the grid cell values on the proximity map, including the managed patches which were assigned values of zero. Large mean values reflected a low degree of forest fragmentation while maximum fragmentation occurred when the mean values approach zero.

The spread function in pMAP was also used to calculate the amount of interior forest habitat. Interior forest was defined as the amount of natural forest remaining after removing an edge zone of 100 m (approximately two tree heights) into the natural forest matrix (Fig. 2). The mean total edge was simply the average of the total managed patch edges for each landscape. A non-parametric test (Wilcoxon rank sum) for the difference between mean values was performed for each landscape and year for each of the variables listed above to assess the ability of these variables to reflect landscape change.

Interior Forest Habitat
50m Cells

Symbol	Label	Percent
	Interior Habitat	57.73
	Edge Effect (100m)	21.87
	Managed Forest	20.40

Fig. 2. An example of an interior forest habitat map generated by pMAP. The interior forest habitat is defined here as the amount of natural forest remaining after removing an edge zone of 100 m into the natural forest.

RESULTS AND DISCUSSION

Landscape structure

The degree of forest fragmentation as measured by the statistics discussed above increased significantly over the 15 years spanning 1972 to 1987 (Table 1 and Fig. 3). Overall, patch area and patch perimeter decreased by 17% and

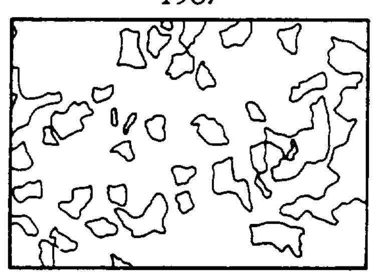
1972

Mean Patch Perimeter (m)	1793	Contiguity	34
Mean Patch Area (ha)	14.7	Interpatch Distance (m)	841
Perimeter/Area	0.014	GIS Fragmentation (km)	0.5
Fractal	1.26	Dispersion	1.40
Diversity Index	1.36	Interior Habitat (%)	79.7
Percent in Patches (%)	8.5	Total Patch Edge (km)	20.2
Patch Density (no./area)	12		

0 km 5

SCALE

1987



Mean Patch Perimeter (m)	1387	Contiguity	2
Mean Patch Area (ha)	8.1	Interpatch Distance (m)	498
Perimeter/Area	0.020	GIS Fragmentation (km)	0.2
Fractal	1.28	Dispersion	1.47
Diversity Index	1.42	Interior Habitat (%)	36.8
Percent in Patches (%)	27.3	Total Patch Edge (km)	68.0
Patch Density (no./area)	38		

Fig. 3. A graphical and statistical illustration of landscape change between the years 1972 and 1987 for one of the landscapes used in the study.

TABLE 1
Descriptive Statistics for 1972 and 1987 Landscapes

Variable	*Mean*	*Standard deviation*	*Minimum*	*Maximum*	*p value*
Managed patch size					
Mean patch perimeter (m) 1972	2 075	512	1 599	3 492	
Mean patch perimeter (m) 1987	1 848	452	1 211	2 797	0·096
Mean patch area (ha) 1972	19·5	9·1	1·7	42·7	
Mean patch area (ha) 1987	16·2	8·3	6·6	38·5	0·027
Managed patch shape					
Perimeter/area 1972	0·013	0·0037	0·007	0·022	
Perimeter/area 1987	0·018	0·0053	0·007	0·028	0·027
Fractal 1972	1·26	0·02	1·24	1·28	
Fractal 1987	1·27	0·02	1·24	1·32	0·023
Diversity index 1972	1·38	0·11	1·22	1·53	
Diversity index 1987	1·40	0·09	1·23	1·52	0·027
Managed patch abundance					
Percent in patches 1972	9·5	7·0	1·3	27·3	
Percent in patches 1987	18·2	9·6	1·3	38·1	0·001
Patch density (no./area) 1972	10·3	8·2	2	28	
Patch density (no./area) 1987	19·6	9·7	2	38	0·001
Managed patch spacing					
Nearest-neighbor distance (m) 1972	928	251	589	1 375	
Nearest-neighbor distance (m) 1987	661	247	454	1 375	0·001
Dispersion 1972	1·22	0·27	0·79	1·66	
Dispersion 1987	1·24	0·23	0·79	1·60	1·000
Matrix characteristics					
GIS fragmentation (km) 1972	0·8	0·5	0·1	1·8	
GIS fragmentation (km) 1987	0·5	0·5	0·1	1·8	0·003
Matrix contiguity 1972	36·6	19·9	1	59	
Matrix contiguity 1987	20·6	19·1	1	59	0·003
Interior habitat (%) 1972	78·2	16·1	38·4	96·9	
Interior habitat (%) 1987	60·2	19·7	25·6	96·9	0·001
Total patch edge (km) 1972	19·9	15·6	2·8	59·5	
Total patch edge (km) 1987	38·4	19·8	2·8	71·6	0·001

These statistics were calculated using patch and matrix variables of the fifteen landscapes to investigate the change in forest fragmentation over time. A non-parametric means test (Wilcoxon rank sum) was performed to evaluate the statistical significance of the differences in the means (see p values).

11% or from 19·5 to 16·2 ha and 2075 to 1848 m, respectively. Although some 1972 patches 'grew' by 1987 as a result of the coalescence of two or more managed patches, individual managed patch size on the whole decreased.

The fractal dimension, the perimeter-to-area ratio, and diversity index indicate a statistically significant increase in patch complexity over time. A contributing factor to this increased complexity may be attributed to the linking of several smaller managed patches which often produces an irregularly shaped patch. The fractal dimension and diversity index appear to be fairly robust measures of an 'average' patch shape and may be capable of discerning subtle changes in patch configuration which are difficult to assess by visual inspection alone. In contrast, because it is not scale-invariant, the perimeter-to-area ratio must be interpreted carefully.

Patch density increased over the 15 years by 98%. The percent of the landscape in managed patches nearly doubled in the 15-year period from 9·5% to 18·2%. The mean of the index of contiguity decreased by 44% by 1987, reflecting the increased number of clearcuts made in the area. Not surprisingly, the mean nearest-neighbor distance also decreased significantly from 928 to 661 m. Dispersion, however, did not register the change in spatial distribution of patches over time. There was no difference in dispersion from 1972 to 1987 (1·22 versus 1·24), which represents a regular, dispersed spacing pattern for both dates. Since dispersion is a function of

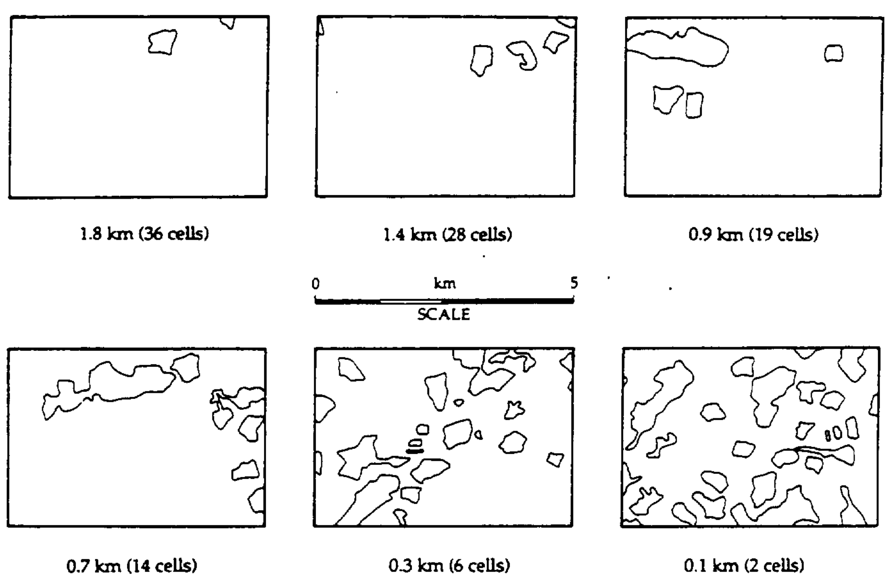


Fig. 4. The GIS fragmentation index (GISfrag) for a progression of landscape fragmentation levels. Low levels of GISfrag reflect high amounts of forest fragmentation in terms of number of clearcut patches and amount of patch aggregation. This figure consists of six different landscapes and does not represent the same landscape over time.

both patch density and interpatch distance, we expected it to change because these other variables were statistically different over time. However, the mean change in these two variables was in opposite directions, i.e. patch density *increased* while nearest-neighbor distance *decreased*. We conclude that their combined effects canceled one another. The amount of interior forest habitat decreased from 78·2% in 1972 to 60·2% in 1987. This decrease in interior habitat (18% loss) was approximately twice the areal increase in managed patches (8·7%).

The GISfrag mean decreased by 0·3 km in 1987, indicating an increase in forest fragmentation (Table 1). Figure 4 shows the GISfrag for a progression of landscape fragmentation levels. The GISfrag seems to be sensitive to the abundance of patches and the amount of unfragmented contiguous natural forest. Figure 5 demonstrates the influence of patch spacing and the amount of contiguous natural forest on the GISfrag by showing the GISfrag for a pair of landscapes that had the same patch density but a different level of patch aggregation. It should be noted that the GISfrag is only comparable among study areas that are the same size and shape, because the truncation of the matrix as it intersects the study area boundary may influence the value of this index.

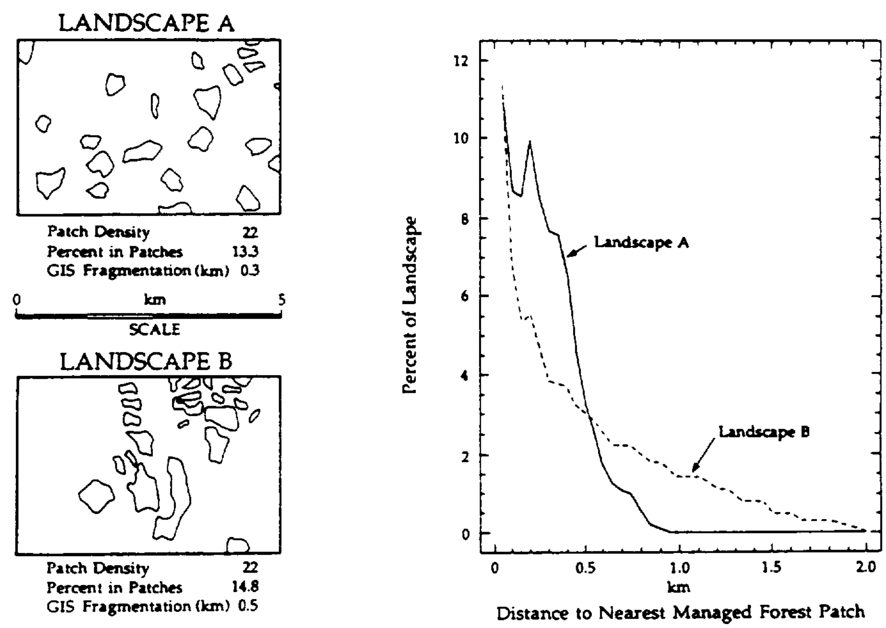


Fig. 5. An example of two landscapes (A and B) having the same patch density (22) but differing level of patch aggregation reflected by correspondingly different GIS fragmentation indices. The accompanying figure to A and B is a graph of percent of each landscape versus distance of each point in the landscape to a clearcut (managed) forest patch. The B profile shows more of the landscape with greater distances to the nearest managed patch than A.

Ecological and management implications

An examination of the results reveals the varying ability of each analysis to describe change in patch and matrix characteristics. The nature and amount of change detected by these different landscape statistics have significance to the ecology and management of forest landscapes. Changes that were evident in patch-level characteristics indicate a trend toward smaller units and a slight trend toward more irregular units. These changes may reflect the increased cutting on steeper, more irregular terrain, more careful 'fitting' of the cutting units as the available cutting area decreased, an effort to optimize big game habitat by increasing the number of cuts, and/or an effort to reduce the visual impact of clearcutting.

The increase in the number of young, managed stand patches and the total amount of edge has implications for habitat potential of the landscapes. Although conclusive information on the effect of edges on vertebrates in western coniferous forest landscapes is not yet available, an increase in edge can benefit some species but prove detrimental to others (Yahner, 1988). For example, it has been observed that big game animals show an affinity for edges (Brown, 1985) and that some bird species occur more frequently on edges than in forest interiors (Rosenberg & Raphael, 1986). The fact that edge density increased in the present study areas over the past 15 years suggests that the habitat potential has increased for such species as elk, which can successfully utilize the edge environment. The increased dispersal of small clearcuts into the matrix of forest cover provides a corresponding increase in the amount of hiding cover close to forage areas used by the elk (Brown, 1985).

Conversely, a number of other vertebrate species, such as the northern spotted owl, Townsend's warbler *Dendroica townsendi*, and *pileated* woodpecker *Dryocopus pileatus*, may avoid edges (Bull, 1975; Brown, 1985; Rosenberg & Raphael, 1986). The increase in edge density indicates that habitat conditions for such species favoring interior forest have probably declined markedly. Specifically, the decrease in the mean distance of matrix to managed patch (as measured by GISfrag) and interior forest area are evidence of the decline in interior species habitat conditions. The manner and degree to which the decline affects interior species populations in the study areas are difficult to assess. Although Rosenberg and Raphael (1986) did not find a strong response of vertebrate communities in northern Californian landscapes which sustained a mean percent clearcut of 18% (roughly the same as the present study), the authors warned that the fragmentation in the region is a relatively recent phenomenon. Long-term vertebrate responses were not yet discernible.

The continued use of dispersed clearcutting increases fragmentation of forest landscapes at a rate more rapid than the rate of cutting on a per area basis. Given that the present cutting patterns are decreasing potential interior habitat at a rapid rate, alternative cutting patterns should be considered to reduce the loss of interior habitat and retain the area of large forested patches. Alternative models which aggregate cutting (Franklin & Forman, 1987) are available and may not require altering current standards and guidelines.

CONCLUSIONS

In an attempt to describe forest fragmentation, five groups of statistics were employed: patch abundance, patch shape, patch size, patch spacing, and matrix characteristics. By comparing two sets of data representing two dates over a 15-year period, we found that patch abundance, patch spacing measures, and matrix characteristics were most useful in capturing the amount of forest fragmentation over time. Patch size and shape statistics contribute information on specific characteristics of the individual patches and may be useful for applications designed to study specific interior and edge habitats or for the prescription of new clearcuts. In addition, a GIS fragmentation index was developed which proved sensitive to both the quantity and spacing of patches. The GIS provides an automated method of quantifying forest fragmentation to aid in forest and wildlife management decisions, and a means by which field and image data may be used in concert.

Current concerns over forest fragmentation are typically related to a landscape condition in which forest islands occur in a matrix of managed forest plantations. This study suggests that on many Forest Service lands in the Cascade Range this condition is not yet realized, in contrast to many privately owned landscapes in the Cascades in which the matrix is the harvested area and the patches are the natural forest. Consequently, in characterizing fragmentation in some landscapes, characteristics of the matrix (the unmanaged forest) may be of more interest than characteristics of the patches (the managed plantations). Where the matrix is of interest, the interior area, the total edge, and the mean distance to the nearest managed patch (GISfrag) will be the useful descriptors of fragmentation. Additional research is needed to document and substantiate the relationship between forest landscape pattern and the subsequent wildlife/ecosystem response.

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A preliminary comparison of Landsat Thematic Mapper and SPOT-1 HRV multispectral data for estimating coniferous forest volume

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Abstract. Digital Landsat Thematic Mapper (TM) and Satellite Probatoire d'Observation de la Terre (SPOT) High Resolution Visible (HRV) images of coniferous forest canopies were compared in their relationship to forest wood volume using correlation and regression analyses. Significant inverse relationships were found between softwood volume and the spectral bands from both sensors ($P<0.01$). The highest correlations were between the log of softwood volume and the near-infared bands (HRV band 3, $r=-0.89$; TM band 4, $r=-0.83$).

1. Introduction

The potential to extract forest volume information from remotely sensed satellite sensor data could substantially aid inventories of forest resources. In this study, we examined the capabilities of both the Landsat Thematic Mapper (TM) and SPOT HRV multispectral satellite data to extract forest volume information. The TM has six reflective spectral bands with 30 m spatial resolution plus a thermal band, while SPOT has three reflective spectral bands with 20 m resolution (table 1).

There is normally an inverse relationship between vegetation amount and both visible reflectance (0·4 to 0·7 μm) and middle-infrared reflectance (1·3 to 2·6 μm) because of highly absorbing plant pigments and strong absorption from water in the leaves respectively (Curran 1980, Ripple 1986). Conversely, near-infrared reflectance typically has a direct relationship with vegetation amount because of scattering and little or no absorptance (Curran 1980). Researchers have found that this direct relationship in the near-infrared with increasing amounts of conifer biomass does not always exist. The near-infrared response can be flat (Franklin 1986, Peterson *et al.* 1987) or inverse (Danson 1987, Spanner *et al.* 1990) depending upon the amount and brightness of exposed understory or background.

A number of researchers have evaluated the potential for using SPOT HRV data (Danson 1987, De Wulf *et al.* 1990, Hame *et al.* 1988), TM simulator data (Franklin 1986), and TM data (Horler and Ahern 1986, Spanner *et al.* 1990) for estimating forest stand parameters, but only one study was found that compared SPOT and TM for monitoring forest stand parameters (Brockhaus *et al.* 1988). In the comparison study, several TM and SPOT bands were significantly correlated with basal area and stand age in coastal plain forests of North Carolina, although the authors concluded that these correlations were too low to be used in developing a predictive model (Brockhaust *et al.* 1988). Chavez and Bowell (1988) compared the spectral information content of TM and HRV data over geologic, urban, and agricultural sites in

Table 1. Comparison of Landsat TM and SPOT-1 HRV satellite sensor spectral characteristics for McDonald Forest study site (42,419 SPOT pixels and 21,120 TM pixels).

Landsat TM					SPOT-1 HRV				
		Digital numbers					Digital numbers		
Band	Spectral width (μm)	Mean	S.D.	Range	Band	Spectral width (μm)	Mean	S.D.	Range
1 (blue-green)	0·45–0·52	72·7	6·1	144					
2 (green)	0·52·0·60	27·5	4·6	67	1 (green)	0·50–0·59	28·2	4·2	105
3 (red)	0·62–0·69	24·9	8·6	95	2 (red)	0·61–0·68	17·8	5·4	91
4 (near-IR)	0·76–0·90	94·7	20·4	145	3 (near-IR)	0·79–0·89	75·3	15·0	112
5 (midle-IR)	1·55–1·75	59·5	23·2	139					
6 (thermal-IR)	10·40–12.50	136·2	6·2	44					
7 (middle-IR)	2·08–2·35	17·6	10·1	89					

Arizona, U.S.A., and found the TM data contained more spectral information than the SPOT multispectral data, with TM bands showing field/soil information not available in the SPOT bands.

The objectives of our study were to compare information content of digital TM and HRV multispectral data over a forested test site and to examine the relationships between the sensor data and softwood volume. The study site was in McDonald State Forest located in the foothills of the Coast Range mountains just outside of Corvallis, Oregon (centred on 44° 37′ N lat., 122° 19′ long.). This forest is managed by the College of Forestry at Oregon State University. Douglas-fir (*Pseudotsuga menziesii*) is the dominant tree species. The study area consists of young forest plantations established after clearcutting and older forest stands (range 25–148 years old). Both deciduous shrub and herb understory are common throughout the forest. Precipitation averages 107 cm per year with most of it falling as rain from October through May.

2. Methods

The satellite sensor images were from a 25 July, 1988 HRV scene and a 30 July 1988 TM scene. The SPOT data were obtained at a +10·7° incidence angle and processed at level 1B. Sub-areas representing approximately the same geographic areas were extracted from each image using the following procedure. Screen images of both the TM and HRV images were projected to permit selection of a set of pixels representing the same ground features. For each feature, a pair of x, y locations were determined, one from the HRV data and one from the TM data. A least-squares multiple regression was used to develop a transformation model (Isaacson *et al.* 1987). This approach permitted the direct transformation between TM image space and HRV image space without a resampling or rectification to a common scale or projection, thereby better preserving the original data values for more direct comparison of the study area images. The study area consisted of 169 rows and 251 columns of HRV data and 120 rows and 176 columns of TM data. No corrections were made for path radiance.

Forty-six forest stands were delineated on both the TM and HRV screen images. The delineations were made at locations of approximately one pixel width in from the edge of the stands to reduce edge effect. The mean pixel digital number (DN) was calculated for each band for each of the 46 stands for both the TM and HRV data. Softwood volumes ranged from 28·3 to 839·5 $m^3 ha^{-1}$. Forest stand areas ranged from 2·1 to 74·1 ha. The number of interior pixels in the 46 stands ranged from 12 to 566 for the HRV data and 3 to 258 for the TM data.

A correlation analysis was conducted on all spectral bands from both image subsets ($n = 46$). Softwood volume data were obtained from permanent timber inventory plots established and maintained by College of Forestry staff, with volumes determined using local tarif tree volume computation guidelines. Softwood volume represents the volume of wood in the bole of the conifers and is expressed in cubic metres per hectare ($m^3 ha^{-1}$). Correlation, simple, and multiple regression analyses were conducted on the spectral band/softwood volume data set.

3. Results and discussion

All of the spectral bands were highly correlated, both within and between the TM and HRV data sets (table 2). Because of the relatively homogeneous forested area in our test site, the correlation coefficients between any two bands tended to be higher

Table 2. Correlation coefficient matrix for the HRV, TM, softwood volume, and log softwood volume variables ($n = 46$). All correlations were significant at the 0·01 probability level.

	HRV1	HRV2	HRV3	TM1	TM2	TM3	TM4	TM5	TM6	TM7	Volume	Log volume
HRV1	1·00											
HRV2	0·90	1·00										
HRV3	0·87	0·62	1·00									
TM1	0·77	0·82	0·62	1·00								
TM2	0·93	0·79	0·87	0·80	1·00							
TM3	0·88	0·87	0·72	0·85	0·91	1·00						
TM4	0·81	0·57	0·97	0·69	0·88	0·73	1·00					
TM5	0·87	0·70	0·86	0·73	0·91	0·81	0·87	1·00				
TM6	0·60	0·65	0·44	0·76	0·61	0·65	0·49	0·50	1·00			
TM7	0·87	0·74	0·77	0·64	0·88	0·78	0·75	0·95	0·48	1·00		
Softwood volume	−0·77	−0·63	−0·82	−0·61	−0·72	−0·69	−0·76	−0·63	−0·40	−0·55	1·00	
Log softwood volume	−0·83	−0·65	−0·89	−0·58	−0·82	−0·72	−0·83	−0·76	−0·38	−0·71	0·92	1·00

than would normally be expected. This was also found to be the case with spectrally homogeneous areas analysed by Chavez and Berlin (1984). Similar inverse relationships were found between softwood volume and the spectral bands from both sets of data. The correlation coefficients ranged from $-0{\cdot}63$ to $-0{\cdot}82$ for HRV bands and from $-0{\cdot}40$ to $-0{\cdot}76$ for TM bands. The slightly higher correlations for the SPOT data may be due to differences in spatial resolution, the wavelength range or position of the bands, or the signal to noise ratios. The higher spatial resolution of the SPOT HRV may be more sensitive to vegetation density than the lower spatial resolution of the TM data set, although the correlation coefficient is larger for TM band 3 ($-0{\cdot}69$) when compared to HRV band 2 ($-0{\cdot}63$). An examination of the scatter diagrams revealed inverse curvilinear relationships between many of the spectral bands and softwood volume.

