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LAND OWNERSHIP AND LAND-COVER CHANGE IN THE SOUTHERN APPALACHIAN HIGHLANDS AND THE OLYMPIC PENINSULA[1]

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Abstract. Social and economic considerations are among the most important drivers of landscape change, yet few studies have addressed economic and environmental influences on landscape structure, and how land ownership may affect landscape dynamics. Watersheds in the Olympic Peninsula, Washington, and the southern Appalachian highlands of western North Carolina were studied to address two questions: (1) Does landscape pattern vary among federal, state, and private lands? (2) Do land-cover changes differ among owners, and if so, what variables explain the propensity of land to undergo change on federal, state, and private lands? Landscape changes were studied between 1975 and 1991 by using spatial databases and a time series of remotely sensed imagery. Differences in landscape pattern were observed between the two study regions and between different categories of land ownership. The proportion of the landscape in forest cover was greatest in the southern Appalachians for both U.S. National Forest and private lands, compared to any land-ownership category on the Olympic Peninsula. Greater variability in landscape structure through time and between ownership categories was observed on the Olympic Peninsula. On the Olympic Peninsula, landscape patterns did not differ substantially between commercial forest and state Department of Natural Resources lands, both of which are managed for timber, but differed between U.S. National Forest and noncommercial private land ownerships. In both regions, private lands contained less forest cover but a greater number of small forest patches than did public lands.

Analyses of land-cover change based on multinomial logit models revealed differences in land-cover transitions through time, between ownerships, and between the two study regions. Differences in land-cover transitions between time intervals suggested that additional factors (e.g., changes in wood products or agricultural prices, or changes in laws or policies) cause individuals or institutions to change land management. The importance of independent variables (slope, elevation, distance to roads and markets, and population density) in explaining land-cover change varied between ownerships. This methodology for analyzing land-cover dynamics across land units that encompass multiple owner types should be widely applicable to other landscapes.

Key words: land use; land-cover change; landscape ecology; Olympic Peninsula; remote sensing, southern Appalachians; spatial analysis.

INTRODUCTION

Landscapes are dynamic mosaics of natural and human-created patches that vary in size, shape, and arrangement. Although considerable attention has been given to describing changes in landscapes through time (e.g., Johnson and Sharpe 1976, Whitney and Somerlot 1985, Iverson 1988, Turner and Ruscher 1988, Turner 1990*a*, Hall et al. 1991, Kienast 1993, LaGro and DeGloria 1992, and many others), few studies have attempted to understand economic and ecological influences on landscape structure (Turner 1987, Parks and Alig 1988, Parks 1991) and land ownership (Lee et al. 1992, Spies et al. 1994, Wear and Flamm 1993).

Several authors have noted the critical need for knowledge about why landscape changes occur and how environmental factors and market processes interact (e.g., Turner 1987, Baker 1989). In the southern Appalachian highlands, Wear and Flamm (1993) found that the likelihood of forest cover being disturbed was a function of (1) the type of owner; (2) environmental attributes such as slope, aspect, and elevation; and (3) locational variables, such as distance to roads or market centers, which related to the economics of forest harvest and residential development. In Rondonia, Brazil, Dale et al. (1993) demonstrated that land use and land-

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TABLE 1. Percentages of land by ownership category in the Little Tennessee (LTRB), Hoh (HORB), and Dungeness (DURB) River Basins.

Ownership	LTRB	DURB	HORB
U.S. Forest Service (USFS)	35	22	<1
Private (includes commercial and small private owners)	65	24	17
National Park Service (NPS)		32	58
Washington Department of Natural Resources (DNR)		<1	24
Wilderness		18	

cover changes were a function of individual parcel sizes and shapes, attributes of individual land owners, site characteristics such as soils and agricultural suitability, and the distances to the road network. In the Cascade Range in Oregon, Spies et al. (1994) found the greatest decline in coniferous forest on private lands, least in wilderness, and intermediate in public nonwilderness areas. Clearly, social and economic considerations are among the most important drivers of landscape change, and a broad-based understanding of landscape structure and function is essential for promoting integrated management where human sustenance and environmental integrity are considered part of the same system (Lee et al. 1992).

The developing paradigms of sustainability (e.g., Lubchenco et al. 1991) and ecosystem management (e.g., Agee and Johnson 1988, Overbay 1992, Slocombe 1993, Grumbine 1994) require understanding of landscape dynamics and ecological processes across coherent land units, which typically encompass multiple owner types. The units for which land management and planning are undertaken frequently bear little resemblance to the ecological systems within which they reside or to their connections to economic and social processes (Slocombe 1993). Rivers and streams, for example, frequently are used as jurisdictional boundaries, bisecting watersheds. Protected lands often are not sufficiently large to encompass natural disturbance dynamics and are embedded within a matrix of lands of mixed ownerships. Understanding landscape dynamics in areas of mixed ownerships is an important component of ecosystem management, yet managers and scientists have only begun to develop strategies that recognize ecological conditions outside jurisdictional boundaries.