Log transformations of softwood volume increased the correlation coefficients for a number of the HRV and TM bands (table 2). This was probably due to the asymptotic nature of these relationships as tree volumes increased in older stands with higher canopy closure (Ripple 1985). Figure 1 shows similar relationships for both TM band 4 with log volume ($r=-0{\cdot}83$) and HRV band 3 with log volume ($r=-0{\cdot}89$). These results confirm those found by Danson (1987), where HRV band 3 was inversely related to mean height ($r=-0{\cdot}83$) of conifers and DeWulf *et al.* (1990) who found HRV band 3 to have the strongest relationship with forest stand parameters. We speculate that these inverse relationships were caused by (1) increased canopy shadowing within larger older stands, and (2) decreased understory brightness with the closing of the conifer canopy (Spanner *et al.* 1990).

With the HRV data, stepwise multiple regression ($P \leqslant 0{\cdot}05$) resulted in only one independent variable (band 3) for predicting log softwood volume ($r=-0{\cdot}89$). With the TM data set, bands 4 and 2 were significant using the stepwise procedure with log softwood volume as the dependent variable ($R^2=0{\cdot}73$). The simple regression of log volume on TM band 4 alone explained just slightly less variance ($r^2=0{\cdot}69$) than the stepwise procedure above. This indicates that the additional spectral bands available on the TM sensor may not contribute significantly to increasing the accuracy of

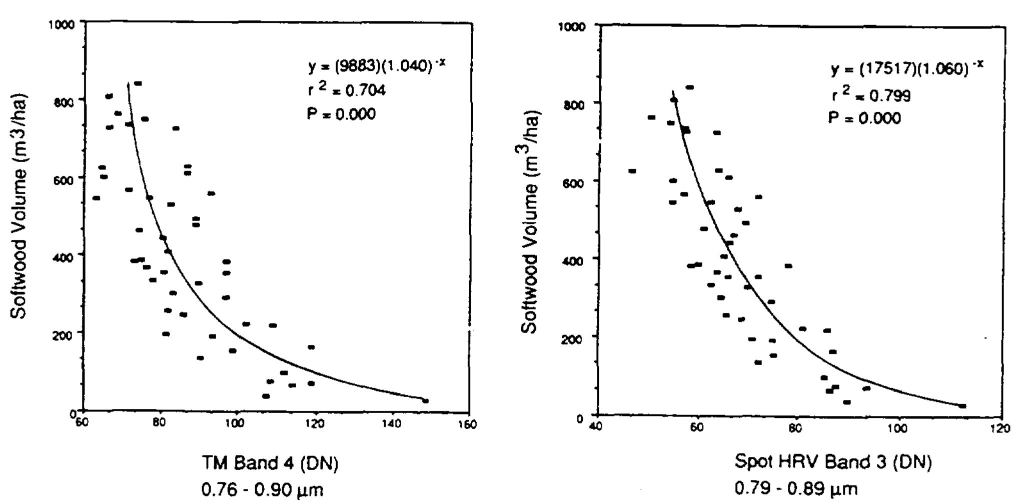

Figure 1. Scatter diagrams and results of regressing softwood volume on both TM band 4 and HRV band 3 for the McDonald Forest study site. Each point on the scatter diagrams represents 1 of 46 observations.

estimates of softwood volumes in relatively homogeneous Douglas-fir stands. Simple regression is preferred because when spectral bands are used as independent variables in multiple regression, intercorrelations among spectral bands can affect the regression coefficients and the sum of squares. It should be noted that the regressions in figure 1 involving softwood volumes were intended only for the comparison of HRV with TM data. In actual applications, line fitting corrections may be necessary (Curran and Hay 1986).

We believe the success of using the single-infrared bands to determine softwood volume was due to the following:

(*a*) Since the understory had a highly reflective shrub and herb layer, the young open conifer stands with low softwood volume had higher radiance than the older voluminous stands having more shadows, thus causing the strong inverse relationships.
(*b*) The dynamic range of the infrared bands were higher than the other spectral bands.
(*c*) The study area was relatively homogeneous and dominatd by Douglas-fir trees. In study areas that are heterogeneous with more than one vegetation cover type, multiband data sets will probably be critical for estimating softwood volume.

Under the conditions of this study, the following conclusions can be drawn:

(i) Both HRV and TM data sets exhibited high band to band correlations.
(ii) Both HRV and TM data showed similar significant inverse relationships with softwood volume.
(iii) We speculate that the inverse relationships were caused by canopy shadowing and/or the extent of the bright deciduous understory exposed to the sensor.
(iv) With the exception of the red bands (HRV band 2 and TM band 3), the HRV data had slightly higher correlations with softwood volume than the TM data. This was probably due to the higher spatial resolution of the HRV although differences in the location and width of the bands or the signal to noise ratios could also be factors.
(v) The near-infrared band was the most accurate for predicting softwood volume from both the HRV and the TM.
(vi) We speculate that inverse curvilinear relationships between spectral bands and softwood volume were caused by asymptotic charactristics of the forest canopy/spectral relationship.

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3P

OLD-GROWTH AND MATURE FORESTS NEAR SPOTTED OWL NESTS IN WESTERN OREGON

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Abstract: We investigated how the amount of old-growth and mature forest influences the selection of nest sites by northern spotted owls (*Strix occidentalis caurina*) in the Central Cascade Mountains of Oregon. We used 7 different plot sizes to compare the proportion of mature and old-growth forest between 30 nest sites and 30 random sites. The proportion of old-growth and mature forest was significantly greater at nest sites than at random sites for all plot sizes ($P \leq 0.01$). Thus, management of the spotted owl might require setting the percentage of old-growth and mature forest retained from harvesting at least 1 standard deviation above the mean for the 30 nest sites we examined.

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The northern spotted owl is associated with large tracts of old forest habitat found in the Pacific Northwest (Forsman et al. 1977, 1984; Forsman and Meslow 1985), and it has been hypothesized that the number of birds is declining as old forests are harvested and become more fragmented (Gould 1977, Barrows 1981, Gutiérrez 1985, Forsman et al. 1987, U.S. Fish Wildl. Serv. 1989). Carey et al. (1990), for example, found that home-range size was negatively correlated with the proportion of old-growth forest in the home range. However, little is known about how the amount of old-growth and mature forests influences the selection of nest sites by spotted owls. We hypothesized that if northern spotted owls benefit from old growth, then the proportion of old-growth and mature forests at northern spotted owl nest sites should be greater than at randomly selected sites. We tested this hypothesis by testing for significant differences in the mean values of percent old-growth and mature forest for various plot sizes at 30 spotted owl nest sites compared to 30 random sites.

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STUDY AREA

The study area lies on the western slope of the Cascade Range in central Oregon (43°43′–44°37′N, 121°55′–122°50′W) and encompasses the Sweet Home, Blue River, McKenzie, and Lowell ranger districts on the U.S. Forest Service (USFS) Willamette National Forest. The 100-km (north–south) by 71-km (east–west) study area falls within portions of the western hemlock (*Tsuga heterophylla*), Pacific silver fir (*Abies amabilis*), and mountain hemlock (*Tsuga mertensiana*) zones with the major tree species consisting of Douglas-fir (*Pseudotsuga menziesii*), western hemlock, Pacific silver fir, noble fir (*Abies procera*), and Western redcedar (*Thuja plicata*) (Franklin and Dyrness 1973). Elevation ranges from 300 to 1,600 m. The topography is dissected by many steep slopes. The climate is maritime. Annual precipitation averages 230 cm at low and 330 cm at high elevations. Winter snowpack ranges from 1 to 3 m above 500 m elevation.

The study area has been logged extensively during the last 60 years, with dispersed clearcutting the prevalent harvest practice. A minor proportion of the study area is composed of scattered, privately-owned lands which are managed for timber production.

METHODS

We used techniques described by Forsman (1983) to survey the study area and to locate spotted owl nests. Thirty-seven nest locations were discovered during an intensive 3-year search of the study area. Thirty nest sites from the total of 37 were randomly selected for analysis in our study. Of the 30 nests, 3 were located in 1987, 23 in 1988, and 4 in 1989. For each of the 30 nest sites, we recorded the Universal Transverse Mercator (UTM) coordinates and plotted the nest sites on orthophoto quadrangles

Table 1. Percentage of area in old-growth and mature forest near 30 spotted owl nest sites and 30 random sites for 7 plot sizes in the Cascade Mountains of Oregon, 1988.

Plot size (ha)	Radius (m)	Nest sites[a]				Random sites[a]				P-value[b]	
		$\bar{x}$	Min.	Max.	SD	$\bar{x}$	Min.	Max.	SD	$\bar{x}$	Variance
260	910	78.2	51	100	11.8	63.2	15	100	20.2	0.0019	0.0025
440	1,183	76.3	48	100	11.9	63.5	25	97	17.7	0.0026	0.0182
620	1,405	76.5	47	97	11.3	63.3	28	95	15.7	0.0006	0.0409
800	1,596	75.6	45	97	11.1	62.5	26	90	15.3	0.0004	0.0447
980	1,766	75.1	42	97	11.1	61.6	27	87	14.5	0.0002	0.0781
1,826	2,411[c]	73.6	45	94	9.9	60.8	26	86	13.5	0.0001	0.0502
3,588	3,379[d]	65.0	40	79	8.7	51.3	22	78	13.2	0.0101	0.0141

[a] All values are expressed as percentage of total plot area.
[b] P-values based on arc-sine square-root proportion transformation.
[c] Equal to a 1.5-mile radius.
[d] Equal to a 2.1-mile radius.

(scale 1:24,000). We also plotted the locations of 30 randomly selected sites on orthophotos. The random sites were selected from the entire study area; points falling in water or on major lava flows were excluded. All nest sites and random sites were located on USFS land, and none was in designated wilderness, roadless, or research natural areas. We analyzed 30 nest sites and 30 random sites for 7 circular plot sizes: 260, 440, 620, 800, 980, 1,826, and 3,588 ha. The 1,826- and 3,588-ha plot sizes have radii of 2.4 km (1.5 miles) and 3.4 km (2.1 miles), respectively.

Percent old-growth and mature forest (>80 yr old) was determined through photo interpretation and field work for all plot sizes with a dot grid (150-m spacing) on the orthophotos (Avery 1977:76–77). We used 1988 aerial photographs and a zoom transfer scope to update the orthophotos to show recent clear-cuts. A Student's t-test was performed on the arc-sine square root transformed mean values of percent old-growth and mature forest for each plot size to determine if there were significant differences between the amount of old-growth and mature forest at the nest sites compared to random sites.

RESULTS

Differences between nest site means and random plot means were significant ($P \leq 0.01$) at all plot sizes. The standard deviations were higher for the random sites (Table 1) compared to the nest sites (P's < 0.08).

DISCUSSION

Apparently, the percentages of old-growth and mature forest both adjacent to nests (260-ha plots) and in the surrounding area (3,588-ha plots) are important in nest site selection. Spotted owls probably select nest sites on landscapes with low fragmentation and greater amounts of old-growth and mature forests because of structurally suitable trees for nests, ameliorated microclimates, suitable foraging substrates, refuges from predators, and/or sufficient prey (Barrows 1981, Forsman et al. 1984, Gutiérrez 1985).

We do not know how much fragmentation can be tolerated by spotted owls, or whether this toleration varies with location, quality of habitat, and length of time since timber harvesting. Until more data are available, we suggest that landscapes be managed conservatively for spotted owls in the Oregon Cascade Mountains. If the percentage of old-growth and mature forest was set at least 1 standard deviation above the mean for the 30 nest sites we examined, it would result, for instance, in a minimum of 83.5% old-growth and mature forest in a 1,826-ha area (2.4-km radius), and 73.7% old-growth and mature forest in a 3,588-ha area (3.4-km radius) around nests. When there is, on average, 83.5% old-growth and mature forest within the 2.4-km radius of a spotted owl nest, it would result in a minimum of 63.6% old-growth and mature forest in the concentric band between the 2.4-km and 3.4-km radii.

We did not address whether spotted owls should be managed on the basis of individual nest sites or extensive areas capable of supporting multiple pairs of owls, but in either case, the individual nest site remains a convenient reference for expression of habitat needs.

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2P

AN ABSTRACT OF THE THESIS OF

G.A. Bradshaw for the degree of Doctor of Philosophy in Forest Science presented on September 10, 1991.

Title: Hierarchical Analysis of Spatial Pattern and Processes of Douglas-fir Forests.

Abstract approved: _______________________

Susan G. Stafford

William J. Ripple

William J. Ripple

There has been an increased interest in the quantification of pattern in ecological systems over the past years. This interest is motivated by the desire to construct valid models which extend across many scales. Spatial methods must quantify pattern, discriminate types of pattern, and relate hierarchical phenomena across scales. Wavelet analysis is introduced as a method to identify spatial structure in ecological transect data. The main advantage of the wavelet transform over other methods is its ability to preserve and display hierarchical information while allowing for pattern decomposition.

Two applications of wavelet analysis are illustrated, as a means to: 1) quantify known spatial patterns in Douglas-fir forests at several scales, and 2) construct spatially-explicit hypotheses regarding pattern generating mechanisms. Application of the wavelet variance, derived from the wavelet transform, is developed for forest ecosystem analysis to obtain additional insight into spatially-explicit data. Specifically, the resolution capabilities of the wavelet variance are compared to the semi-variogram and Fourier power spectra for the description of spatial data using a set of one-dimensional stationary and non-stationary processes. The wavelet cross-covariance function is derived from the wavelet transform and introduced as an alternative method for the analysis of multivariate spatial data of understory vegetation and canopy in Douglas-fir forests of the western Cascades of Oregon.

AN ABSTRACT OF THE THESIS OF

Maria Fiorella for the degree of Master of Science in Forest Resources presented May 26, 1992.

Title: Forest and Wildlife Habitat Analysis Using Remote Sensing and Geographic Information Systems

Abstract approved: William J. Ripple
William J. Ripple

Forest and wildlife habitat analyses were conducted at the H.J. Andrews Experimental Forest in the Central Cascade Mountains of Oregon using remotely sensed data and a geographic information system (GIS). Landsat Thematic Mapper(TM) data were used to determine forest successional stages, and to analyze the structure of both old and young conifer forests. Two successional stage maps were developed. One was developed from six TM spectral bands alone, and the second was developed from six TM spectral bands and a relative sun incidence band. Including the sun incidence band in the classification improved the mapping accuracy in the two youngest successional stages, but did not improve overall accuracy or accuracy of the two oldest successional stages. Mean spectral values for old-growth and mature stands were compared in seven TM bands and seven band transformations. Differences between mature and old-growth successional stages were greatest for the band ratio of TM 4/5 ($P = 0.00005$) and the

multiband transformation of wetness ($P = 0.00003$). The age of young conifer stands had the highest correlation to TM 4/5 values ($r = 0.9559$) of any of the TM band or band transformations used. TM 4/5 ratio values of poorly regenerated conifer stands were significantly different from well regenerated conifer stands after age 15 ($P = 0.0000$). TM 4/5 was named a "Successional Stage Index" (SSI) because of its ability to distinguish forest successional stages.

The forest successional stage map was used as input into a vertebrate richness model using GIS. The three variables of 1) successional stage, 2) elevation, and 3) site moisture were used in the GIS to predict the spatial occurrence of small mammal, amphibian, and reptile species based on primary and secondary habitat requirements. These occurrence or habitat maps were overlayed to tally the predicted number of vertebrate at any given point in the study area. Overall, sixty-three and sixty-seven percent of the model predictions for vertebrate occurrence matched the vertebrates that were trapped in the field in eight forested stands. Of the three model variables, site moisture appeared to have the greatest influence on the pattern of high vertebrate richness in all vertebrate classes.

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Analysis of Conifer Forest Regeneration Using Landsat Thematic Mapper Data

Abstract

Landsat Thematic Mapper (TM) data were used to evaluate young conifer stands in the western Cascade Mountains of Oregon. Regression and correlation analyses were used to describe the relationships between TM band values and age of young Douglas-fir stands (2 to 35 years old). Spectral data from well regenerated Douglas-fir stands were compared to those of poorly regenerated conifer stands. TM bands 1, 2, 3, 5, 6, and 7 were inversely correlated with the age ($r \geq -0.80$) of well regenerated Douglas-fir stands. Overall, the "structural index" (TM 4/5 ratio) had the highest correlation to age of Douglas-fir stands ($r = 0.96$). Poorly regenerated stands were spectrally distinct from well regenerated Douglas-fir stands after the stands reached an age of approximately 15 years.

Introduction

Standard forestry practices require that harvested timber areas be reforested. Once a site is replanted, the stand needs continual monitoring to determine how reforestation is progressing. Information on stand condition is needed to manage forests for both timber and wildlife habitat objectives. Forest variables that are traditionally monitored are tree density, distribution, and quality; understory vegetation; seedling growth rate; and stand composition (Cleary *et al.*, 1978).

The western Cascade Mountains of Oregon are dominated by stands of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco.). Extensive areas of these natural forests have been harvested and replanted (Ripple *et al.*, 1991a). Newly replanted conifer stands usually progress from a herbaceous stage to one dominated by shrubs and conifer seedlings and saplings (Dyrness, 1973). Typically, these stands develop into a closed canopy condition where conifers dominate the site.

In this study, well regenerated stands were defined as stands which were progressing to canopy closure at an expected rate and were not dominated by hardwood trees and shrubs. These stands also had a relatively even tree size and spatial distribution. A closed canopy stand condition was defined to have at least 60 percent canopy closure (Brown, 1985). At 60 percent canopy closure, light to understory vegetation is limited and changes in understory species composition typically occur. These successional stage changes also influence wildlife species abundance and diversity (Brown, 1985; Harris, 1984; Hansen *et al.*, 1991).

Recent emphasis on landscape and regional analyses necessitates monitoring forest regeneration over large areas. Conditions within regenerating stands change quickly and, therefore, stand condition information must be updated periodically. Analysis of remotely sensed data from satellites has potential for assessing forest regeneration and wildlife habitat because it provides coverage over large geographic areas on a regular basis. Harvested areas on U.S. Forest Service land average 10 to 20 hectares (110 to 220 TM pixels). Landsat Thematic Mapper (TM) data may be suitable to monitor within stand condition because of the increased spatial and spectral resolution as compared to Multispectral Scanner (MSS) data.

In the past, radar data have been used with some success to identify clearcut stage by a photointerpreted method (Hardy, 1981) and by digital texture analysis (Edwards *et al.*, 1988). Thematic Mapper (TM) data have been used to update stand boundaries (Pilon and Wiart, 1990; Smith, 1988) and to monitor age of 0 to 12 year old conifer plantations (Horler and Ahern, 1986). Matejek and Dubois (1988) found a TM band 3, 4, 5 composite image useful in identifying young clearcuts and different age groups of conifer regeneration in Ontario, Canada. Spanner *et al.* (1989) used TM data to identify forest disturbance classes for a portion of this study area. Other related studies have used TM data to assess the influence of forest understory vegetation on satellite spectral data values (Spanner *et al.*, 1990; Stenback and Congalton, 1990) and to measure canopy closure with satellite data (Butera, 1986; Spanner *et al.*, 1990).

Objectives of this study were to use TM data to (1) describe the relationships between spectral data and age of young Douglas-fir forests, and (2) determine whether poorly regenerated stands could be separated from well regenerated stands.

Study Area

The research was conducted at the H.J. Andrews Experimental Forest located in the western Cascade Mountains of Oregon. Elevations range from 414 to 1630 metres above mean sea level. The majority of the study area falls within the Western Hemlock (*Tsuga heterophylla*) zone (Franklin and Dyrness, 1973). The dominant tree species in this zone is Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco.), a subclimax species. Western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) is a common understory or codominant species. The remaining portion of the study area falls into the Silver fir (*Abies amabilis*) zone occurring above 1100 to 1200 metres (Franklin and Dyrness, 1973). Most of these higher elevations are dominated by noble fir (*Abies procera* Rehd.) or Pacific silver fir (*Abies amabilis* (Dougl.) Forbes). The study area is representative of western Cascade Mountain forests managed by the U.S. Forest Service.

The traditional harvest method for Forest Service lands in this region included harvesting all trees in 10 to 20 hectare units. The harvested areas were usually burned to pre-

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pare the site for planting and to control brush (Cleary *et al.*, 1978). One to three year old seedlings were planted within two years of burning. Planting densities and species composition have varied, although the dominate species planted has been Douglas-fir. Stands which had high seedling mortality may have been replanted. Timber harvest in the experimental forest began in 1950, and these replanted managed stands represent approximately 25 percent of the forest landscape.

Methods

Data Development

An area representing the H.J. Andrews Experimental Forest was extracted from a 30 July 1988 TM quarter scene (scene ID Y5161218271). The TM data were rectified to a Universal Transverse Mercator (UTM) grid using a nearest neighbor resampling method. A 3 by 3 area of pixels was extracted from the TM data for well regenerated Douglas-fir stands within the study area. Well regenerated stands were defined as stands that were replanted to Douglas-fir (*Pseudotsuga menziesii*), were completely replanted within a two-year period, and were progressing to canopy closure at an expected rate. These stands had relatively even tree size and spatial distributions and were not overgrown with hardwood trees and shrubs. Stand age ranged from 2 to 35 years. Ancillary data, such as tree densities from field stand examinations (when available), and 1988, 1:12,000-scale color aerial photographs, were used to assess the success of stand regeneration. Multiple samples from each stand were typically used in large stands to account for topographic or stand level variability. A total of 61 samples were taken from 45 stands.

A 3 by 3 area of pixels was sampled from Douglas-fir stands that did not meet the criteria for well regenerated stands in the aerial photograph evaluation. These stands were dominated by shrubs, and deciduous trees, or herbaceous vegetation, and had few, sparsely distributed conifers. Based on the criteria for conifer regeneration, these stands were considered to be poorly regenerated. Age for poorly regenerated stands was determined from the last date in which the entire area had been planted or replanted. The 1988 color aerial photographs were used to estimate conifer canopy closure for all stands.

The aerial photographs were also used to describe understory vegetation in open stands and assign poorly regenerated stands to two different categories based on understory composition: (1) herbs and (2) shrubs. These two categories of poorly regenerated stands will be referred to as herb stands and shrub stands in the following text. There were 21 samples taken from 20 different herb stands and another 21 samples taken from 16 different shrub stands. The herb category included stands in which two-thirds or more of the non-conifer vegetation was low growing grasses, ferns, or other herbaceous plants. The shrub category included stands in which two-thirds or more of the non-conifer vegetation was shrubs or deciduous trees. Conifer canopy closure ranged from 0 to 45 percent (average 10 percent) in the herb stands and from 0 to 30 percent (average 11 percent) in the shrub stands. Common shrubs and deciduous tree species included the following: sitka alder (*Alnus sinuata* (Reg.) Rydb.), red alder (*Alnus rubra* Bong.), mountain alder (*Alnus tenuifolia* Nutt.), vine maple (*Acer cincinatum* Pursh), big leaf maple (*Acer macrophyllum* Pursh), and snowbrush ceanothus (*Ceanothus velutinus* Dougl.).

Data Analysis

Correlation analysis was used to determine the relationship between stand age and TM band values of well regenerated conifer stands. Seven TM bands, four band transformations (normalized difference vegetation index (NDVI) [(TM 4 − TM 3) / (TM 4 + TM 3)], and TM 4/3 and TM 4/7 band ratios), the structural index (SI), and the three TM Tasseled Cap features of brightness, greenness, and wetness were included in the analysis. The SI is simply the TM 4/5 band ratio (Fiorella and Ripple, 1993). The TM Tasseled Cap features are a linear transformation of TM bands 1, 2, 3, 4, 5, and 7 (Crist and Cicone, 1984). Correlations for linear, log-linear, and log-log relationships of stand age with band values were computed. The log-linear relationship was the correlation of the log of stand age with the TM data.

The band or band transformation with the highest correlation with stand age was then regressed against stand age (this was the SI - see results). These data were fit with a regression line with a 95 percent prediction interval. This prediction interval represented the range of possible values for a new observation from the population of well regenerated conifer stands. For comparison purposes, the mean SI values for the two subgroups of poorly regenerated stands were plotted against stand age and with the 95 percent prediction intervals from the well regenerated stands. The percentage of points which fell outside the 95 percent prediction intervals was computed for both graphs.

SI values for young well regenerated conifers, shrub stands, and herb stands were divided into two age groups: (1) 5 to 14 years old and (2) 15 to 24 years old. A one-way analysis of variance and the protected least significant difference (LSD) were used to test for significant differences in mean SI values between each of the three stand types (conifers, shrub stands, and herb stands) in each of the two age groups.

Stepwise multiple regression was used to examine which band combinations were useful in predicting the age of well regenerated Douglas-fir stands. TM band 4 values were also plotted against the stand age of well regenerated conifer stands because previous investigations had reported that the near infrared was sensitive to changes in forest age and biomass (Horler and Ahern, 1986; Ripple *et al.*, 1991).

Results

Correlations between stand age and individual TM band values were highest with the log-linear relationship (Table 1). Conversely, the TM band ratios and NDVI had their highest correlations to stand age with the log-log relationship. All single bands with the exception of TM band 4 ($r = -0.01$) had high correlations with stand age ($r = -0.95$ to -0.87). Overall, the log-log relationships of stand age with SI ($r = 0.96$) and stand age with TM 4/7 ($r = 0.95$) had the highest correlations. Among the Tasseled Cap features, the log-linear relationship of stand age with wetness had the highest correlation ($r = 0.95$). NDVI and TM 4/3 had lower correlations to stand age ($r \leq 0.84$) than all single bands except for TM 4 and TM 5.

The SI had a direct curvilinear relationship to stand age in well regenerated stands (Figure 1). TM bands 1, 2, 3, 5, 6, and 7 had inverse curvilinear relationships to stand age. The relationship between SI values and age for the poorly regenerated stand subgroups (1) herb stands and (2) shrubs stands are shown (Figures 2 and 3, respectively). Fifty-two percent of the poorly regenerated stands with a herb understory fell outside the prediction intervals, and 57 percent of the poorly regenerated stands with a shrub understory fell outside the

TABLE 1. SUMMARY OF THE RELATIONSHIP BETWEEN THE AGE OF WELL REGENERATED DOUGLAS-FIR (*PSEUDOTSUGA MENZIESII*) STANDS, AND 7 TM BANDS, AND 7 BAND TRANSFORMATIONS. ALL P-VALUES ARE SIGNIFICANT AT 0.0000 EXCEPT WHERE NOTED (A = 0.2734, B = 0.9206, C = 0.9949, D = 0.1728, E = 0.0083). INCLUDED ARE BOTH LINEAR, LOG-LINEAR, AND LOG-LOG CORRELATIONS BETWEEN STAND AGE AND BAND VALUES. THE LOG-LINEAR RELATIONSHIP IS THE LOG OF STAND AGE VERSUS BAND VALUES. LOG REFERS TO THE NATURAL LOGARITHM.

TM Band or Band Transformation	Correlation Coefficient (r) linear	Correlation Coefficient (r) log-linear	Correlation Coefficient (r) log-log
TM 1 (0.45 − 0.52 μm)	−0.82	−0.90	−0.89
TM 2 (0.52 − 0.60 μm)	−0.85	−0.86	−0.85
TM 3 (0.63 − 0.69 μm)	−0.86	−0.95	−0.93
TM 4 (0.76 − 0.90 μm)	−0.14[a]	−0.01[b]	−0.00[c]
TM 5 (1.55 − 1.75 μm)	−0.84	−0.87	−0.80
TM 6 (10.4 − 12.5 μm)	−0.80	−0.89	−0.89
TM 7 (2.08 − 2.35 μm)	−0.86	−0.93	−0.89
NDVI [(TM 4−3)/(TM 4+3)]	0.67	0.83	0.84
TM 4/3	0.61	0.72	0.80
Structural Index (TM 4/5)	0.90	0.93	0.96
TM 4/7	0.90	0.91	0.95
Brightness	−0.71	−0.68	------
Greenness	0.17[d]	0.34[e]	------
Wetness	0.86	0.95	------

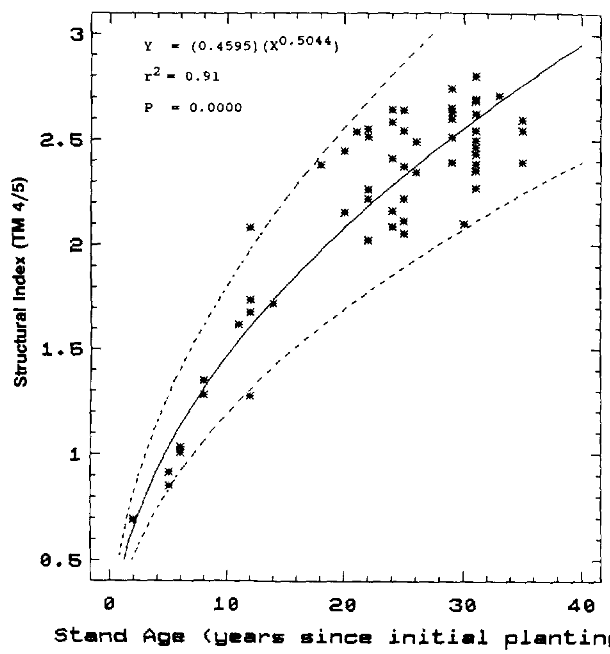

Figure 1. The relationship between the Structural Index, SI (TM 4/5 ratio) and young well regenerated Douglas-fir (*Pseudotsuga menziesii*) stands (n = 61 from 45 stands). A prediction interval (95 percent) for new observations is also shown

prediction intervals. Ninety-one percent of the herb stand observations which were greater than 12 years old fell outside the prediction intervals for well regenerated stands. Seventy-five percent of the shrub stands observations which were

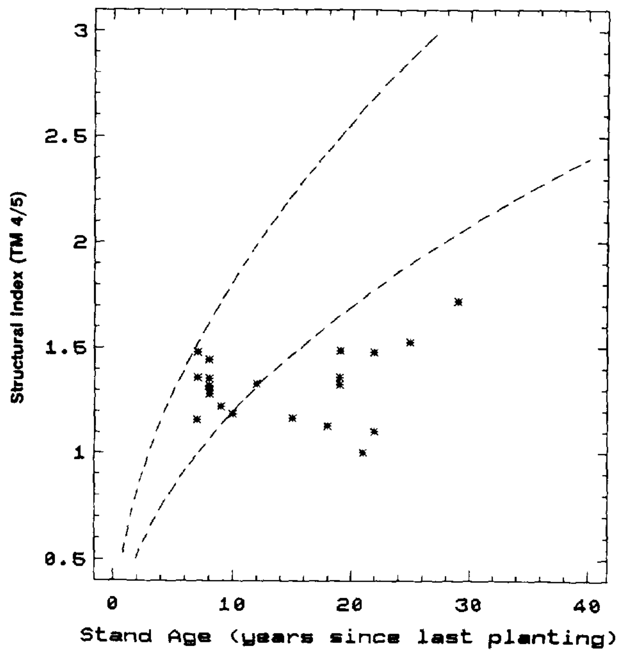

Figure 2. The relationship between the Structural Index, SI (TM 4/5 ratio) and the age of poorly regenerated conifer stands where the understory is predominately herbs (herb stands) (n = 21 from 20 stands). The prediction interval from the regression of well regenerated stands (Figure 1) is included to illustrate the differences between herb stands and well regenerated stands. Age is determined from the most recent planting date.

greater than 16 years old fell outside the prediction intervals for well regenerated stands.

The one way analysis of variance results showed that there were no significant differences in SI values among well regenerated conifer stands, shrub stands, and herb stands in the 5 to 14 year age group (P = 0.1376), but that there were significant differences among these stand types in the 15 to 24 year age group (P = 0.0000). The results from protected LSD method for the comparison of means showed that all three stand types were significantly different in the 15 to 24 year age group (α = 0.05).

In the stepwise multiple regression of the original TM band values and log of stand age, TM 3 and TM 4 were the only two independent variables included in the model (adjusted r^2 = 0.91). The simple regression of TM 3 on log of stand age explained slightly less variance (adjusted r^2 = 0.89). In the stepwise multiple regression of the log of individual TM bands on the log of stand age, TM 3, 4, and 5 were all included in the model (adjusted r^2 = 0.92). The simple regression of the log of SI on log of stand age explained approximately the same amount of variance (adjusted r^2 = 0.91).

A scatter plot of TM band 4 band values and stand age of well regenerated stands indicated that before the stands reach age 18, TM band 4 had a direct linear relationship with stand age (r^2 = 0.54) (Figure 4). After age 18, TM band 4 had little relationship with stand age (r^2 = 0.02). Well regenerated stands reached 60 percent canopy closure approximately 18 years after planting.

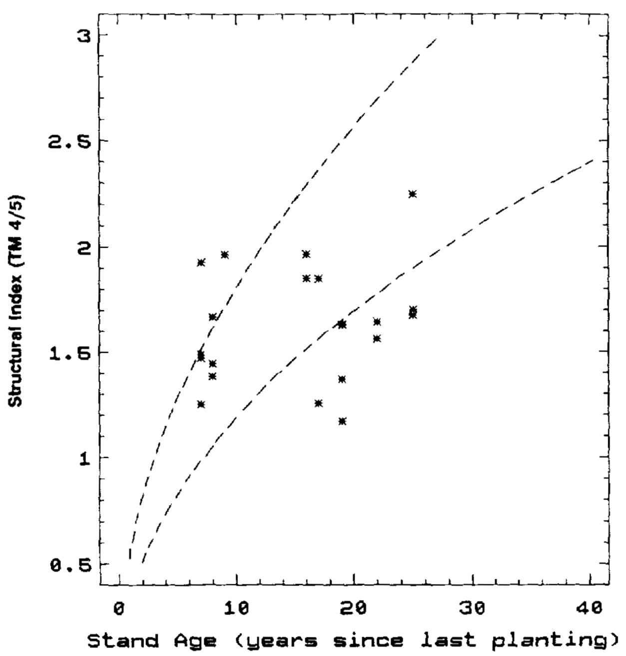

Figure 3. The relationship between the Structural Index, SI (TM 4/5 ratio) and the age of poorly regenerated conifer stands where the understory is predominately shrubs and/or deciduous trees (shrub stands) (n = 21 from 16 stands). The prediction interval from the regression of well regenerated stands (Figure 1) is included to illustrate the differences between shrub stands and well regenerated stands. Age is determined from the most recent planting date.

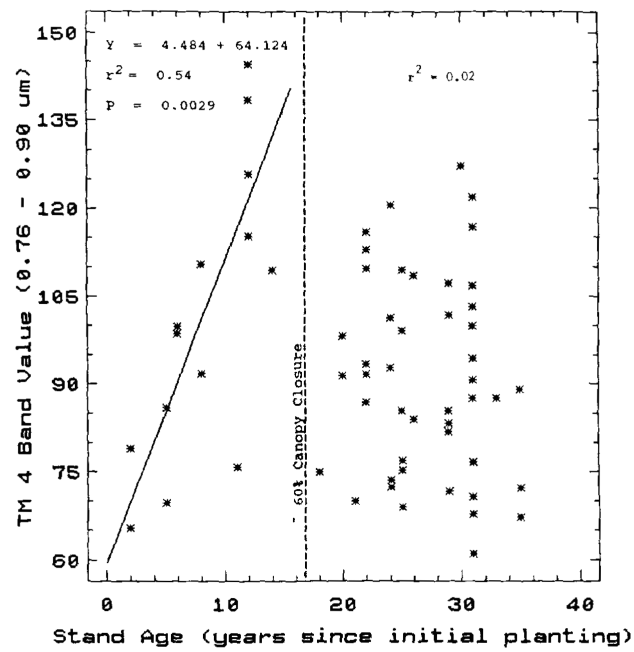

Figure 4. The relationship between TM band 4 and stand age for well regenerated Douglas-fir stands (n = 61 from 45 stands). Prior to age 18 and 60 percent canopy closure, TM band 4 had a direct linear relationship with stand age (r^2 = 0.54). After age 18 and at greater than 60 percent canopy closure, TM band 4 had little relationship with stand age (r^2 = 0.02).

Discussion

With the exception of TM 4, all TM bands showed a strong inverse correlation with stand age. Because stand age was closely tied to canopy closure in well regenerated stands, it is not surprising that both the visible and middle infrared bands were well correlated with stand age. As leaf area and biomass increased with stand age, the absorption of energy by plant pigments and moisture also increased (Butera, 1986; Spanner *et al.*, 1989). Other important factors include the decrease in the amount of bright understory vegetation exposed to the sensor as the conifer canopy closed, and the increase in shadowing from the growing conifer crowns.

The correlation of SI to stand age showed improvement over single bands and even over the TM 4/3 ratio and NDVI. The usefulness of the SI for estimating forest age is similar to the results obtained by Spanner *et al.* (1989) where forest age was described in terms of disturbance/successional stage classes. Fiorella and Ripple (1993) found that the SI and wetness, and Cohen and Spies (1992) found that wetness, were useful transformations for estimating structural attributes of older Douglas-fir stands, in separating mature from old-growth forests, and in reducing topographic or shadowing influences.

From our studies and the work by Spanner *et al.* (1989), it appears that the SI has very good potential for estimating structural characteristics and successional stages in both young and old conifer forests by reducing topographic effects. It should be noted, however, that SI values for old-growth can be similar to young forests that have not reached canopy closure (Spanner *et al.*, 1989; Fiorella and Ripple, 1992, unpublished data). With simple regression, the SI accounted for approximately the same amount of variance in stand age as the results from stepwise multiple regression of individual spectral bands. The model developed for the relationship between stand age and SI should be tested with an independent data set to determine if the model is as strong as it appears to be.

TM 4 had a weak relationship with stand age over the entire range of ages (2 to 35 years) in this study, but near-infrared bands have been found to have a strong relationship to structure in older forests (Ripple *et al.*, 1991b; Eby, 1987). When TM 4 values were plotted against stand age, it became clear that TM 4 has two different relationships with stand age. Prior to canopy closure, TM band 4 values increased with age. This rise in TM 4 values may be due to increased scattering of radiation with the increase in vegetation amount as succession proceeded from herbs to shrubs (Spanner *et al.*, 1989). Horler and Ahern (1986) found similar results in western Ontario, Canada in that TM band 4 values increased with age in 0 to 12 year old conifer plantations. After canopy closure, at greater than 60 percent closure, there was little relationship between TM 4 and stand age in our study. The variability in TM band 4 values after canopy closure may be due to offsetting influences of increasing biomass (increased brightness) and increasing conifer canopy shadowing (decreased brightness), and variability in the amount of broadleaf vegetation (Ripple *et al.*, 1991b).

The results of our study indicate that differences in mean SI values between poorly regenerated and well regenerated stands were significant after age fifteen. Although it would be difficult to use TM satellite data to assess regeneration in Douglas-fir plantations less than 15 years old, the success in identifying poorly regenerated stands should be high after this initial period. The length of time required to find poorly regenerated stands may decrease if differences in site preparation, aspect, and planting density were accounted for. This length of time will also be dependent upon site productivity and how long it takes the conifer canopy to develop and dominate a site.

TM satellite data may also be very useful in identifying successional stage for wildlife habitat analysis. Herb and shrub successional stages are important habitat and forage areas for some wildlife species. Successful reforestation methods have reduced the time to reach a closed canopy condition, and consequently reduced the time a stand spends in herb and shrub stages. In a landscape context, these poorly regenerated stands can be important for enhancing wildlife and plant biodiversity.

Conclusions

Based on the results of this study, we derived the following conclusions:

- With the exception of TM 4, all bands showed a strong inverse correlation with age of young Douglas-fir stands (2 to 35 years old). This was attributed to decreasing amounts of the bright understory exposed to the sensor, increased shadowing from the conifers, and increased absorption of radiation by pigments for the visible bands and moisture for the middle infrared bands.
- The near-infrared band, TM 4, showed a direct relationship to stand age before canopy closure (2 to 18 years) and a weak relationship to stand age after canopy closure (18 to 35 years). This weak relationship was attributed to the variability in offsetting influences of increasing biomass, increasing shadowing, and variability in the understory.
- The SI had a very strong relationship with stand age and should be tested in future studies involving the analysis of vegetation structure.
- TM spectral data from poorly regenerated stands were significantly different from well regenerated stands only after the plantations reached an age of approximately 15 years in the Cascade Mountains of Oregon. This length of time (15 years) would be shorter in areas of higher site productivity and longer in areas of lower site productivity and related to the rate of development of the young conifer crowns.

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Determining Successional Stage of Temperate Coniferous Forests with Landsat Satellite Data

Abstract

Thematic Mapper (TM) digital imagery was used to map forest successional stages and to evaluate spectral differences between old-growth and mature forests in the central Cascade Range of Oregon. Relative sun incidence values were incorporated into the successional stage classification to compensate for topographic induced variation. Relative sun incidence improved the classification accuracy of young successional stages, but did not improve the classification accuracy of older, closed canopy forest classes or overall accuracy. TM bands 1, 2, and 4; the normalized difference vegetation index (NDVI); and TM 4/3, 4/5, and 4/7 band ratio values for old-growth forests were found to be significantly lower than the values of mature forests ($P \leq 0.034$). The Tasseled Cap features of brightness, greenness, and wetness also had significantly lower old-growth values as compared to mature forest values ($P \leq 0.010$). Wetness and the TM 4/5 and 4/7 band ratios all had low correlations to relative sun incidence ($r^2 \leq 0.16$). The TM 4/5 band ratio was named the "structural index" (SI) because of its ability to distinguish between mature and old-growth forests and its simplicity.

Introduction

The identification of successional stages in the conifer forests of the Pacific Northwest region of the U.S.A. is important to both forest managers and wildlife biologists. Current information on the location and distribution of all forest ages and structures is needed to manage public lands for multiple use objectives. Recently, identifying remaining stands of old-growth forest has been highlighted (Ripple *et al.*, 1991a). However, information on younger stand development is necessary and critical in determining future timber supply and wildlife habitat (Brown, 1985; Harris, 1984).

Satellite imagery has been used in the past to classify Northwest forests. Landsat Multispectral Scanner (MSS) data have been used in western Washington to identify old-growth forests (Eby, 1987), and to identify forest species groups (Cibula, 1987). Landsat Thematic Mapper (TM) data have been used in northern California to quantify stand structural characteristics of basal area and foliage biomass (Franklin, 1986). Recent studies in western Oregon and Washington have employed TM data to map old-growth forests (Green and Congalton, 1990); to relate the structural attributes of young, mature, and old-growth stands to the TM Tasseled Cap indices (Cohen and Spies, 1992); and to measure stand volume and basal area (Ripple *et al*, 1991b).

Distinguishing old-growth from mature forests has been difficult because both successional stages tend to have large trees, and high basal and leaf areas. Most forest stand parameters such as biomass, leaf area index, volume, and, in general, vegetation amount have asymptotic relationships to single band spectral data beginning at moderate to high levels of these stand parameters (Ripple *et al.*, 1991b; Spanner *et al.*, 1990a; Horler and Ahern, 1986). Differences in old-growth and mature forests are determined by a combination of overstory and understory structural and compositional factors from ground based surveys. However, remote sensing data primarily measure only canopy overstory characteristics.

Two of the most distinguishing features observed at the canopy level are differences in the number and size of gaps in the forest canopy and the heterogeneity of tree sizes (Spies *et al.*, 1990; Spies and Franklin, 1991). In general, old-growth canopy gaps tend to be horizontally larger ($85m^2$ versus $19m^2$), but less numerous than those characteristic in mature stands (Spies *et al.*, 1990). Old-growth forests also have a greater range of tree sizes (Spies *et al.*, 1990). Both of these features create dark shadows in the old-growth forest canopies which contrast sharply with sunlit tree crowns.

Identifying forest successional stages in dissected mountainous terrain is complicated due to the dark shadowing on steep north-facing slopes and the high variability in illumination conditions. Eby (1987) used Landsat MSS near-infrared band 4 imagery to identify old-growth forest stands with 80 percent accuracy. Sun incidence angle was used to stratify the study area into normally illuminated and shaded areas for post-classification sorting. Walsh (1987) found that terrain orientation in southern Oregon mountains was often more important in determining TM band values than either crown size or crown density. We hypothesized that reflectance of forest stands in the Cascade Range of Oregon would also be significantly influenced by terrain orientation due to the steep elevational gradients and sharply dissected valleys. Previous studies also indicate that both sun angle (Guyot *et al.*, 1989; Conese *et al.*, 1988; Leprieur *et al.*, 1988; Pinter *et al.*, 1985), and plant architecture or canopy structure can influence measured reflectance (Williams, 1991; Guyot *et al.*, 1989; Pinter *et al.*, 1985). Slope, sun angle, plant architecture, and canopy structure can determine whether vegetation response to incident sun light is best modeled by a Lambertian or non-Lambertian model (Jones *et al.*, 1988; Leprieur *et al.*, 1988; Hugli and Frei, 1983; Teillet *et al.*, 1982; Smith *et al.*, 1980).

Primary objectives of this study were (1) to develop a successional stage map for use in wildlife habitat analysis, (2) to compare the spectral characteristics of old-growth and mature forests, and (3) to evaluate the topographic influence

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on spectral signatures and the usefulness of sun incidence data in mapping forest successional stage.

Methods

Study Site

The H.J. Andrews Experimental Forest study site is located in the Central Cascade Range of Oregon. Elevations range from 414 to 1630 metres above mean sea level (MSL). The study area falls within the western hemlock (*Tsuga heterophylla*) zone (Franklin and Dyrness, 1973), and the dominant tree species in this zone is Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco.), a subclimax species. Western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) is a common understory or codominant species in older stands. Above 1100 metres, the study area falls into the Pacific silver fir (*Abies amabilis*) zone (Franklin and Dyrness, 1973). These higher elevation forests are dominated by noble fir (*Abies procera* Rehd.) or Pacific silver fir (*Abies amabilis* (Dougl.) Forbes).

Timber harvesting in the Experimental Forest began in 1950. These replanted, managed stands represent approximately 25 percent of the forest landscape. The slope-aspects in the study area are predominately north, northwest, south, and southeast, and slope-aspect determined moisture gradients are evident. South facing slopes tend to be drier and more susceptible to wildfires than are north facing slopes (Teensma, 1987).

Computer-Assisted Classification and Mapping

An area including the H.J. Andrews Experimental Forest was extracted from a 30 July 1988 TM quarter-scene (scene ID Y5161218271). The TM data were rectified to a Universal Transverse Mercator (UTM) grid using a nearest neighbor resampling method.

Preliminary classification results indicated that topographic shadowing was an important factor in mapping successional stages. Eby's (1987) study indicated that the use of sun incidence angle could reduce confusion between mature and old-growth forests due to topography (Eby, 1987). Thus, a relative sun incidence band was used with the six TM reflective bands in the classification to evaluate its utility in compensating for the unequal illumination on southern versus northern slopes.

Relative sun incidence was calculated from a 1:250,000-scale digital elevation model (DEM) data using the ERDAS RELIEF program (Figure 1). This program calculates direct illumination which is equal to the cosine of the angle between the surface normal and the incident beam for each pixel. The output values of -1 to 1 were rescaled from 0 to 255. The sun incidence band was resampled to a 30- by 30-metre cell size using a nearest neighbor resampling method to register it to the TM data. These data represented the relative amount of incident sunlight to a surface based on sun azimuth and sun elevation at the time of the satellite overpass, but did not consider atmospheric influences or diffuse illumination of adjacent slopes. The sun incidence model assumed a uniform Lambertian behavior of all vegetation types. The model was not used to correct band values, but was used to guide spectral class formation by further dividing classes by relative illumination levels. Two spectral classes could potentially have the same true spectral vector, but could be distinguished based on illumination levels.

All six TM reflective bands (1,2,3,4,5, and 7) were used in two different classifications. The first classification used only the six TM bands and the second classification used the six TM bands plus the relative sun incidence band. An unsupervised, iterative self-organizing data analysis technique (ISODATA) was used to develop 99 spectral classes. The spectral signatures generated with ISODATA provided the input to a maximum-likelihood classifier. Ancillary data, which included the locations of old-growth and mature forest reference plots, and 1988, 1:12,000- and 1972, 1:20,000-scale true color aerial photographs were used in assigning class groupings. The output classes were grouped into one of the five successional stage categories. Final maps were smoothed to remove isolated pixels using a moving 3 by 3 window and a majority rule.

Successional Stage Mapping

Five successional stage categories were defined based on wildlife habitat requirements (Brown, 1985): (1) grass-forb and shrub, (2) open sapling pole, (3) closed sapling pole and small sawtimber, (4) mature or large sawtimber, and (5) old-growth (Table 1). These successional stages provide significantly different habitats for wildlife species and are determined primarily by stand structural characteristics rather than species composition (Brown, 1985).

An accuracy assessment was performed on both classifications using a stratified random sample of pixels. Pixels were stratified by the successional stages from the classification which used relative sun incidence (Congalton, 1988). Twenty pixels were selected for classes which occupied a small area in the classified image, and 40 pixels were selected for classes which occupied a large area in the classified image. Pixels selected for accuracy assessment had to be surrounded by pixels of the same class to allow for error in locating the point on aerial photographs.

Sample selection in managed stands was limited to one sample per stand, and to the coverage of the 1988, 1:12,000-scale photographs. The sampling points were transferred from the digital successional stage map to the raw Landsat image file, and the image file was then used to locate the points on the aerial photographs. The successional stage at each accuracy point was determined by a forest ecologist who was very familiar with the area, but not familiar with either of the Landsat classifications. The same set of points was used to evaluate the classification without the sun incidence band. Error matrices with percent correct, percent commission error, and Kappa statistics (Congalton *et al.*,

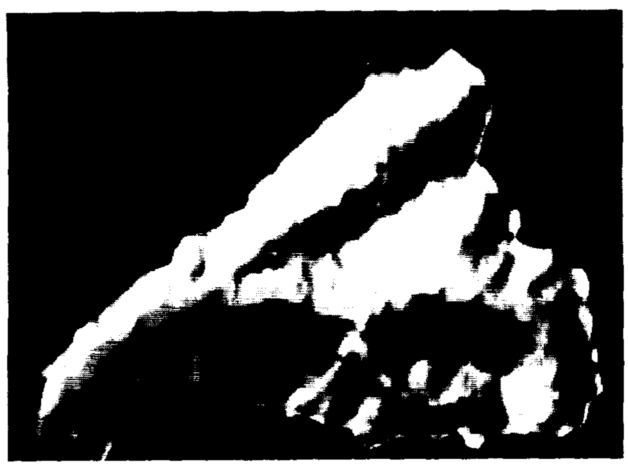

Figure 1. Shaded relief map for the H.J. Andrews. Dark areas indicate areas of low incident light, and light areas indicate areas of high incident light.

Table 1. Description and Criteria for Successional Stage Classes (Adapted from Brown, 1985)[1], DBH = Diameter at Breast Height.

Stand Condition	Description/Criteria
Grass-forb, Shrub	Grasses and/or shrubs dominate Trees < 10 ft (3.0 m) tall Conifer closure < 30% Soil, Rock 0–10 years old (± 5 years)
Open Sapling-Pole	Trees > 10 ft (3.0 m) tall Canopy closure 30-60% Trees ≥ 1 in. (2.5 cm) DBH 10–15 years old (± 5 years)
Closed Sapling-Pole Small Sawtimber	Canopy closure 60-100% (usually close to 100%) Little understory vegetation 20–90 (± 20 years) years old
Mature, Large Sawtimber	Tree mean DBH ≥ 21 in. (53.3 cm) Trees ≥ 100 ft tall 1 or 2 story stands Crown cover < 100% 90–200 (± 20 years) years old
Old-growth	Tree mean DBH ≥ 26 in. (66.0 cm) 2 or 3 story stands Crown cover < 100% Large amounts of snags and down woody debris 200+ years old (± 30 years)

[1] Ages adapted for study area.

1983) were calculated to evaluate the differences between the classifications.

Spectral Characteristics of Old-Growth and Mature Forest

Mean spectral values of a 3 by 3 window from each of 22 old-growth and 19 mature forest stands were compared to determine whether old-growth and mature forest stands were spectrally distinct. TM spectral data were extracted from locations of known U.S. Forest Service old-growth and mature forest reference plots that fell within, or were adjacent to, the H.J. Andrews Forest. Accuracy sampling points which were identified as mature forest stands, and were not already represented by the reference plots, were also used in this analysis. The Wilcoxon Rank Sum test (Devore and Peck, 1986), a non-parametric measure of the difference in the mean values, was used to determine which single TM band, TM band transformation, or Tasseled Cap feature (Crist and Cicone, 1984) provided the best separation between 22 old-growth and 19 mature forest stands.

Sun Incidence

Regression analysis was used to examine the relationship of TM band 4 band values for old-growth, mature, and young conifer forest stands, to relative sun incidence. The analysis was used to determine if these successional stages fit the Lambertian model, and to determine whether the forest canopy response differed between forest successional stages. A Lambertian model assumes that light is scattered uniformly in all directions and is determined by slope orientation with respect to the sun. Differences in land cover do not influence model values.

The old-growth and mature stand data from the rank sum test were regressed against sun incidence. Data for young stand observations were taken from managed Douglas-fir stands in the H.J. Andrews forest that had regenerated to a closed canopy condition (ages ranged from 29 to 35 years old). TM band 4 was selected for analysis because of its ability to distinguish mature and old-growth spectral values ($P = 0.0009$) and because it had the highest correlation coefficient ($r^2 = 0.78$) to relative sun incidence values of older stands. Regression lines were fitted using the reduced major axis technique to minimize error in both the x and y directions (Curran and Hay, 1986).

Results

Successional Stage Mapping

Accuracy assessment results are found in Tables 2 and 3. Both the classification performed with the relative sun incidence band, and the classification performed without the relative sun incidence band, had the identical overall percent accuracy (78.3). The highest level of confusion was found between mature and old-growth forests. In both classifications, the mature category had both the lowest percent correct (69) and the highest percent commission error (55) of any category. The classification with sun incidence had higher mapping accuracy for both the grass-forb, and shrub, and open sapling-pole classes (85 and 81 percent, respectively) than the classification performed without sun incidence (80 and 69 percent, respectively).

Spectral Characteristics of Old-Growth and Mature Forest

All mean TM band values for old-growth forest were lower than those of mature forest (Table 4). The nonparametric test (Wilcoxon Rank Sum, Table 4) for the difference in mean old-growth and mature forest stand band values showed that wetness ($P = 0.00003$) and the TM 4/5 ratio ($P = 0.00005$) were the best band transformations, and TM band 4 ($P = 0.0009$) was the best single TM band for distinguishing between these two successional stages. Thematic Mapper bands

Table 2. Error Matrix for the Classification with Six TM Bands (1,2,3,4,5,7)

	REFERENCE DATA					
TM DATA	Grass-forb, Shrub	Open Sapling-Pole	Closed Sapling-Pole-Small Sawtimber	Mature, Large Sawtimber	Old-Growth	Percent Commission Error
Grass-forb, Shrub	16	2	0	0	0	11
Open Sapling-Pole	3	11	1	0	0	27
Closed Sapling-Pole Small Sawtimber	1	2	21	1	2	22
Mature, Large Sawtimber	0	1	3	9	7	55
Old-Growth	0	0	0	3	37	8
Percent Correct	80	69	84	69	80	78.3

Overall Percent Correct = 78.3
Kappa = 0.717

1 and 2; the normalized difference vegetation index (NDVI); and the TM 4/3, TM 4/7, brightness, and greenness band transformations were also highly significant ($P \leq 0.0341$). Correlations between relative sun incidence and wetness, TM 4/5 band ratio, and TM 4/7 band ratio were lower (r = 0.16, 0.14, and 0.10, respectively) than those to any other band or band transformation (r ranged r = 0.49 to 0.80 for all other bands; and r = 0.71, 0.78, 0.67, and 0.69 for TM 3, TM 4, NDVI, and TM 4/3 ratio, respectively).

Sun Incidence

The mean TM 4 band values for old-growth, mature, and young forest stands are shown with their respective incident light levels (Figures 2, 3, and 4, respectively). Mature forest stands had a higher correlation ($r^2 = 0.74$) with relative incident light than old growth forest stands ($r^2 = 0.39$) or young stands ($r^2 = 0.32$). Young forests had a steeper regression slope ($0.768X$) than either the mature stands ($0.284X$) or the old-growth stands ($0.255X$).

Discussion

Successional Stages

The relative sun incidence band improved classification accuracy in younger successional stages (grass-forb and shrub, and open sapling pole), but did not improve classification accuracy for older successional stages. In both the grass-forb and shrub, and open sapling pole stands conditions, the understory vegetation (i.e., grass, herbs, and shrubs) rather than the conifer canopy has the greatest affect on band values (Franklin, 1986; Spanner *et al.*, 1990a). The broad leaf species that dominate these earlier seral stages may be better represented by a Lambertian surface, and therefore better modeled by the relative sun incidence band than are the older stands. The lack of improvement in classification accuracy of older successional stages in the classification with sun incidence may be explained in part by the non-Lambertian characteristics of these stands.

Jones *et al.* (1988) found vegetation type (deciduous forest, coniferous forest, agriculture) determined whether a Lambertian or non-Lambertian sun incidence model was more successful in correcting illumination problems in SPOT HRV multispectral imagery. Deciduous vegetation had higher percent accuracy with the Lambertian model, while conifer vegetation had higher percent accuracy with a non-Lambertian model. The non-Lambertian model was a Minnaert reflectance model where the surface exhibits less than Lambertian characteristics.

Canopy architecture can also affect band values in that planophile canopies tend to have higher band values than do erectile canopies, and may more likely fit the Lambertian reflectance model (Pinter *et al.*, 1985). Deciduous vegetation tends to be more "planophile" or horizontally flat than coniferous vegetation. Because deciduous plants dominate younger successional stages, the Lambertian sun incidence model may be better suited to these successional stages.

Spectral Characteristics of Old-Growth and Mature Forests

The interpretation of the mature and old-growth forest rank sum results requires attention on two points: (1) old-growth and mature forests can be very similar, and (2) mainly overstory features are directly measured by remote sensing data. Old-growth forests have larger canopy gaps and a greater heterogeneity of tree sizes than do mature forests. Both the large gaps and tree size heterogeneity create dark shadows in old-growth canopies which are in sharp contrast to the sunlit tree crowns. Spectral bands which accentuate the high contrast between pixels dominated by shadows and gaps from pixels dominated by tree crowns are likely to best distinguish old-growth and mature forests.

While not all single TM bands significantly differentiated between old-growth and mature forests, the mean TM values for all old-growth stands was always lower than that for all mature stands. Old-growth Douglas-fir forests have higher leaf area per unit ground area than mature forests of 90 to 130 years old (Franklin *et al.*, 1981). Douglas-fir forests can continue to accumulate live biomass until 400 to 500 years of age and maintain those levels without declining appreciably through 700 to 900 years of age (Franklin and Spies, 1988). Because of spectral asymptotes, it is unlikely that higher leaf areas and live biomass accounted for lower band values for old-growth forests in absorption bands (bands 1, 2, 3, 5, and 7) as compared with mature forest values. Band values become asymptotic above LAIs of approximately 6 (Spanner *et al.*, 1990b), and old-growth LAIs are normally higher than 6 in this location (Franklin *et al.*, 1981). Lower old-growth values were most likely due to shadowing from the uneven tree sizes and the high number of large canopy gaps in old-growth forests.

Also, the fire history of the study area indicates that the majority of the wild fires of the 1800s occurred on the drier, south facing slopes (Teensma, 1987). These fires were responsible for the establishment of the mature and natural closed canopy stands in the study area. Because most of the mature forests are located on south facing slopes, these forests are highly illuminated at the time of the Landsat overpass and would subsequently have higher band values due to topographic influences. Rank sum test results for differences

Table 3. Error Matrix for the Classification with Six TM Bands (1,2,3,4,5,7) and the Relative Sun Incidence Band.

	REFERENCE DATA					
TM DATA	Grass-forb, Shrub	Open Sapling-Pole	Closed Sapling-Pole-Small Sawtimber	Mature, Large Sawtimber	Old-Growth	Percent Commission Error
Grass-forb, Shrub	17	2	1	0	0	15
Open Sapling-Pole	3	13	4	0	0	35
Closed Sapling-Pole Small Sawtimber	0	1	19	0	0	5
Mature, Large Sawtimber	0	0	1	9	10	55
Old-growth	0	0	0	4	36	10
Percent Correct	85	81	76	69	78	78.3

Overall Percent Correct = 78.3
Kappa = 0.718

TABLE 4. RESULTS OF A NONPARAMETRIC TEST (WILCOXON RANK-SUM) FOR THE DIFFERENCE BETWEEN OLD-GROWTH (N = 22) AND MATURE (N = 19) FOREST MEAN VALUES OF SEVEN TM SPECTRAL BANDS, SEVEN BAND TRANSFORMATIONS, AND RELATIVE SUN INCIDENCE. STANDARD ERROR (SE) VALUES ARE IN PARENTHESES.

TM Band/Index	Mature $\bar{X}$ (SE)	Old-Growth $\bar{X}$ (SE)	Two Tailed Probabilities
TM 1 (0.45 − 0.52μm)	60.03 (0.348)	59.02 (0.331)	0.0341
TM 2 (0.52 − 0.60μm)	21.00 (0.252)	20.20 (0.127)	0.0103
TM 3 (0.63 − 0.69μm)	16.84 (0.217)	16.47 (0.149)	0.2597
TM 4 (0.76 − 0.90μm)	60.51 (1.961)	51.29 (1.297)	0.0009
TM 5 (1.55 − 1.75μm)	28.04 (1.071)	26.52 (0.656)	0.3333
TM 6 (10.4 − 12.5μm)	134.28 (0.637)	134.20 (0.409)	0.9686
TM 7 (2.08 − 2.35μm)	7.74 (0.349)	7.61 (0.208)	0.7633
NDVI [(TM 4 − 3)/(TM 4 + 3)]	0.56 (0.008)	0.51 (0.007)	0.0001
Ratio TM 4/3	3.58 (0.083)	3.11 (0.065)	0.0004
Ratio TM 4/5 (SI)	2.17 (0.040)	1.94 (0.024)	0.00005
Ratio TM 4/7	7.93 (0.211)	6.78 (0.143)	0.0004
Relative Sun Incidence	214.84 (6.904)	197.67 (5.080)	0.0917
Brightness	81.46 (1.874)	74.80 (1.181)	0.0100
Greenness	13.48 (1.228)	7.41 (0.869)	0.0010
Wetness	15.87 (0.426)	13.44 (0.230)	0.00003

in relative sun incidence on old-growth and mature forests supports this theory ($P = 0.0917$). Walsh (1987) found similar results in that aspect was often more important in determining TM band values than crown size or crown density.

Rank sum results indicate that TM bands 1, 2, and 4 showed significant differences between old-growth and mature forests, whereas TM bands 3, 5, 6, and 7 did not. Previous studies indicate that TM bands 3, 5, and 7 are sensitive to vegetation amount, but are not useful for distinguishing between moderate to high values of LAI or biomass (Spanner *et al.*, 1990a; Horler and Ahern, 1986). Old-growth and mature stands both have high vegetation amounts and all pixels, whether in gaps or in mature or old-growth canopies, will be relatively dark.

TM bands 1, 2, and 4 may highlight differences between gaps and forest. This contrast results from mixed pixels dominated by either gaps and shadows (dark) or trees (light). TM band 1 has been found to be able to distinguish between healthy and defoliated conifer canopies where living and dead parts of the canopy are contrasted (Nelson *et al.*, 1984).

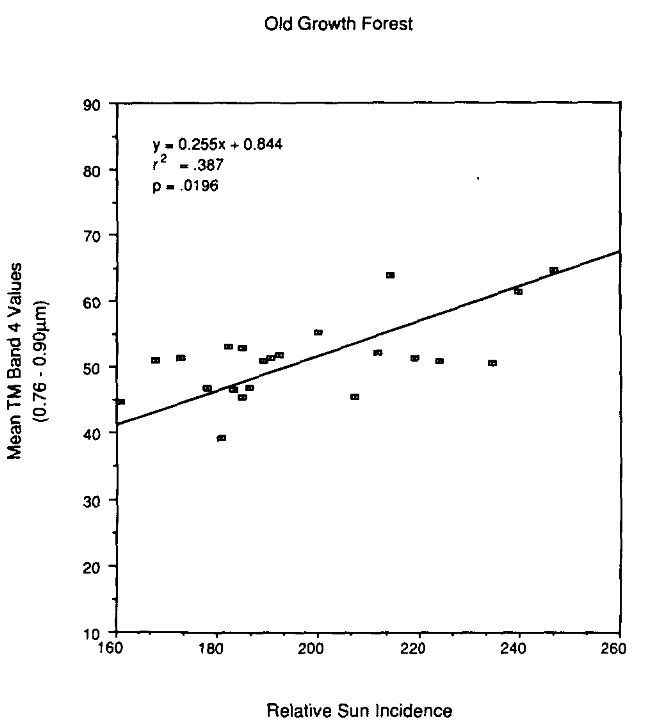

Figure 2. Relationship between mean near-infrared band TM4 values for old-growth forests and the relative incident sunlight on those stands. Higher sun incidence values indicate higher illumination levels ($n = 22$).

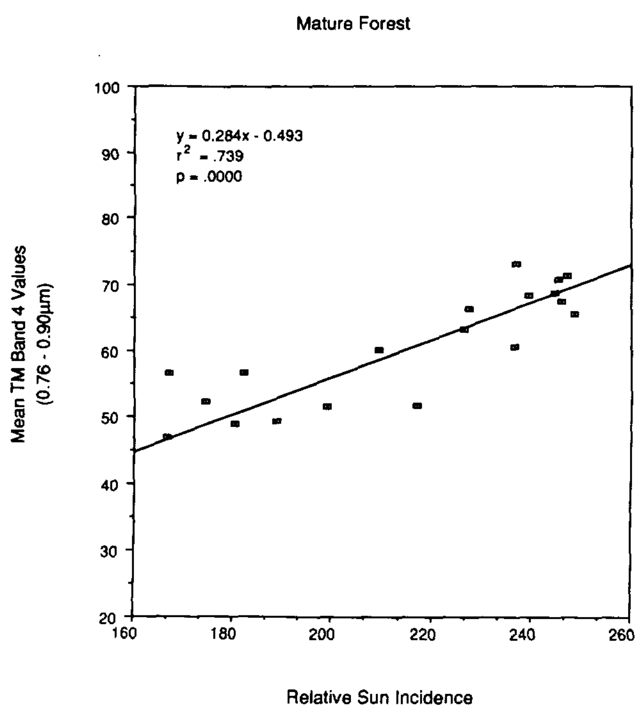

Figure 3. Relationship between mean near-infrared band values for mature forests and the relative incident sunlight on those stands. Higher sun incidence values indicate higher illumination levels ($n = 19$).

In a similar way, dark canopy gaps and sunlit crowns may be contrasted. TM band 4 values generally increase with vegetation amount due to the scattering of light by internal leaf structure (Knipling, 1970). High TM 4 band values on sunlit crowns should contrast sharply with dark shadows in old-growth forests. In mature forests, overall TM 4 band should be bright because sunlit crowns dominate. NDVI and the TM 4/3 band ratio were also able to distinguish between mature and old-growth forests. The utility of these band transformations for spectral discrimination was probably due to the inclusion of TM band 4 and the reduction in topographic induced variation.

Cohen and Spies (1992) found that wetness was the best Tasseled Cap feature for distinguishing old-growth from mature forest stands, while brightness and greenness did not appear to distinguish between these two forest types. In this study, brightness and greenness did separate old-growth from mature forests, but wetness was more highly significant than either of these. Wetness was also better than all other single TM bands and most band ratios. TM 4/5 had a similar significance level ($P = 0.00005$) to wetness, and was also highly correlated to wetness ($r^2 = 0.97$). Although the Tasseled Cap wetness index is a contrast between TM bands 1 through 4 versus TM bands 5 and 7, it appears that, for older forests, the significant feature in wetness may be the contrast between TM 4 versus TM 5. Therefore, we suggest using TM 4/5 rather than wetness because it is simpler to calculate and interpret. The TM 4/5 was named the "structural index" (SI) because we have also found the TM 4/5 ratio to be an excellent predictor of stand age in young managed Douglas-fir forests.

Correlations between SI and stand age ($r = 0.96$, $n = 61$) were higher than all individual bands and six multiband transformations, including the Tasseled Cap indices (Fiorella and Ripple, unpublished data). Figure 5 shows the response of TM 4, TM 5, and SI on a natural forest in the Three Sisters Wilderness, an area adjacent to the H.J. Andrews forest. This figure illustrates how much of the topographic shadowing effect is removed with the SI.

Wetness, SI, and TM 4/7 band ratios all had low correlations with relative sun incidence. Cohen and Spies (1992) found that the topographic effect was minimized in wetness, and it appears that this is also the case for SI and TM 4/7 based on correlations of these band ratios to relative sun incidence. In both this study and the study by Cohen and Spies (1992), old-growth forests had lower mean wetness values than did mature forests. Cohen and Spies (1992) found that lichen, bark, and wood all have lower wetness values than sunlit Douglas-fir, western hemlock, and western red cedar canopies. They hypothesized that the lower wetness values for old-growth forest was due to the increase of lichen, snags, and broken topped trees in old-growth forests as compared with mature forests.

Sun Incidence

Results from the successional stage classifications indicate that the inclusion of sun incidence as an extra spectral band did not significantly improve overall classification results, and, in fact, decreased the mapping accuracy of older forests. One explanation for the decreased accuracy observed for young, mature, and old-growth conifer stands is that each

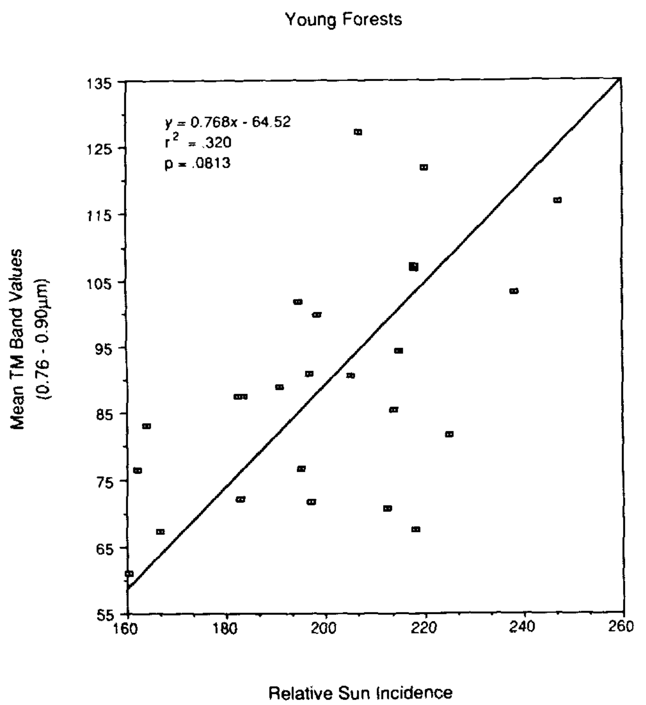

Figure 4. Relationship between mean near-infrared band values for young closed canopy forests and the relative incident sunlight on those stands. Higher sun incidence values indicate higher illumination levels ($n = 25$).

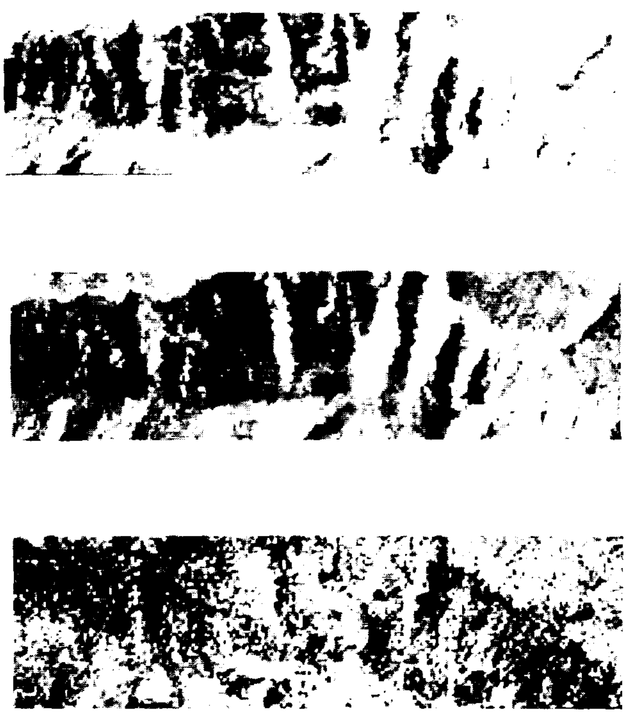

Figure 5. TM 4 (top), TM 5 (middle), and TM 4/5 ratio, the structural index (SI) (bottom), for a 1650-metre by 6000-metre natural forest area adjacent to the H.J. Andrews forest in the Three Sisters Wilderness.

had a different response, as measured by slope, to increased incident light.

There was a general decrease in regression slopes between relative sun incidence and TM band 4 values from young to old-growth forests. These differences could be due to changes in canopy structure. The relatively minor change in regression slope between old-growth and mature successional stages is reasonable, because these successional stages represent a continuum of forest development, where old-growth overstory structure is much closer to mature forests than mature forests are to young forests. In young closed canopy forests there are many tiny crowns packed tightly together, and few, if any, large gaps. More incident light to these canopies results in higher TM band 4 values.

As the stand matures, tree crowns get larger, and spaces between crowns become more pronounced. In old-growth forests the canopy gaps and uneven canopy structure trap much of the incident light, and old-growth forest canopies appear dark whether or not they are highly illuminated. While a Lambertian model may be appropriate for young stands, old-growth canopies as well as mature canopies may require a non-Lambertian model which accommodates canopy gap structure. Further research should be conducted to determine the best model.

The second observation that indicates forest canopy response differs between successional stages is that the r^2 values between TM band 4 and relative sun incidence values for the three successional stages were not equal. TM 4 values from old-growth had low correlations with sun incidence due to the heterogeneous nature of their canopies and because they are dark regardless of the illumination angle. As discussed previously, old-growth forest band values do not always respond evenly to increases in incident light. Mature forests have gaps which are much smaller than the size of a pixel, and generally have even canopies. Because mature forest canopy variability is low, the response to incident sunlight was more predictable. One would also expect young, closed canopy stands to have a high correlation with incident light. While all these stands have at least 95 percent canopy closure, they are plantations and may have been managed differently (different planting densities, thinned versus unthinned, competition with shrubs).

Eby (1987) found that correlations between sun incidence angle (angle normal to the surface) and MSS infrared band 4 were highest for old-growth stands ($r = -0.82$), and that the correlations decreased with younger stands (mature, $r = -0.77$; small sawtimber, $r = -0.44$). He hypothesized that the contrast caused by the complex structure of older stands would be enhanced on slopes with higher sun angles. The difference in spatial resolution between MSS and TM data may account in part for these dissimilar results.

Conclusions

TM data were useful in mapping successional stages. Other conclusions from this study are

- The mean TM band value for all old-growth stands was lower than the mean value for all mature stands due to (a) the many large dark canopy gaps and shadows in old-growth stand, and (b) fire history which shows that mature forests are more likely to be found on south facing slopes which are highly illuminated during the Landsat overpass.
- Including relative sun incidence only improved the classification accuracy of successional stages with minimal or no conifer canopy, probably because the Lambertian model was more appropriate for the understory vegetation.
- Conifer vegetation structure affected the response to incident light. Younger forest successional stages tend to respond more like a Lambertian surface than do older forest canopies.
- A reflectance model that accounts for different vegetation responses may be more appropriate than a pure Lambertian model.
- Band transformations that contrast near-infrared values with middle infrared values such as SI (structural index = TM 4/5) and wetness appear to decrease topographic shadowing and thereby permit the detection of spectral differences between old-growth and mature forest types.

Acknowledgments

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Determining Coniferous Forest Cover and Forest Fragmentation with NOAA-9 Advanced Very High Resolution Radiometer Data

ORIGINAL CONTAINS COLOR ILLUSTRATIONS

William J. Ripple

Abstract

NOAA-9 satellite data from the Advanced Very High Resolution Radiometer (AVHRR) were used in conjunction with Landsat Multispectral Scanner (MSS) data to determine the proportion of closed canopy conifer forest cover in the Cascade Range of Oregon. A closed canopy conifer map, as determined from the MSS, was registered with AVHRR pixels. Regression was used to relate closed canopy conifer forest cover to AVHRR spectral data. A two-variable (band) regression model accounted for more variance in conifer cover than the Normalized Difference Vegetation Index (NDVI). The spectral signatures of various conifer successional stages were also examined. A map of Oregon was produced showing the proportion of closed canopy conifer cover for each AVHRR pixel. The AVHRR was responsive to both the percentage of closed canopy conifer cover and the successional stage in these temperate coniferous forests in this experiment.

Introduction

On 7 October 1989 the United States Congress approved legislation requiring that federal agencies preserve contiguous stands of old-growth forests in the Pacific Northwest rather than harvest in a patchwork pattern resulting in forest fragmentation (Lehmkuhl and Ruggiero, 1991; Lord and Norton, 1990). This legislation was intended to protect the habitat of the northern spotted owl which needs large stands of old-growth forest to survive extinction (Ripple *et al.*, 1991b). With this legislation in place, developing methods for measuring the extent of forest cover and fragmentation at the landscape and regional level is important. Forest fragmentation has been documented at the landscape scale using a geographic information system to describe patch size, shape, abundance, spacing, and forest matrix characteristics (Ripple *et al.*, 1991a). Spatial data on forest fragmentation at a regional scale is typically not available.

The main objective of the research reported here was to describe the relationship between the proportion of coniferous closed canopy forest cover and AVHRR pixel radiance values. A second objective was to obtain spectral signatures from various forest successional stages using AVHRR data. With a ground resolution of approximately 1.1 km, AVHRR data can not detect individual disturbances such as clearcuts that average between 10 and 20 ha on public lands. However, it was hypothesized that radiance as measured by the AVHRR would have an inverse linear relationship with the proportion of coniferous closed canopy forest cover.

Apparently, few studies have been designed to determine cover of temperate coniferous forests with AVHRR data. Nelson (1989) attempted to use Global Area Coverage (GAC) AVHRR data with a 4-km resolution to estimate forest area for the entire United States. His results showed that GAC and MSS were not highly correlated. Loveland *et al.* (1991) included forests (coniferous and deciduous) in the development of their land-cover database for the conterminous United States. Their remote sensing estimates of forest cover were compared to other inventory approaches by Turner *et al.* (in press).

AVHRR satellite data have been used for regional estimates of forest cover in the tropical forests of South America and the eastern and southern hardwood forests of the United States. Nelson and Holben (1986) and Woodwell *et al.* (1987) used AVHRR data in conjunction with calibrations from the finer resolution Landsat Multispectral Scanner (MSS) data to discriminate cleared areas from primary forest in Brazil. Iverson *et al.* (1989) and Zhu and Evans (1992) used regression equations based on estimates of forest cover from Landsat Thematic Mapper (TM) data to determine the relationship between AVHRR data and forest cover. Townshend and Tucker (1984) found that AVHRR data represented 70 percent of MSS data variation during a land-cover mapping study. The normalized difference vegetation index (NDVI) [(near infrared − visible)/(near infrared + visible)] explained 70 percent and 79 percent of the variation in leaf area index in coniferous forest stands in western United States (Spanner *et al.*, 1990a). The authors concluded that reflectance was related to the proportion of surface cover types within AVHRR pixels as well as changes in leaf area index.

Study Area

The study area was in the western Cascade mountains of Oregon and encompassed both public USDA Forest Service land

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on the Willamette National Forest and private land (Figure 1). The study area size was approximately 258,930 hectares, with elevations ranging from approximately 240 m to 1700 m above mean sea level. This area falls within portions of the western hemlock (*Tsuga heterophylla*) and Pacific silver fir (*Abies amabilis*) vegetation zones with the major tree species consisting of Douglas-fir (*Pseudotsuga menziesii*), western hemlock, Pacific silver fir, noble fir (*Abies procera*), and western red cedar (*Thuja plicata*) (Franklin and Dyrness, 1973). The topography is highly dissected by steep slopes with slope gradients ranging between approximately 0 and 60 degrees. The maritime climate has wet-mild winters and warm-dry summers. The landscape consists of large areas of old-growth Douglas-fir/western hemlock forests over 400 years old. Major disturbances include both fire and logging, resulting in a patchwork mosaic of herbaceous areas, deciduous shrubs and trees, and both natural and man-made closed canopy conifer forests. Areas of big leaf maple (*Acer macrophyllum*) and red alder (*Alnus rugosa*) are also found in the study area.

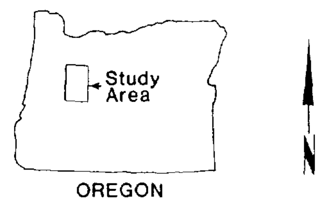

Figure 1. Map showing the study area which was located on the west slopes of the Cascade Mountains in western Oregon.

Methods

Part of a Landsat MSS scene acquired on 31 August 1988 was rectified using 7.5-minute orthophoto quadrangles. A nearest neighbor interpolation method was used to resample to a 50- by 50-m pixel size from the original 57- by 79-m pixel size. Using an unsupervised classified scheme, the delineated data were grouped into "closed canopy conifer forests" and "other areas" as part of a Landsat forest change project (Spies *et al.*, in press). The "closed canopy conifer forest cover" was old-growth, mature, and other conifer stands greater than approximately 30 to 40 years of age, and within-stand conifer canopy cover exceeded 60 percent (Brown, 1985). The term "closed canopy conifer forest cover" will be referred to as simply "closed conifer cover" in this paper. The "other areas" included clearcuts, young pre-canopy closure conifer plantations, shrubs, bare soil and rock, natural meadows, and water. Approximately 100 spectral classes were generated and were aggregated into the binary classification described above.

An accuracy assessment was conducted using a systematic sampling of 135 points across the study area. High-altitude, color-infrared photographs (scale 1:60,000) from July 1988 were used to check each of the 135 points. The cover at each point was identified on the aerial photography and compared to the category at the corresponding location on the classified image.

The AVHRR data were acquired in a local area coverage format from NOAA-9 orbit 18558 on the afternoon of 19 July 1988. A sub-scene of these data was rectified to UTM coordinates and registered to the MSS data set. AVHRR pixels were resampled to 1,000 by 1,000 m using a nearest-neighbor algorithm. The mean-square-error was 0.6 AVHRR pixels for the registration of the AVHRR data to the MSS data. Eighty-nine AVHRR pixels were selected systematically from the AVHRR set, and a window of 20- by 20-MSS pixels was selected to correspond to each AVHRR pixel. The visible (0.58 to 0.68 μm) and near infrared (NIR, 0.725 to 1.100 μm) band values were recorded for the 89 AVHRR pixels, and the percent closed conifer cover of each corresponding 20 by 20 MSS pixel window was computed. The percent closed conifer cover was defined as the proportion of an area or pixel containing closed conifer stands. Correlations were computed for the closed conifer cover, visible, and NIR variables along with the NIR/visible band ratio and the NDVI. In addition, the AVHRR band values were converted to albedo using pre-launch calibration equations for the computation of a corrected band ratio and corrected NDVI (NOAA, 1991). Computing albedo from the AVHRR band values helps to allow for a direct comparison of NDVI values with other studies using AVHRR data (Spanner *et al.*, 1990b). A stepwise multiple regression was used to develop a model for predicting closed conifer cover from the AVHRR data. The model was validated by comparing predicted proportions of closed conifer cover for the entire study area to the proportion of closed conifer cover for the study area as determined from the MSS data set.

To determine spectral variation associated with successional changes, spectral signatures were extracted from this AVHRR data set for a set of homogeneous landscapes consisting of four different successional stages. These included a very large, disturbed clearcut area with an herbaceous/shrub canopy cover of approximately 90 percent (21 AVHRR pixels, 3 by 7); managed, closed-canopy Douglas fir forests approximately 40 years old with approximately 15 percent of the area in bigleaf maple and red alder (16 AVHRR pixels; 4 by 4); natural mature Douglas fir forests approximately 90 to 130 years old (21 AVHRR pixels, 3 by 7); and old-growth Douglas fir and western hemlock forests 400 to 600 years old (20 AVHRR pixels, 4 by 5). The mean AVHRR band values and standard deviations for each of these forest successional stages were plotted for the visible, NIR, and NDVI response.

Results

The MSS classification showed 57.8 percent of the study area in closed conifer cover with a total accuracy of 91 percent. A total of 74 out of 83 samples (89 percent) fell correctly into the closed conifer class, and a total of 49 out of 52 samples (94 percent) fell correctly into the "other land" category. The relationship between AVHRR pixel band values and closed conifer cover was inverse for both bands as expected (Figure 2). The visible ($r = -0.64$, $r^2 = 0.41$, $P<0.0001$) and NIR ($r = -0.63$, $r^2 = 0.40$, $P<0.0001$) bands showed much higher correlations with closed conifer cover than the albedo-corrected band ratio ($r = -0.14$, $r^2 = 0.02$, $P=0.190$) and the albedo-corrected NDVI ($r = -0.15$, $r^2 = 0.02$, $P=0.168$). The uncorrected band ratio ($r = -0.46$, $r^2 = 0.21$, $P<0.0001$) and the uncorrected NVDI ($r = -0.46$, $r^2 = 0.21$, $P<0.0001$) had significantly higher correlations with closed conifer cover than the albedo-corrected transformations (Figure 3).

Scatter diagrams between closed conifer cover and red or infrared AVHRR bands showed inverse linear relationships with random dispersal of points. The step-wise multiple

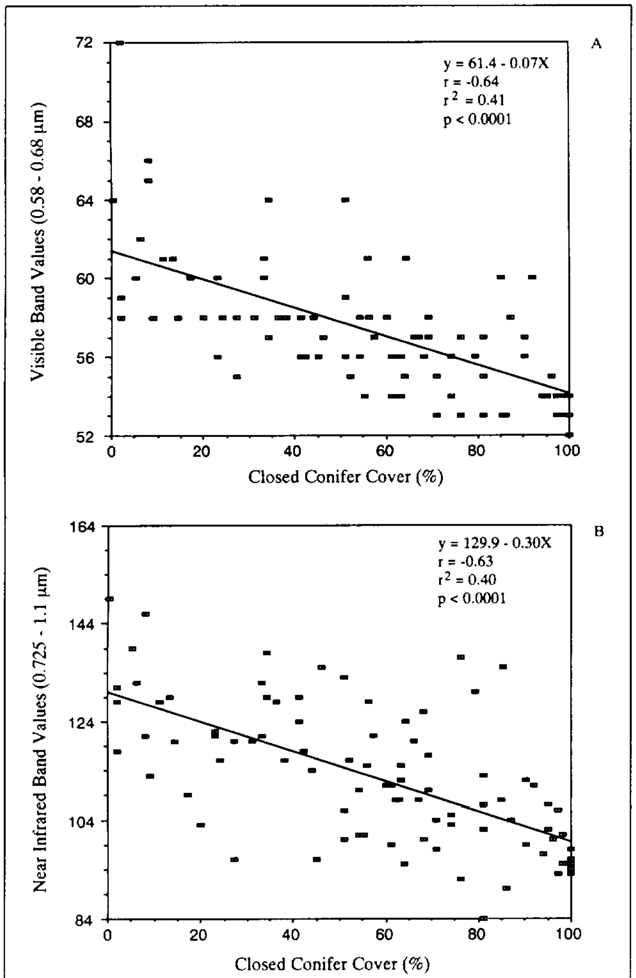

Figure 2. Scatter diagram and simple linear regression of (a) AVHRR visible band values and percent closed conifer cover along with simple linear regression of (b) AVHRR near infrared band values and percent closed conifer cover.

regression resulted in the model: percent closed conifer cover = 335.53 − 3.42 (visible) − 0.73 (NIR) with an adjusted coefficient of determination of 0.46 ($P<0.001$). When the above model was applied to the entire study area of 2405 AVHRR pixels, the mean percentage of closed conifer cover was 54.7 percent which compared favorably with the original MSS estimate of 57.8 percent. Spatial patterns in the AVHRR data from the model output were similar to the MSS patterns (Figure 4).

Figure 5 shows AVHRR band values related to forest successional stage. Band values decreased with forest succession for both the visible and NIR bands. It should be noted that these decreases were after the establishment of herbs and shrubs on the landscape. The visible band showed the greatest difference between mean band values for herb/shrub and young closed canopy conifers. The NIR showed the greatest difference between mean band values between young conifer and mature forests as well as between mature and old-growth band values. The NDVI values were lowest for the herb/shrub class, highest for the young conifer class, and intermediate for mature and old-growth forests.

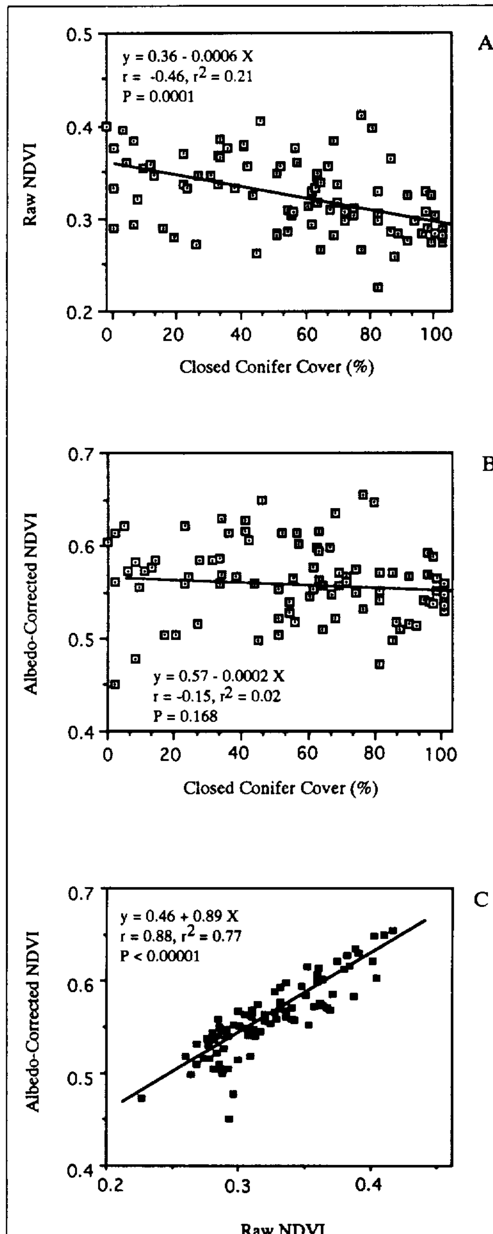

Figure 3. Scatter diagrams and simple linear regression of (a) raw NDVI and percent closed conifer cover, (b) albedo-corrected NDVI and percent closed conifer cover, and (c) albedo-corrected NDVI and raw NDVI.

Discussion

The results from this experiment indicate that it may be possible to use AVHRR data to obtain landscape and regional estimates of closed conifer cover in the temperate coniferous forests. Bands 1 and 2 were related to both the proportion of

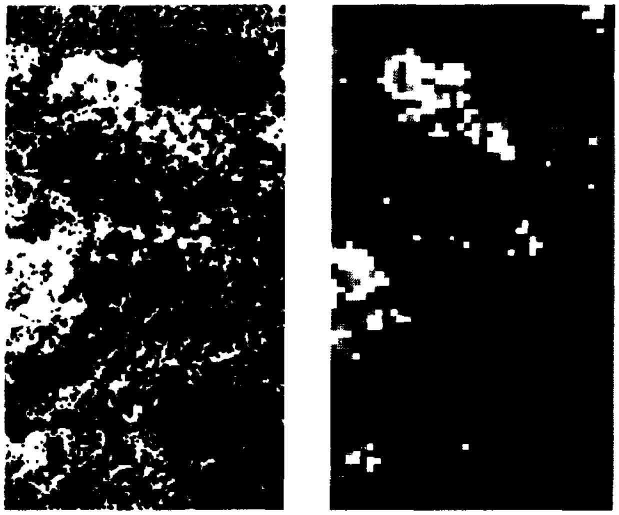

Figure 4. The image on the left shows closed conifer cover in black and other lands in white. It was produced from classified Landsat MSS data (50-m pixels). The right image is a gray scale of AVHRR data (1,000-m pixels) showing the percentage of closed conifer cover within AVHRR pixels (low percentages are light and high percentages are dark).

closed conifer cover within AVHRR pixels and the seral stage of the vegetation. Both visible and NIR band values decreased linearly with the proportion of closed conifer cover within a pixel. This result is in contrast to the typically found direct relationship between NIR and vegetation amount for many vegetation types. Researchers have found that this direct relationship between NIR with measures of conifer amount does not always exist, especially for canopies that are not closed. When the background or understory is brighter than the conifer canopy, the relationship can be inverse (Ripple *et al.*, 1991c), and when the background is darker or the canopy is closed, it can be direct (Spanner *et al.*, 1990a). When the background has about the same brightness as the conifers or is highly variable, the relationship to NIR may be flat or weak. It has also been hypothesized that conifer canopy shadowing contributes to an inverse NIR relationship as the canopy structure becomes more complex and shaded in older stands (Ripple *et al.*, 1991c). The reason for the inverse relationships between the band values and closed conifer cover in this study was attributed to the deciduous shrubs and trees along with herbaceous vegetation being more highly reflective than the conifer canopy in both the visible and NIR bands (Fiorella and Ripple, 1993). Areas within AVHRR pixels not covered by a conifer canopy consisted mostly of herbaceous or deciduous woody vegetation with very little bare soil. Therefore, band values decreased as the proportion of the darker conifer cover increased.

The NDVI relationship with closed conifer cover was also inverse, but the correlation was not as high as with the visible and NIR bands. The two-variable (band) regression model accounted for more variance than either the NDVI or the individual bands. These results are similar to those found by Iverson *et al.* (1989). The variance not accounted for in this regression model was attributed to slight misregistration between the AVHRR and MSS data sets and variability in the broad successional stages. The mean-square error of 0.6 AVHRR pixels, equivalent to approximately 600 metres on the ground, may have been the cause for some of the scatter around the regression lines. Overall, the model worked well, considering that each AVHRR pixel represented 400 times more land area than each Landsat MSS pixel.

It should be noted that the albedo-corrected NDVI and the uncorrected NDVI were not perfectly correlated ($r = 0.88$, $r^2 = 0.77$, Figure 3c). The albedo-corrected NDVI had a lower correlation to the percentage of closed conifer cover than the uncorrected NDVI. The differences between the uncorrected and corrected NDVI values were greater than the rounding errors inherent in the transform from the original AVHRR band values to the albedo-corrected NDVI. Most recent research on the calibration of AVHRR has been concerned with post-launch performance of the sensor, and specifically the degra-

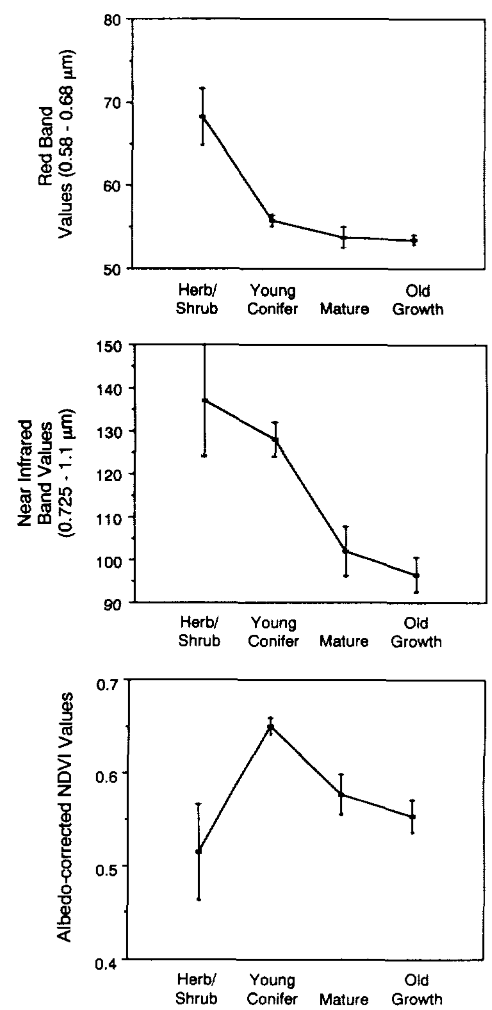

Figure 5. Band value means and standard deviations using the visible, near infrared, and the normalized differences vegetation index (NDVI) from AVHRR data for four successional stages found on the west slope of the Cascades of Oregon.

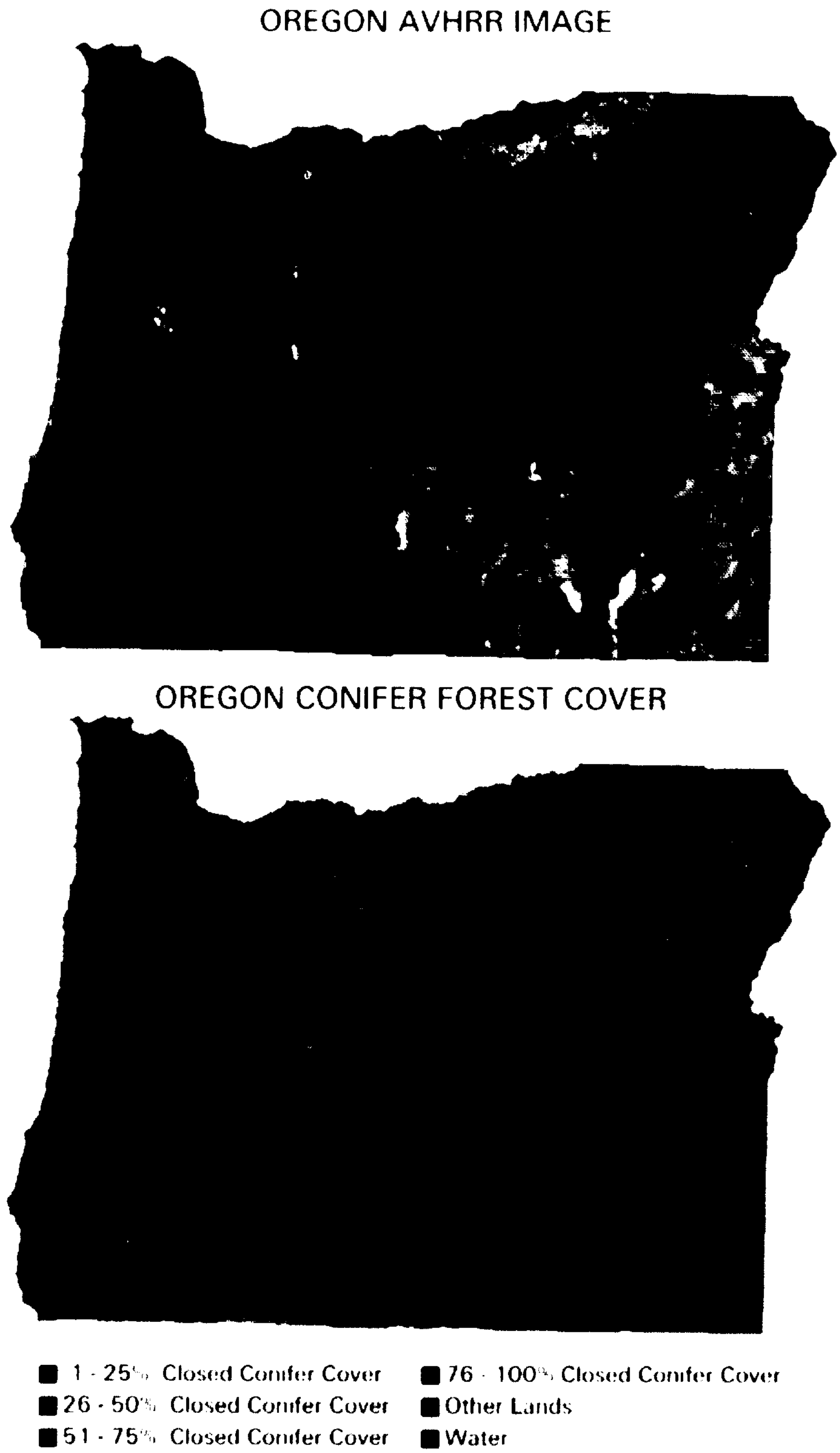

Plate 1. Raw AVHRR image of Oregon (top) with the visible and NIR bands and closed conifer cover patterns in Oregon (bottom) with four broad closed conifer cover classes (25 percent cover intervals). Film recording courtesy of ERDAS production services, Atlanta, Georgia.

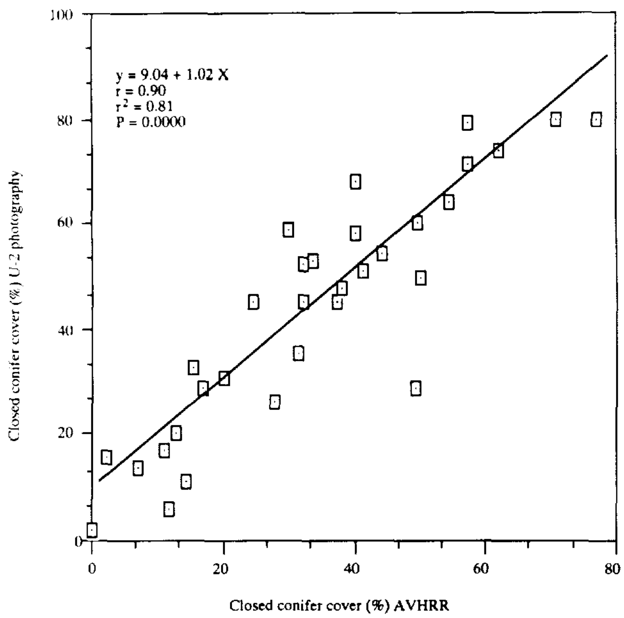

Figure 6. Comparison of estimates of closed conifer cover from AVHRR satellite data and from U-2 aerial photography for 32 7.5-minute quadrangles in Oregon.

dation in the instrument gain values over time because no on-board calibration is available (Che and Price, 1992; Kaufman and Holben, 1993). Because of this degeneration over time, it is especially important to use post-launch calibration procedures in studies comparing multiple AVHRR images. Off-nadir viewing, atmospheric variability, and instrument precision also creates errors in NDVI observations (Goward *et al.*, 1991). Little information is available on the effects transforming raw AVHRR values into calibrated values and then in turn converting those calibrated values into NDVI space. The discrepancy between raw NDVI and Albedo-corrected NDVI values apparently result from two different locations in cartesian space (NIR, red coordinates) being non-linearly transformed into polar (NDVI) space. Similar discrepancies could also exist in other situations where satellite band values are corrected before transforming them into a ratio like the NDVI.

The relationship between visible band values and forest successional stage was inverse and somewhat asymptotic after canopy closure. The largest mean differences between the seral stages of young and mature, and mature and old-growth were found using the NIR band. NIR band values between the seral stages of herb/shrub and closed canopy young conifer decreased because of decreased deciduous vegetation exposed to the sensor. This trend continued between the closed canopy young conifer and the mature conifer seral stage as the increase in canopy shadowing and decrease of the bright deciduous component apparently dominated off-setting increases in NIR due to increases in leaf area index (Ripple *et al.*, 1991c; Spanner *et al.*, 1990a). NIR values for old-growth were slightly lower than mature forest. This was probably due to the increase in shadowing and canopy gaps dominating the signal resulting in a lower radiance for old-growth (Fiorella and Ripple, 1993). This was not likely due to the higher leaf area index in the old-growth, because light becomes asymptotic above a LAI of 6 (Spanner *et al.*, 1990b) and old-growth LAIs are normally higher than 6 in this area (Franklin *et al.*, 1981).

NVDI increased from the herb/shrub stage to the young conifer stage and then decreased through the mature to the old-growth stage indicates a potential problem with using NDVI for successional stage mapping. Within these temperate coniferous forests, it appears that both pre-and post-canopy closure NDVI values could be confused. Spanner *et al.* (1989) had similar results when using the similar TM band 4/3 ratio in these forests. Box *et al.* (1989) also concluded that there did not seem to be any reliable relationship, across different vegetation structures, between biomass and NDVI when using AVHRR data. Multiple regression of individual bands may be the best approach for successional stage mapping with AVHRR.

Application of Results

The results presented above have applications in both measuring the total extent of coniferous forests and the level of forest fragmentation for larger regions. For example, the regression model was also applied to the entire state of Oregon to determine the spatial distribution of conifer forests and levels of forest fragmentation. The resulting state map showed closed conifer cover ranging from 0 to 100 percent in 1 percent increments just as in Figure 4 above for the small area. Because it was not feasible to show a map of all 100 classes for the entire state, a generalized map was produced showing the estimates of the proportion (in 25 percent steps) of closed conifer cover for each AVHRR pixel in the state (Plate 1). Within western Oregon, the map showed the highest levels of forest fragmentation in the Oregon Coast Range and the lowest fragmentation in the southern Cascade Mountains, especially in and around the Umpqua National Forest. The map can also be used to analyze landscape linkages for biodiversity planning and ecosystem management. For example, it showed the various levels of fragmentation on forest corridors linking the Cascade range with both the Coast Range and the Klamath Mountains.

The AVHRR-based Oregon map showed 10,891,000 ha of conifer forests in Oregon. This is 4 percent higher than estimated by the U.S. Forest Service under the Resources Planning Act (RPA). The RPA estimated 10,463,827 ha of conifer and 890,750 ha of hardwoods for a total of 11,354,577 ha of forest land in Oregon. The land-cover database for the conterminous United States shows total forest land in Oregon at 11,990,000 ha (Loveland *et al.*, 1991; Turner *et al.*, in press).

This state-wide data set of percentage of closed conifer cover was verified through a comparison with U-2 color-infrared aerial photography flown by NASA on 19 July 1988. The flight covered parts of the central Oregon Coast range, the north Cascade range, the central Cascade range, and the Blue Mountain range in northeastern Oregon. Eight 7.5-minute quadrangles were randomly selected in each of these four geographic areas to compare with the AVHRR set. The proportion of closed conifer cover in each of these 32 quadrangles was estimated using a dot grid on the U-2 photography. These same 32 quadrangles were windowed out of the AVHRR closed conifer cover data set.

The proportion of closed conifer cover as estimated from AVHRR was highly correlated ($r^2 = 0.81$) with estimates from the U-2 photography (Figure 6). The regression between these two sets showed a linear relationship with the U-2 photos providing slightly higher estimates of closed conifer cover than the AVHRR set (intercept = 9.04). This difference was probably due to the ability to detect very small conifer patches on the U-2 photography.

The relationship between AVHRR pixel values with the proportion of closed conifer cover indicate that these sensor data may be well suited as an integrator for large area forest fragmentation studies in this Pacific Northwest region. For example, these AVHRR results were used in designing proposed federal legislation for locating biological forest reserves to protect the northern spotted owl and other important wildlife species (Forest Ecosystem Management Team, 1993).

Conclusions

This research has shown that band values in the AVHRR channels 1 and 2 are related to the continuum of forest landscape conditions. The regression model was successful because areas that were highly fragmented by clearcutting had high band values in both the visible and near infrared bands. Conversely, areas dominated by late-successional forests and low fragmentation consistently had the lowest band values.

Additional research should be conducted to confirm the spectral/successional stage relationships found here. This work could include research using pixel mixture models (Smith *et al.*, 1990) to develop a better understanding of how the proportions of successional stages and forest cover determine the spectral response in large AVHRR pixels. Caution should be used when making calibration corrections before calculating the NDVI and in attempting to use the NDVI when considering data sets that include both natural and managed stands with a range of structural characteristics and successional stages (i.e., herbs, deciduous shrubs, conifers, etc.). It appears that AVHRR has potential for estimating forest fragmentation in temperate coniferous forests and for mapping the spatial distribution of forest resources at continental and global scales.

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DYNAMICS AND PATTERN OF A MANAGED CONIFEROUS FOREST LANDSCAPE IN OREGON[1]

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Abstract. We examined the process of fragmentation in a managed forest landscape by comparing rates and patterns of disturbance (primarily clear-cutting) and regrowth between 1972 and 1988 using Landsat imagery. A 2589-km² managed forest landscape in western Oregon was classified into two forest types, closed-canopy conifer forest (CF) (typically, >60% conifer cover) and other forest and nonforest types (OT) (typically, <40 yr old or deciduous forest).

The percentage of CF declined from 71 to 58% between 1972 and 1988. Declines were greatest on private land, least in wilderness, and intermediate in public nonwilderness. High elevations (>914 m) maintained a greater percentage of CF than lower elevations (<914 m). The percentage of the area at the edge of the two cover types increased on all ownerships and in both elevational zones, whereas the amount of interior habitat (defined as CF at least 100 m from OT) decreased on all ownerships and elevational zones. By 1988 public lands contained ≈45% interior habitat while private lands had 12% interior habitat. Mean interior patch area declined from 160 to 62 ha. The annual rate of disturbance (primarily clear-cutting) for the entire area including the wilderness was 1.19%, which corresponds to a cutting rotation of 84 yr. The forest landscape was not in a steady state or regulated condition which is not projected to occur for at least 40 yr under current forest plans. Variability in cutting rates within ownerships was higher on private land than on nonreserve public land. However, despite the use of dispersed cutting patterns on public land, spatial patterns of cutting and remnant forest patches were nonuniform across the entire public ownership. Large remaining patches (<5000 ha) of contiguous interior forest were restricted to public lands designated for uses other than timber production such as wilderness areas and research natural areas.

Key words: clear-cutting; disturbance; edge habitat; forest management; habitat fragmentation; heterogeneity; interior habitat; landscape pattern; patchiness; remote sensing.

INTRODUCTION

Habitat fragmentation has become a major issue in forest management in recent years. Breaking large blocks of mature forest into a mosaic of young plantations, mature forests, and nonforest land has altered disturbance regimes, and contributed to loss of habitat, and reduced habitat quality for some species (Harris 1984, Lovejoy et al. 1984, Franklin and Forman 1987). Although habitat fragmentation has been recognized as one of the major classes of human impact on biodiversity and a high-priority research topic (Lubchenco et al. 1991, Soulé 1991), few studies have quantitatively characterized the process of fragmentation or pattern development in managed forest landscapes and how the patterns and rates differ among ownerships.

For about the last 50 yr, timber on National Forest land in the Pacific Northwest has been harvested using a staggered-setting system, in which 10–20 ha clearcuts are dispersed across large areas of older forest. Franklin and Forman (1987) used a simple checkerboard cutting model to simulate the landscape structure and ecological consequences of different clear-cutting patterns. Among several models they examined, the checkerboard (staggered-setting) model created landscapes with the most edge and resulted in a rapid loss of large uncut blocks of forest. Although the results of these simple simulations have led to a call for alternative cutting patterns, few studies have examined how cutting patterns actually develop and affect edge and interior habitat conditions on National Forests (Ripple et al. 1991*a*). Although idealized models of landscape change

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(Franklin and Forman 1987) and conservation area design (Harris 1984) provide a useful conceptual underpinning for management, implementing the conceptual models has to begin within the constraints of current landscape patterns and trends. Characterizing current patterns and understanding how they developed may help in designing future landscapes to meet a variety of ecological objectives within these constraints.

Many ecological processes transcend ownership boundaries. Managers and scientists are struggling to develop strategies and policies that recognize ecological conditions outside their jurisdictional boundaries (Johnson and Agee 1988, Schonwald-Cox 1988, Society of American Foresters 1991). We know of no studies that have examined how cutting patterns develop on adjacent public and private ownerships. Consequently, information is lacking on the potential effects of the juxtaposition of different forest management regimes on biological diversity over large areas.

Numerous researchers have used multidate maps (Curtis 1956) and remotely sensed data to illustrate changes in forest cover over time (e.g., Nelson et al. 1987, Iverson et al. 1989, Green and Sussman 1990, Sader and Joyce 1990, Hall et al. 1991). With the exception of Hall et al. (1991) these studies have examined the pattern of conversion from forest land to nonforest land. Few studies have measured changes in spatial landscape characteristics such as patch size and amount of edge environment (Skole and Tucker 1993).

In the Pacific Northwest, management for timber production has converted large areas of old-growth and other natural coniferous forest habitat to young conifer plantations in various stages of development and patches dominated by early successional deciduous shrubs and trees. This loss of older coniferous forest has threatened the populations of the Northern Spotted Owl and other old-growth associated species (Thomas et al. 1990, Johnson et al. 1991, Ripple et al. 1991*b*). While considerable attention has been paid to the ecological value of natural coniferous forests and the differences between natural forest structure and plantation forest structure (Franklin et al. 1981, Spies and Cline 1988, Hansen et al. 1991) at the stand level, few researchers have characterized how the coniferous forest landscape structure has changed as a consequence of forest management (Ripple et al. 1991*a*). Analyses of rates and patterns of coniferous forest landscape change are needed to better understand how management practices affect important habitat characteristics, such as amount of edge and interior habitat and patch size, and to provide a basis for making management and policy decisions that affect these ecosystems at a landscape level.

Our objectives were to: (1) Evaluate satellite imagery as a tool for monitoring and characterizing landscape change and structure in mountainous landscapes dominated by coniferous forests; (2) characterize rates of change in closed-canopy conifer forests for the 16-yr period between 1972 and 1988; (3) characterize changes in landscape pattern resulting from clear-cutting; and (4) contrast the pattern and dynamics of public and private forest landscapes.

Study Area

We chose a study area where we were familiar with forest conditions from previous ground-based studies (Spies and Franklin 1991) and where we could examine a large public land planning unit (in this case the Willamette National Forest) and the private lands bordering it (Fig. 1). The study area is on the western slope of the Cascade Range in Oregon, extending from about 44.0° to 44.6° North Latitude and 122.0° to 122.5° West Longitude. The total area is 258 930 ha. Elevations range from 244 to 1706 m above mean sea level (MSL) with slope gradients between 0 and 45 degrees. The area is primarily in the Western Hemlock (*Tsuga heterophylla* [Raf.] Sarg.) and Pacific Silver Fir (*Abies amabilis* Dougl. ex Forbes) Zones (Franklin and Dyrness 1973). Major forest tree species include Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), western hemlock, Pacific silver fir, noble fir (*Abies procera* Rehd.), and western redcedar (*Thuja plicata* Donn).

The dominant timber-management objective in the area has been even-aged coniferous plantations, primarily Douglas-fir; deciduous species that establish after clear-cutting are typically reduced or eliminated through thinning and herbicide treatment on intensively managed lands. The public ownership of ≈70% of the study area is primarily managed by the USDA Forest Service, Willamette National Forest (Fig. 1). The Bureau of Land Management and State of Oregon make up <10% of the total area of public lands. Private lands consist primarily of industrial land ownerships, with some areas of nonindustrial forest land.

Relatively little clear-cutting occurred on public lands until the early 1950s. Consequently, most stands >40 yr old regenerated naturally after wildfires, so logging primarily occurs in naturally developed forests, the majority of which are dominated by even-aged or multiple-aged conifers 100–500 yr old. On private lands logging began earlier in the century and we estimate that about half of the closed-canopy conifer forests on these lands were natural conifer forests 50–500 yr old in 1972. Early successional forests after fire or logging are initially dominated by evergreen and deciduous shrubs, and occasionally deciduous trees such as red alder (*Alnus rubra* Bong.) and big-leaf maple (*Acer macrophyllum* Pursh). By 25–50 yr after major disturbances, natural stands are typically characterized by closed canopies dominated by conifers. As conifer canopies close, major changes in the plant and animal communities occur and most conifer forest species begin to find suitable habitat at this stage, with the exception of old-growth habitat specialists and those species dependent on large standing dead trees (Brown

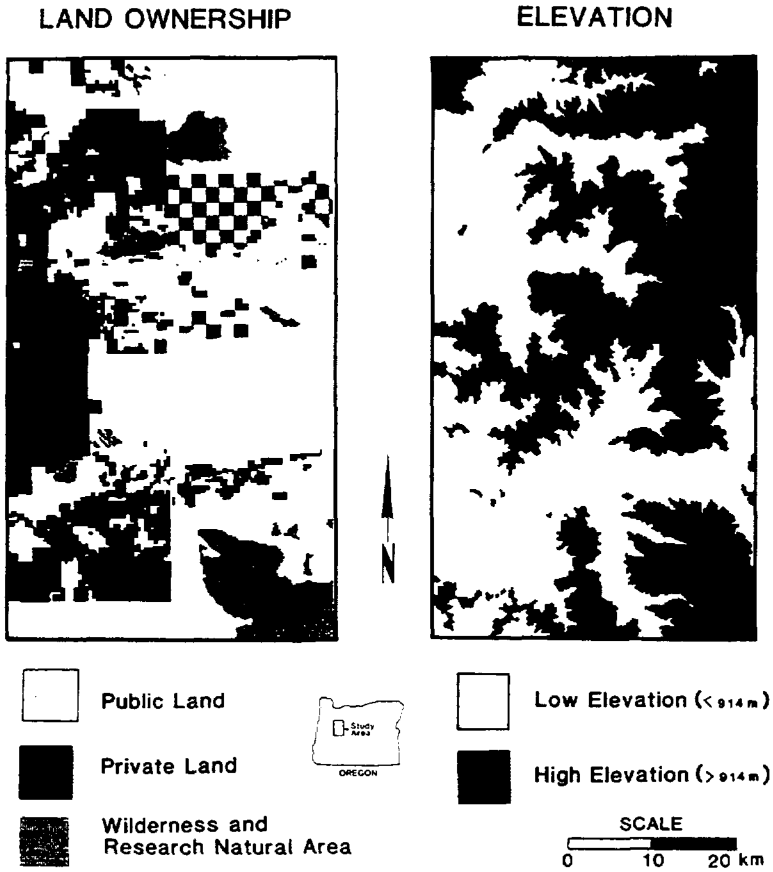

FIG. 1. The study area in western Oregon showing distributions of major ownerships and land allocations and elevation zones.

1985, Schoonmaker and McKee 1988, Ruggeiro et al. 1991).

METHODS

Raw digital satellite data sets for 1972, 1976, 1981, 1984, and 1988 were selected from Landsat Multi-Spectral Scanner (MSS) scenes and rectified using orthophoto quadrangles to a universal transverse mercator projection. Additional information on the characteristics of Landsat MSS data can be found in Lillesand and Kiefer (1987). To minimize vegetation differences from seasonal changes, all of the MSS image data used in the study were from the three summer months. The Earth Resources Data Analysis System (ERDAS) was used to implement a nearest neighbor interpolation method with resampling at 50 × 50 m pixel size from the original 57 × 79 m pixel size.

After rectification, each image was classified into three broad cover types: closed canopy conifer forest (CF), other forest and nonforest types (OT), and water. Closed-canopy conifer forests, i.e., stands in which conifer crowns occupied at least 60% of the area, ranged in age from 40 to 750 yr, and were typically at least 15–30 m tall. The OT class included recent clearcuts, brush fields, young pre-canopy-closure conifer plantations, and closed-canopy stands dominated by deciduous trees. Pixels classed as water consisted primarily of large reservoirs.

Only two forest cover classes were used to simplify the change analysis, to emphasize cutting patterns, and to minimize the ground-truthing needed to develop the classification. Our purpose was to quantify rates of disturbance and changes in spatial pattern relative to the initial matrix of conifer forest. Consequently, we did not distinguish among different classes of closed-canopy conifer forest including old-growth, or different classes of early successional and deciduous cover. Although this simplification may limit the relevance of our results for old-growth issues, the ecosystem transitions represented by the two classes encompass the two most dramatic changes in biological diversity and ecosystem function that occur in these landscapes: (1) the change from a natural conifer forest to an early successional clearcut, and (2) the change from an open or deciduous ecosystem to a conifer-dominated ecosystem.

We used an unsupervised classification scheme in which the image data were aggregated into 100–150 natural spectral classes, then grouped into the three classes described above (Lillesand and Keifer 1987). The accuracy of the 1972 and 1988 image classifications was assessed using aerial photography. A system-

TABLE 1. Area of closed-canopy coniferous forest (CF) and other forest and nonforest types (OT) in 1972 and 1988 and net change in CF by ownership and allocation.*

Land ownership and allocation	Amount in 1972				Amount in 1988				Net change in CF	
	CF		OT		CF		OT			
	ha	%	ha	%	ha	%	ha	%	ha	%
Wilderness	18 571	93.4	1307	6.6	18 374	92.4	1504	7.6	−197	−1.1
Public nonwilderness	125 785	78.6	33 881	21.2	109 500	68.4	50 094	31.3	−16 285	−12.9
Private	39 493	50.0	38 677	49.0	21 777	27.6	56 369	71.0	−17 716	−44.9
Entire area	183 849	71.0	73 865	28.5	149 651	57.8	107 967	41.7	−34 198	−18.6

* Percentages may not sum to 100 because area of water is not included.

atic sampling of 135 points was applied across the study area. High-altitude color-infrared photographs were used to check each point for correctness of classification. Limited reconnaissance on the ground and previously established vegetation plots (Spies and Franklin 1991) were also used to verify classification success.

The classification of cover types was overlaid with the spatial variables of land ownership and elevation using ERDAS. Wilderness, Research Natural Areas, and State Parks were put in a separate public ownership class because they will remain uncut. Elevation was divided into two classes; >914 m (3000 feet) above MSL and ≤914 m. The 914-m elevation approximates the division between the Western Hemlock and Pacific Silver Fir Zones (Franklin and Dyrness 1973), and it is a transition where many warm climate vertebrate species do not occur and many cool climate vertebrate species begin to occur (Harris 1984).

Maps of forest edge and interior were constructed from the classified Landsat maps by using a geographic information system (GIS). Edge length was defined as the total linear distance along the closed-canopy forest boundary, and the percentage of edge was defined as the percentage of pixels with edge of the total pixels. Interior habitat was defined as the amount of closed-canopy forest remaining after designating a 100-m edge zone. The width of the edge zone was based on a study of high-contrast edges (50–60 m tall conifer forests that border recent clearcuts) in which edge effects as measured by microclimate and vegetation dynamics extended from 20 to at least 240 m into the forest, depending on the variable examined (Chen 1991). A 100-m edge zone was applied because many edge effects are considerably reduced by this distance, and edges in the study area are lower contrast than are edges between a clearcut and old-growth edge and presumably have narrower zones of edge effect than the maximum.

Three analyses of subsamples (5000 × 5000 m or 2500 ha) of private and public nonreserve forest land were conducted to provide a more controlled comparison of the differences in development of landscape pattern between the two types of forest ownerships. In the first analysis, the changes in forest patterns from early to later stages of logging entry were examined in a single subsample with a high percentage of CF in 1972 and relatively high rates of cutting. This subsample was subjectively chosen from each ownership and compared over the five dates. In the second analysis, relations of area cutover to edge and interior habitat were compared. Six subsamples were subjectively chosen to obtain a wide range of percentage CF for each ownership from either the 1972 or 1988 classification, and then the percentage of edge and percentage of interior habitat were plotted against the percentage CF. In the third subsampling analysis, within-ownership variability (coefficient of variation) in cutting rates was estimated and compared from a random selection of six subsamples from each ownership for which changes in CF between 1972 and 1988 had been calculated.

RESULTS

Accuracy of classification

Classification accuracy was estimated to be ≈91% for both the 1972 and 1988 scenes. In 1972, 91% of the closed-canopy conifer forest and 92% of the other cover types were correctly classified. In 1988, 89% of the closed-canopy conifer forest and 94% for the other cover types were correctly classified. The overall accuracy of change statistics between 1972 and 1988 was 83%, using a simple joint probability assuming a uniform distribution of error throughout the maps.

Forest cover

The percentage of closed-canopy conifer forest (CF) in the entire landscape declined from 71.0% in 1972 to 57.8% in 1988, equivalent to a decline of ≈34 000 ha (Table 1). Amounts and changes in the proportions of the two forest types differed strongly by land ownership and allocation (Table 1 and Fig. 2). The highest percentage of CF and least amount of change in percentage of CF between 1972 and 1988 (93.4 to 92.4%) occurred on public wilderness and reserve lands, and the lowest percentage of CF and the greatest decline in percentage of CF (50.0 to 27.6%) occurred on private forest lands. Public nonwilderness was intermediate to the other ownerships and allocations in percentage of CF and decline in percentage CF between 1972 and 1988 (78.6 to 68.4%).

Change in closed-canopy conifer forest was roughly linear during the 16 yr for both ownerships (Fig. 3b). The greatest 4-yr declines in forest cover occurred be-

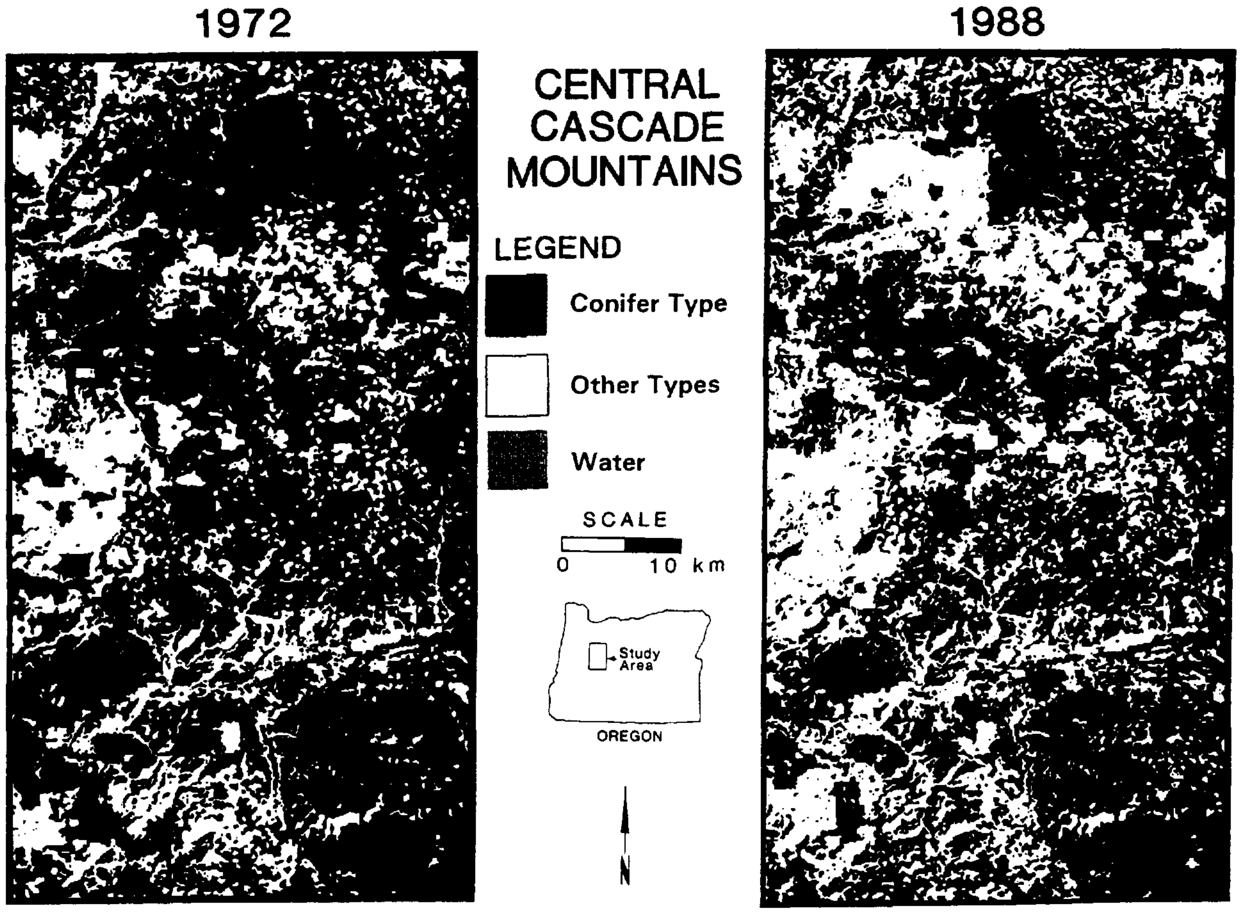

FIG. 2. Distribution of the conifer type and other types in the study area in 1972 and 1988.

tween 1984 and 1988 on public lands and between 1981 and 1984 on private lands. The pattern of change was similar at low and high elevations, although the low elevation sites had a lower percentage of closed canopy forest during the entire period (Fig. 3a).

Transitions between forest cover types

Pixels of CF in 1972 that became other forest and nonforest types (OT) by 1988 represent disturbances, primarily clear-cutting, that created early successional forests. Transitions in the other direction, pixels of OT in 1972 that became CF by 1988, represent succession and stand development. During this period the disturbance-caused transition rate between forest types was greater than the succession-caused transition rate between forest types. Over the entire area, the percentage of CF in 1972 that changed to OF by 1988 was 26.8, whereas the percentage of OT in 1972 that changed to CF by 1988 was 20.5 (Table 2). This trend was very strong on private lands, where disturbance transition rates were over three times greater than successional transition rates (Table 2). On public lands, the pattern was opposite to that of private lands: successional transition rates were slightly higher than disturbance transition rates on nonreserve lands and they were ≈10 times higher on reserve lands.

Disturbance rates and rotations

The annual rate of disturbance (based on area disturbed between 1972 and 1988 as a percentage of the total study area) and corresponding rotation age (inverse of the annual percentage of cutting rate) for the entire study area including wilderness was 1.19% and 84 yr. The annual rates of cutting on public nonwilderness land and private lands were 0.95 and 2.14%. These rates translate into forest rotations of 105 and 47 yr.

Changes in edge and interior habitat

Edge habitat.—The percentage of edge increased slowly over the 16-yr period (Fig. 3c, d). The density of edge increased from 1.9 to 2.5 km/km^2. The greatest increase in edge occurred between 1984 and 1988 (Fig. 3). Low elevations had higher proportions of edge than did high elevations (Fig. 3). The proportion of edge was lower on public lands than on private lands until 1988, when the proportion of edge on public lands increased rapidly to nearly the same proportion as on private lands (Fig. 3d).

Interior habitat.—The amounts and percentages of interior CF steadily declined during the study period (Fig. 3). High elevations had a consistently greater percentage of interior forest than did low elevations (Fig.

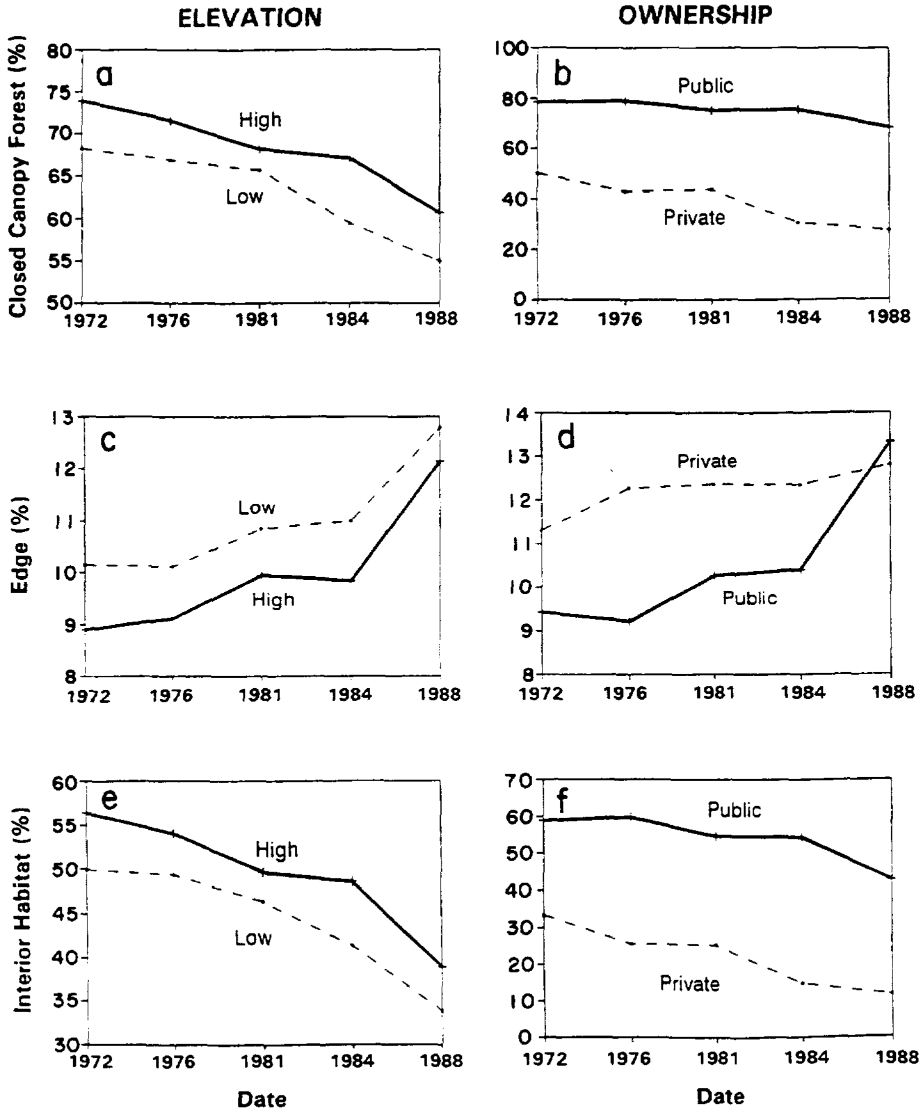

FIG. 3. Percentage of study area in conifer forest (a–b), edge (c–d), and interior habitat for five dates between 1972 and 1988, by elevation (a, c, and e) and ownership (b, d, and f)

3e). Private lands had much less interior forest than did public lands throughout the study period (Fig. 3f). By 1988, only ≈12% of the private lands were covered by interior forest, whereas public lands were ≈43% interior forest.

Interior habitat patch area.—Between 1972 and 1988, forest interior patches became smaller and more numerous (Fig. 4). Mean interior patch area declined from 160 to 62 ha. In addition, the percentage of interior forest area in large patches decreased (Fig. 4). In 1972, 50% of the total study area was in connected concentrations of interior habitat of at least 1000 ha; by 1988, the total had declined to 26%. During the 16 yr, a very large patch of 103 608 ha was broken into several smaller aggregates of <25 000 ha, concentrated primarily on public lands (Fig. 5).

Large patches were all on public lands and concentrated around: Three Sisters Wilderness (21 018 ha), Santiam River corridor (17 009 ha), Santiam Wilderness Area (14 774 ha), H. J. Andrews Experimental Forest (6139 ha), and Hagan Research Natural Area (4097 ha) (Fig. 5). Although these large patches all consisted of connected interior forest, they varied considerably in the amount and pattern of OF and noninterior CF contained within them. For example, the Three Sisters Wilderness in the southeast corner had relatively large aggregations (3000–5000 ha) of interior forest; the Santiam River corridor consisted of smaller, long, narrow interior forest aggregations and noninterior forest area (<2000 ha). In 1972, the study area was still dominated by interior forest habitat; by 1988, interior forest habitat was the dominant landscape condition only in subareas associated with special forest land allocations.

Landscape subsample analyses

Progression of cutting and regrowth.—Cutting patterns developed differently on public and private subareas (Fig. 6). On the private land subsample, where CF declined from 92 to 10%, large cut units in the upper and left portions of the block in 1972 became the foci of the very large cutover areas that appeared in 1981–1988. By 1988, the private subsample was characterized by isolated blocks of CF ranging from a few to ≈70 ha. On the public land subsample, where the percentage of CF declined from 84 to 65%, a dispersed pattern of small cuts in 1972 in the upper right expanded slowly in subsequent years by the dispersal of small units into the large uncut block in the lower portion of the area (Fig. 6). This large block remained relatively uncut until 1984. During this same period, cutting also occurred around the older units in the upper right portion of the image, resulting in aggregation of cutover areas. Until 1984, the conifer forest was entirely a connected matrix. During 1984 and 1988, a few small (<10-ha) isolated forest blocks began to appear in the area in which cutting initially began (Fig. 6).

Because of the occurrence of older plantations and large cutover areas, regrowth of CF was more easily illustrated on private lands than on public lands, where plantations were younger and regrowth was dispersed in relatively small units over large areas. Regrowth of CF for a subsample of private land followed a spatial process of nucleation, in which relatively large areas of CF in 1988 (53.9% CF) regrew from smaller foci of CF in 1972 (33.9% CF) (Fig. 7). Patterns of regrowth were only illustrated for 1972 and 1988 because the changes at 4-yr intervals were so small that they tended to be obscured by differences among the classifications of OT and CF for each date. The change from OT to CF from succession was much more gradual and less contrasting than the sudden change from CF to OT caused by clear-cutting.

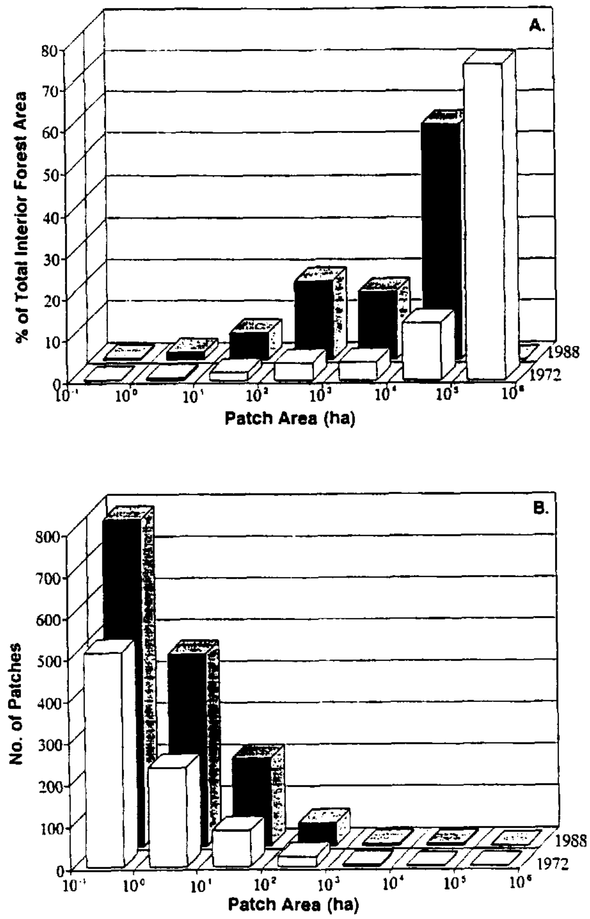

FIG. 4. Distribution of interior forest patch sizes in 1972 and 1988 in relation to percentage of total interior forest (A) area and total number of patches (B).

TABLE 2. Transitions in forest conditions between 1972 and 1988, based on areas in forest condition in 1972 (see Table 1) by owernship and allocation.

Land ownership and allocation	Type of change: Disturbance (CF to OT)	Type of change: Succession (OT to CF)
	Changes in hectares	
Wilderness	655	459
Public nonwilderness	24 210	8001
Private	24 396	6702
Entire area	49 261	15 162
	Changes in percentages	
Wilderness	3.5	35.1
Public nonwilderness	19.2	23.6
Private	61.8	17.3
Entire area	26.8	20.5
	Mean annual rates of change (%/yr)	
Wilderness	0.2	2.2
Public nonwilderness	1.2	1.5
Private	3.9	1.1
Entire area	1.7	1.3

Interior and edge.—Public land subsamples had less interior habitat and more edge habitat than private land blocks over a comparable range of cutover percentages (Fig. 8A). On public lands the percentage of edge rose to a peak at ≈40% cutover and then declined, roughly following a pattern predicted by Franklin and Forman (1987) for a uniform checkerboard model of cutting patterns. The private lands subsamples exhibited less edge per percentage of cutover and showed a relatively flat relation between edge and percentage cutover. The private land was probably cut in different types of patterns that created a relation of edge to cutover area that is a mixture of several spatial cutting models. Aggregated cutting after the landscape was 40% cutover resulted in a decline in percentage of edge for both ownerships, although the amount of edge after the 40% point varied considerably (Fig. 8B).

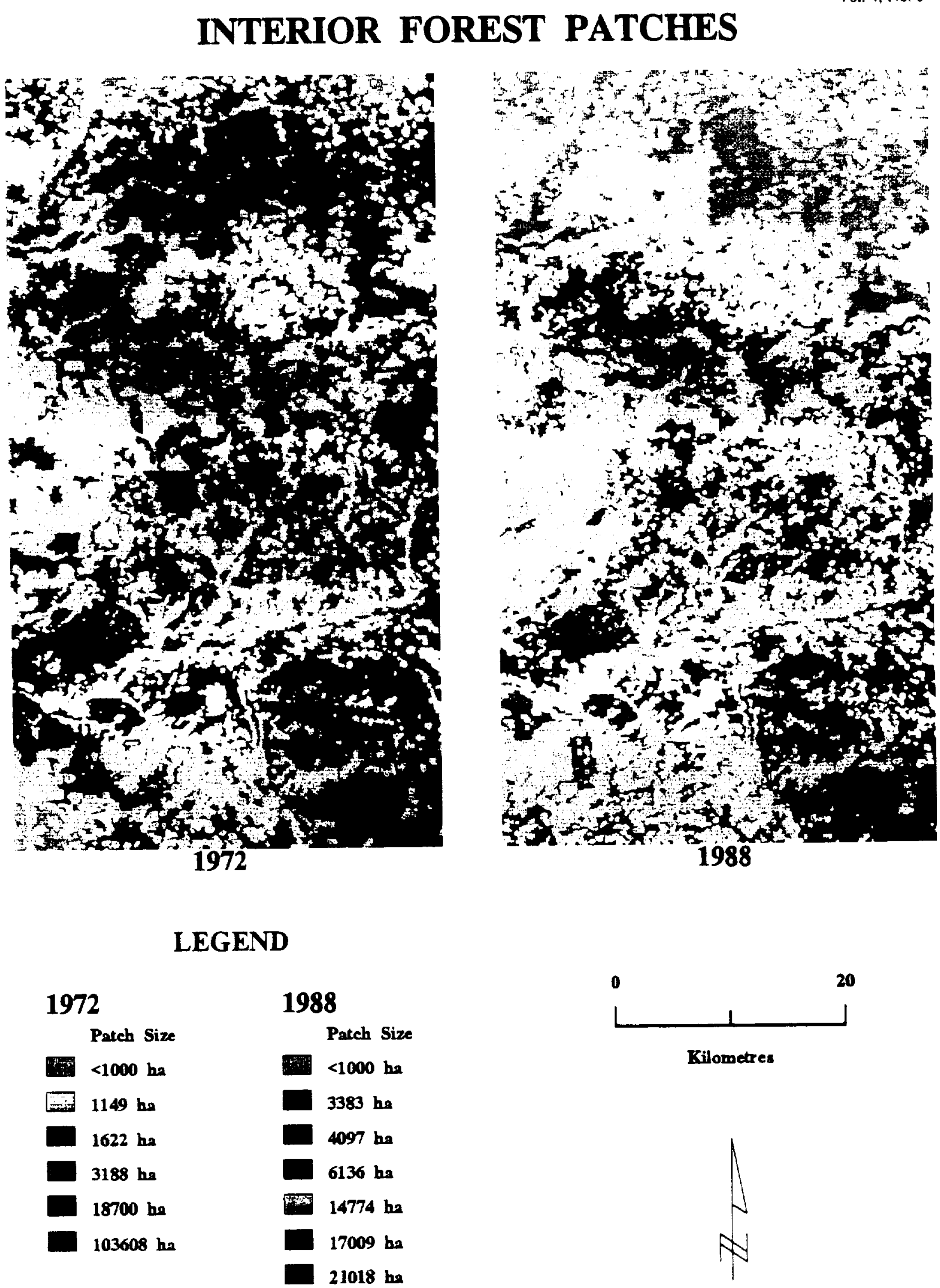

FIG. 5. Pattern of connected interior conifer forest patches in 1972 and 1988. Patches larger than 1000 ha are shown in different colors.

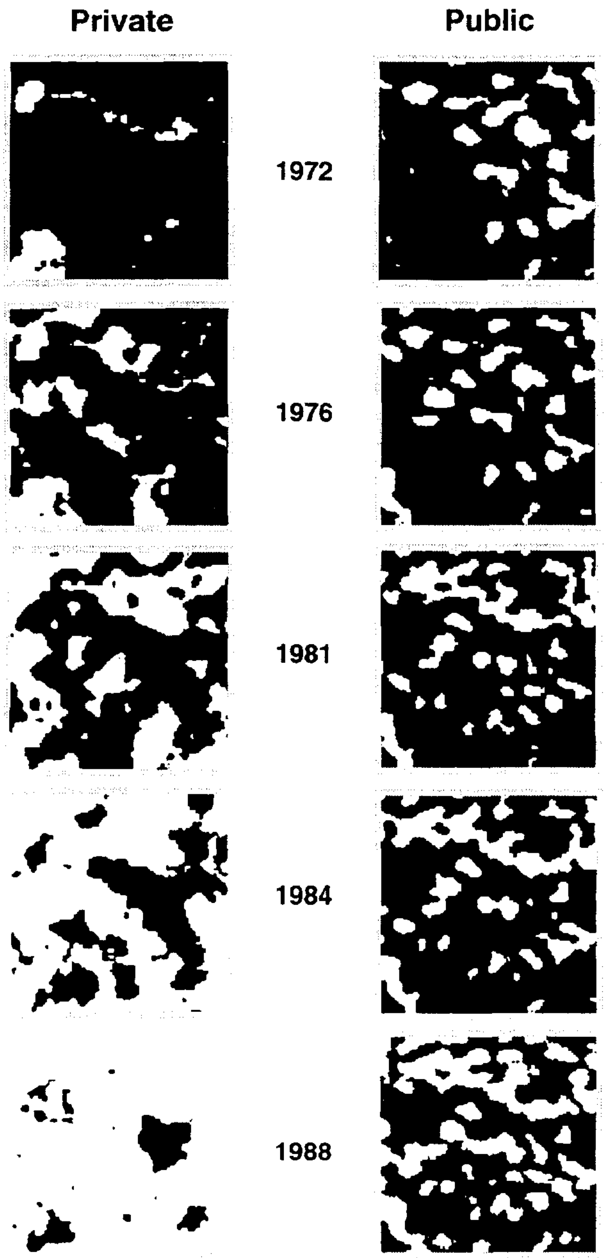

FIG. 6. Changes in conifer type (■) and other types (□) for a private and public sublandscape (2500 ha) with similar initial conditions and rates of change that are relatively high for the ownership type.

Variability in cutting rates.—Variability in rates of change of CF among subsamples from the private lands was greater than among subsamples from the public nonwilderness lands. The coefficient of variation of change in proportion of CF in the landscape, based on six 2500-ha subsamples, was 198% for private lands and 48% for public nonwilderness. The range of change in percentage of CF in a subsample was −4.5 to −14.5% for public nonwilderness and +14.7% to −82.5% for private lands. The differences reflect the application of dispersed cutting practices within the public ownership and the more aggregated cutting practices of private forest owners.

DISCUSSION

Comparison with ground-based estimates of change

The rates of cutting and regrowth of conifer forest that we estimated from satellite imagery were generally compatible with estimates based on ground-based inventories and forest management plans. Commercial forest land on the Willamette National Forest was cut at a rate of 0.9% from 1972 to 1988 (J. Mayo, *personal communication*), which is nearly identical to our estimate. The planned dominant harvest rate as of 1990 was 1.2% per year (80 yr rotation) (Anonymous 1990).

For all private forest lands in two regions in central western Oregon, an area of >4600 km^2, cutting rates averaged ≈1.0% between 1975 and 1985 (Greber et al. 1990), which is considerably lower than our estimate of 2.1%. We probably observed locally high rates of cutting on this portion of private land. However, many industrial and nonindustrial private lands are now using cutting rates of ≈1.8% per year or rotations of 55 yr (Greber et al. 1990:40), which is closer to the rate we found. A few industrial owners are using or planning rotations as short as 25 yr for Douglas-fir (K. N. Johnson, *personal communication*).

Transition rates and condition of the steady-state landscape

The high rate of disturbance (clear-cutting) on private lands is a consequence of management objectives weighted to maximize short-term financial return. The lower rate of successional transition on private land (1.1%) compared to public land (1.5%) probably is a consequence of some private cut units changing to deciduous forest types, such as red alder and bigleaf maple, or staying longer in low-density conifer/shrub stands. These deciduous forest types would eventually succeed to closed canopy conifer-dominated types or sometimes stay in semipermanent, open, conifer-shrub types. Forest management practices on some private

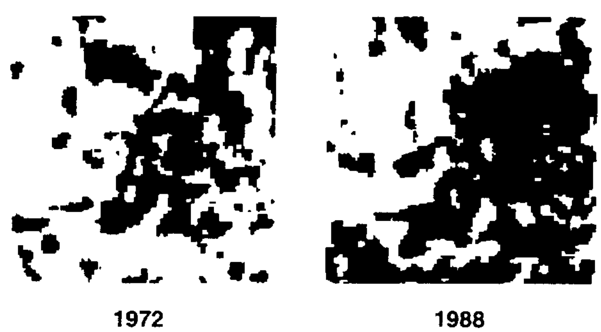

FIG. 7. Example of patterns of regrowth and cutting of closed-canopy conifer forest (■) on private land between 1972 and 1988.

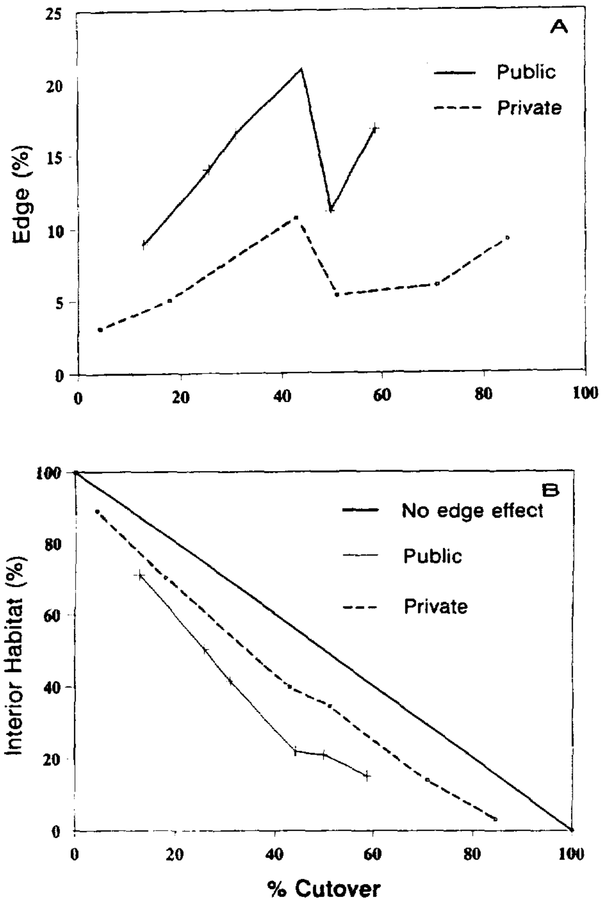

FIG. 8. Relation between percentage of cutover and percentage of interior habitat (A) and percentage of edge (B) for six public and six private sublandscapes (2500 ha) from either 1972 or 1988 conditions.

lands in the 1950s and 1960s resulted in sparse conifer regeneration and dense regeneration of red alder, which can outcompete conifers on some sites. Conifer regeneration failures on private lands led to passage of the Oregon Forestry Practices Act in 1971, which requires land owners to replant lands to commercial tree species within 2 yr. Recent regeneration practices have resulted in shortening the time between clear-cutting and the development of well-stocked plantations of merchantable conifers on most private lands. Sparsely regenerated conifer plantations represent <5% of the industrial private land but as much as 30% of the nonindustrial private lands of central western Oregon (Greber et al. 1990, J. Ohmann, *personal communication*).

This forest landscape is not in steady state (unchanging proportions of forest types) or in a regulated forest condition (i.e., equal proportions of all age classes). A regulated forest condition is not projected for ≈40 yr, based on current age-class distributions and assuming a 100-yr rotation on public land, a 55-yr rotation on private lands, and canopy closure at 40 yr (Fig. 9). The proportion of landscape in CF in a regulated forest condition will depend on the rotation length and the age at which canopy closure occurs. If public lands were to use a 200-yr rotation, then 80% of the landscape would be >40 yr (Fig. 9). If canopy closure were accelerated on private lands and occurred at 30 yr, which is already the case on many private lands, then 40% of that ownership would be in CF (Fig. 9). However, given the likelihood of natural disturbances and future changes in policy, the manager's ideal of a fully regulated forest will probably remain a theoretical target.

Comparison with natural disturbance regimes

The natural disturbance regime of the study area is characterized by both fine-scale (<1 ha) and coarse-scale (>1 ha) disturbances (Spies and Franklin 1988). In this study we focused primarily on coarse-scale disturbances that historically were fires. The natural fire regime of the study area is incompletely understood, but it appears to be complex in terms of frequency, intensity, and patch sizes. In a study of the fire history from 1150 to 1900 AD of two 1940-ha landscapes within our study area, Morrison and Swanson (1990) estimated natural fire rotations of 150 and 276 yr for moderate- to high-intensity fires. Some sites burned every 20 yr, while others burned only once in 400 yr. Fire patches were irregular in shape and intensity, with most fires killing only a portion of the trees within a patch. Patch sizes were highly variable but were typically <10 ha.

In contrast, the management disturbance regime we observed is several times more frequent and much more severe in terms of live trees and coarse woody debris

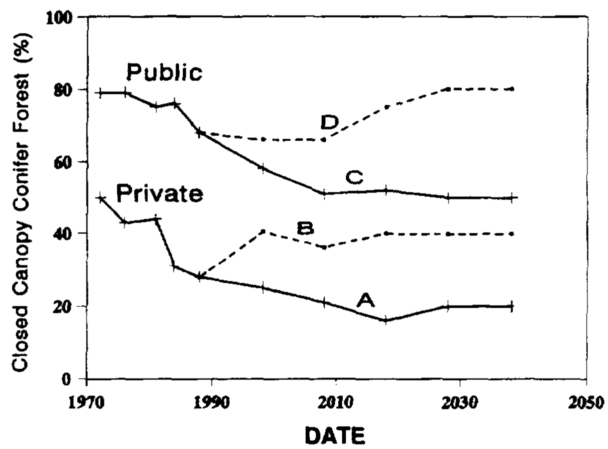

FIG. 9. Past and future projected changes in the proportion of study area covered by closed canopy conifer forest (>40 yr) by ownership. The projections assume a 55-yr rotation on private land (A) and a 100-yr rotation on public land (C). Alternative future proportions of closed-canopy forest include a 200-yr rotation age (D), and closed-canopy conditions occurring at 30 yr (B). The data from 1972–1988 are from this study and the projections are based on Greber et al. (1990) for private lands and the Willamette Forest Plan (Anonymous 1990) for public lands.

left following the disturbance (Spies and Cline 1988). Clearcut patch sizes appear to be similar to the typical fire patch size; however, the distribution of clearcut patch sizes is probably narrower, lacking disturbance patch sizes >100 ha. However, the higher frequency of clear-cutting cumulatively creates large patches of early successional conditions in a relatively short period of time.

The high disturbance rates create a higher proportion of the landscape in an early successional state than on average during the last several hundred years. This results in increases in populations of early successional species at a landscape scale and an increased density of seed rain and establishment of those species in disturbance patches. While these species are not likely to invade the areas of closed-canopy forests, through mass effects they may become more common in natural disturbances such as treefalls and wildfires that occur within natural forest areas. Also, contact zones or edges between early successional deciduous forests and closed-canopy coniferous forests will be more common, resulting in increased edge effects from microclimatic change (Chen 1991) and predation and competition. Finally, the short return intervals between major disturbances will not allow late successional ecosystems to develop and species and processes dependent on longer intervals will be lost on sites where cutting occurs at short intervals.

Comparison with other landscape change studies

Comparable studies of forest landscape change have been done in the tropical forest areas where cutting rates range from <0.5% (Nelson et al. 1987) to 1.7% (Fernside 1982: Table III) to >7.7%/yr (Sader and Joyce 1988). In many of these tropical landscapes, however, the deforestation is a result of agricultural expansion, whereas in Northwest temperate conifer forests, the change is typically from one forest condition to another. The loss of mature and old-growth forest habitat in the study area during this period, however, was at least semipermanent because current rotations and practices do not allow for redevelopment of this forest type.

The landscape dynamics we observed differed in several respects from the dynamics of the Superior National Forest in northern Minnesota between 1973 and 1983 (Hall et al. 1991). In the Minnesota landscape, the percentage of nonwilderness in mid- to late-successional "mixed and conifer states" increased at an annual rate of 0.2% (55 to 57%) between 1973 and 1983; on nonwilderness public lands in Oregon, the percentage of the mid- to late-successional conifer type decreased at an annual rate of 0.7% (79 to 68%) between 1972 and 1988. Stability of individual mid- to late-successional patches, however, appeared to be lower in Minnesota than in Oregon; 54 and 51% of the mixed and conifer types in Minnesota remained or returned to those types during the 10 yr, but in Oregon, 81% of the conifer type was still in that condition after 16 yr. The annual rates of major disturbance (change to "Clearings," "Regenerating," or "Broadleaf" types in the Minnesota study) in the mixed and conifer types on nonwilderness lands in Minnesota were 2.7 and 1.8%, but in Oregon the annual disturbance rate to the conifer type was 1.2%. Wilderness lands in both areas experienced lower disturbance rates than the nonwilderness lands; however, the Minnesota wilderness experienced higher disturbance rates than did the Oregon wilderness (0.7 vs. 0.2% for the conifer types).

Comparison with the checkerboard model

Although the relations of edge and interior habitat to percentage cutover in the public, nonwilderness landscapes generally supported the simple checkerboard model of Franklin and Forman (1987), their model clearly could not account for all of the complexity of landscape pattern and dynamics in a real landscape. On National Forest lands, the dispersed-cutting system did not result in a uniformly dispersed pattern at all scales, as the model assumed. For example, large-scale aggregations of 3000–20 000 ha of interior forest were still present in 1988. These interior forest aggregates, although they are not continuous uncut forest blocks, do retain a matrix that may provide adequate amounts and connectivity of habitat for relatively mobile organisms, such as the Spotted Owl (D. Johnson, *personal communication*), that can stay within the matrix of connected mid- to late-successional conifer forest, or cross short distances of young forest plantations.

The Franklin–Forman model did not include forest stand regrowth, which means that the highly cutover states simulated in the checkerboard model are never reached in a real landscape, at least for areas larger than several hundred hectares. In addition, regrowth of young stands reduces the distance that edge effects occur in adjacent taller stands and lessen the negative effects of cutting for processes and species that return to predisturbance conditions when the conifer canopy closes. On public lands where the predominant rotation is currently planned at 80 yr, the maximum area in canopy closure (assuming closure by 40 yr) would be ≈50% (Fig. 8). Based on this study and the simulations of Franklin and Forman (1987), a landscape with 50% cut in a staggered setting pattern would be close to or past the maximum percentage of edge in the landscape.

Franklin and Forman (1987) predicted landscape pattern thresholds at 30, 50, and 70% cutover for the dispersed patch cutting model. The 30% threshold was the point at which average forest patch size starts to decline because the original forest patch is so perforated that it begins to fragment into separate patches. Lehmkuhl et al. (1991), however, argue that the actual area of the forest patch that is available as habitat decreases linearly between 0 and 30%, as habitat within the patch is changed through cuts in the forest matrix. Consequently, decline in patch area begins with the first cut, and the 30% threshold from the checkerboard model may be more geometric than ecological, although what effect the occurrence of the first few breaks in the original matrix have on organism dispersal is not clear. The 50% cutover threshold—the point of maximum edge density (length of edge per unit area) in the landscape and point of increase in cutover patch size—we observed at 40% cutover in our subsamples. The 70% cutover threshold—the point at which the landscape becomes one contiguous cutover patch—was not observed at the scale of our subsampling. Because of the nonuniform application of the staggered-cutting model in the actual landscape, thresholds in patterns do not occur exactly where the checkerboard model would predict, or they do not occur at all. Furthermore, the occurrence of pattern and function thresholds will be scale dependent (Grant 1990). Small areas may be cut-over relatively rapidly, but, at a larger scale, they may be part of larger, relatively uncut aggregates. This scale-dependent pattern in the actual implementation of the staggered setting system means that effects of cutting patterns on hydrology or wildlife dispersal must be examined across a range of landscape sizes.

Influence of cutting rate on landscape pattern

The influence of cutting pattern on the amount of edge and interior forest has been documented by Franklin and Forman (1987) and is supported by the results of the ownership comparison in this study (Fig. 8). The influence of cutting rate on amount of edge and interior forest, however, has not been addressed in other studies, and the results of this study suggest that cutting rate can have a greater effect on the amount of edge and interior forest in a landscape than cutting pattern. For similar rates of cutover in the subareas, private land had more interior habitat and less edge habitat than public nonwilderness land, which would be expected because private lands tend to use larger clearcuts than public lands and are not constrained by dispersion rules. However, when the two landscapes as a whole were compared, the private land had much less interior forest and much more edge (at least for 1972–1984) than the public land. This difference results from the fact that cutting on private land has progressed further and more rapidly than it has on public lands. While it is obvious that higher rates of cutting result in lower amounts of interior habitat, discussions of alternative landscape management approaches in the region have typically focused on the effects of alternative cutting patterns on habitat (Franklin and Forman 1987, Swanson and Franklin 1992). While altering cutting patterns rather than rotations may be more economically viable, lengthening timber rotations will provide greater areas of older forest and interior habitat.

Stages of landscape pattern dynamics

Fragmentation is only one stage of landscape pattern dynamics that result from the simultaneous operation of disturbance and regrowth. If the rate of disturbance exceeds the rate of regrowth, large forest areas will become increasingly perforated with disturbance and early successional patches. In this perforation stage, gaps occur in the forest matrix but the forest remains connected. Continued relatively high rates of disturbance will lead to a fragmentation stage in which the perforated habitat is broken into isolated patches. A third stage of pattern dynamics occurs when the last remnant habitat patches are lost through disturbance. The spatial process of forest habitat redevelopment occurs in an inverse manner. First, small nuclei of forest habitat reappear as small, widely separated patches in the landscape. This process of nucleation is followed by a stage of coalescence in which the smaller nuclei expand and coalesce into large patches that eventually can completely cover the area, if disturbance rates are low. Because disturbance and regrowth operate at the same time in a landscape, the stages described above do not necessarily occur in sequence.

The study area is in the perforation to early fragmentation stages of landscape dynamics. At this point in landscape development, community responses are probably characterized by increases in species richness, as early successional species increase, but coniferous forest species and interior species may still find adequate habitat. Increases in bird species diversity have been observed in Douglas-fir stands 40–500+ yr old as the percentage of clearcuts in the landscape increased from 0 to 50% (Lehmkuhl et al. 1991). In the early stages of cutting, relatively mobile organisms whose habitat is lost may move to adjacent areas of suitable habitat, resulting in a packing effect (Whitcomb et al. 1981). Lehmkuhl et al. (1991) have observed increased bird abundances in old-growth Douglas-fir stands in landscapes with higher percentages of clearcuts than in landscapes with low percentages of clearcuts. The early stages of cutting may also be characterized by increases in the occurrence of mobile early-successional grazers, predators, and competitors within the forest matrix. For example, Great Horned Owls (*Bubo virginianus*) in the Pacific Northwest, which prefer relatively open habitat and forest edges for hunting (Forsman et al. 1984, Voous 1989), may increase predation on interior forest species that occur near edges or move across open areas within the forest matrix. Barred Owls (*Strix varia*), which may compete (Hamer 1988) or hybridize with Spotted Owls, are more tolerant than Spotted Owls of landscapes composed of a mix of early- and late-successional habitats (Taylor and Forsman 1976). The recent expansions in Barred Owl populations in the Pacific Northwest (Taylor and Forsman 1976) may be facilitated by the increased juxtaposition of early- and late-successional habitats.

Management implications

We have presented a simplified landscape analysis that emphasizes rates and patterns of cutting and closed-canopy forest. Our analysis is too simplified for comprehensive ecosystem management, which would need to include the considerable biological and environmental variation that occurs within the simple black and white patterns we characterized. However, the simplified analysis provides considerable information on the recent rates and major patterns of forest landscape dynamics in this area and has several management implications.

First, as the map of interior forest demonstrates, several large remnant concentrations of interior forest habitat are emerging in the managed landscape. Where management for interior forest habitat is a goal, these concentrations can serve as the foundation of an interior forest habitat network (Noss and Harris 1986). Most of these areas are currently on public lands, such as wilderness and Research Natural Areas, that will not be cut.

Second, if the intervening lands between the interior forest concentrations are to be managed for late successional interior forest habitat, special management practices will be required because much of the area is relatively fragmented. Management for interior forest conditions in the intervening lands will require regrowth of cutover areas while maintaining current interior forest remnants. The process would require delaying or excluding cutting in some areas and concentrating it in other areas. The degree of fragmentation should not be the sole criterion in setting priorities for cutting, however. Although highly fragmented forest areas may be considered ecologically less valuable now, many are in highly productive and species-rich, low-elevation sites or in concentrations of old, diverse forests, where logging started very early. The fragments in these areas could be valuable remnants in a strategy to develop interior forest blocks of mid- to late-successional forest on productive and biologically diverse sites. The current interior forest fragments can also serve as linkages for late-successional organisms moving between large interior forest areas.

Third, the results document that the lower elevation portions of the landscape on private lands will experience higher rates of disturbance and greater proportions of early-successional habitat and species than adjacent public lands. The ecological effects of this juxtaposition of large areas of different forestry land management schemes are not known, but could include mass effects for both animals and plants (Cody 1989). At the scale of the study area, the private lands may act as a source of early successional terrestrial and aquatic species that could disperse into the less frequently disturbed public lands. At a larger scale, the study area is near the center of a human land-use and disturbance gradient that begins to the west of the study area in the urban and agricultural lands of the Willamette Valley and moves through private forest lands into public nonwilderness lands and into wilderness lands at the highest elevations. In this context, the private forest lands could be seen as a buffer against the movement of non-native species and materials from the highly altered non-native aquatic and terrestrial ecosystems of the agricultural and urban lands in the Willamette Valley to the relatively unaltered native ecosystems on public lands.

The patterns of interior forest observed on this portion of private and public lands illustrate the importance of existing conditions in designing future landscape patterns for interior-sensitive species. The development of any plan for managing landscape structure and dynamics will need to identify the opportunities and constraints of existing patterns. Rarely will the manager start with a uniform and clean slate—either a completely intact landscape or a completely cutover landscape. Idealized reserve design systems (Harris 1984) will be useful as a long-term goal, but for the interim, landscape managers will need to practice the art of working with the pattern of what they have inherited from nature and previous managers.

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