We examined the influence of land-ownership patterns on landscape structure in two forest-dominated landscapes: the Olympic Peninsula, Washington, and the southern Appalachian highlands of western North Carolina. The Olympic Peninsula is dominated by extensive tracts of land controlled by few owners, whereas the southern Appalachian highlands region contains smaller tracts of land controlled by many different owners (Table 1). We recognize, as did Mladenoff et al. (1993), that comparison between landscapes, or between ownerships within the same landscape, limits the inferences that can be applied to the general class of forested landscapes. Nonetheless, comparisons between landscapes and within watersheds can provide useful and necessary information on the complex dynamics that occur in landscapes of mixed ownership. We are aware of only one other study that analyzed landscape patterns on adjacent public and private ownerships (Spies et al. 1994). Spies et al. (1994) focused on forest-cutting patterns, addressing the spatial patterns and rates of change of coniferous forest between 1972 and 1988 in and around the Willamette National Forest. They observed differences in cutting patterns between the public and private lands, with lower elevation portions of the landscape on private lands having higher rates of disturbance and a greater proportion of early successional habitats. In this study, we analyze land-cover patterns and dynamics on public and private lands in two different forested regions, and test for significant differences in how land-cover transitions vary between owners.

Public lands may be managed in various ways by different institutions. We recognize that the true owners of the public lands are the citizens of the country, state, or other political jurisdiction, and that agencies simply manage the lands for the public. For simplicity of presentation in this paper, however, we will use "land ownership" to refer to the entity managing public lands (e.g., National Forests, State Department of Natural Resources, etc.).

We hypothesized that land-ownership patterns, i.e., the abundance of land in different ownership classes, such as public and private, would have a discernible and predictable influence on landscape structure and land-cover change. This general hypothesis was examined by comparing landscape changes in our study regions during a 16-yr time period (1975 to 199 1). Two specific questions were addressed: (1) Does landscape pattern vary between federal, state, and private lands? (2) Do land-cover changes differ among owners, and, if so, what variables explain the propensity of land to undergo change on federal, state, and private lands? Landscape pattern was analyzed by ownership class within each watershed to address the first question. The relationship between land-cover changes and several explanatory variables by ownership class was examined to address the second question. Our analysis assumed that land-use choices are based on comparisons of revenues and costs of various land uses at each site. Observed changes in land cover (or the lack of a change) reflect these choices about land use and the ranking of land values.

The value of land in any particular use is defined by the prices for its associated goods and services and a set of cost factors. Our study areas are relatively small (in a market sense), so that we can assume that prices (e.g., the prices of delivered logs and agricultural products) and unit costs (e.g., the wage rate and costs of capital and energy) are constant throughout each watershed. Accordingly, these factors should not affect land use specialization within a watershed for a specific time period. However, there is a set of factors that are variable within each watershed and that define cost differentials between locations within the area. These factors include: (1) steepness of the site, measured as its slope; (2) elevation of the site, which serves as a proxy for vegetation and climatic differences; (3) distance between the site and the nearest road, defining access costs; and (4) distance between the site and the nearest market for its goods and services, measured along the road network. Distance to market defines a set of transportation costs for goods and services. Additionally, (5) population density in the neighborhood of a site should influence its comparative advantage in different uses. These variables define either the quality of the site (slope, elevation, population density) or the location of the site within a physical/human landscape (access distance and distance to market), and should therefore influence land rents and land uses (Katzman 1974). As part of the second question, we examine whether or not these explanatory variables influence land-cover change within the study areas by developing statistical models and testing three hypotheses:

Hypothesis 1. Temporal change in transition models.—Factors not included in the mode1 (e.g., changes in prices for wood and agricultural products, or changes in policies and laws) may vary in time and may shift the probability of land-cover transitions between periods (e.g., land-cover changes may differ between 1975–1980 and 1980-1986). Thus, we test the null hypothesis that the relationship between the explanatory variables and the probabilities of land-cover change did not change between periods.

Hypothesis 2. Effects of ownership on transition models.-We may similarly test for identical transition models between the ownerships represented in each watershed. Here, the null hypothesis is that the relationship between explanatory variables and the probabilities of land-cover change did not differ between owners.

Hypothesis 3. Effects of spatial variables on transition models.-We hypothesized that the five explanatory factors would be positively related to the costs of productive activities, in general, within the watershed (summarized in Table 2). A negative relationship was hypothesized between slope, elevation, distance to roads, and distance to market and the probability of forest conversion (e.g., land-use conversion or forest

TABLE 2. Hypothesized direction of marginal effects of independent variables on specific land-cover transitions. A plus indicates an expected positive relationship, and a minus indicates an expected negative relationship.

Variable	Land-cover transition					
	Forest to grass	Forest to unvegetated	Grass to forest	Grass to unvegetated	Unvegetated to forest	Unvegetated to grass
Elevation			+		+	+
Slope			+		+	+
Distance to road	–	–	+	–	+	+
Distance to market			+		+	+
Population	–	+	–	+	–	–

management), as well as the probability of agricultural land conversion to developed uses (transition from grassy to unvegetated cover). In contrast, we expected that increased population density would create more development pressure (e.g., conversion from forest or grassy covers to unvegetated cover), but would decrease the probability of forest harvesting. Conversely, population density would be negatively related to transitions from grassy or unvegetated cover to forest cover.

STUDY AREAS

We studied two forested landscapes: the Olympic Peninsula, Washington, and the Southern Appalachian Man and Biosphere (SAMAB) region, a multistate zone of cooperation within the U.S. Man and the Biosphere Program. These landscapes were selected because they reflect vastly different land-ownership patterns and may serve as microcosms for many land-cover changes observed in forested regions of temperate North America.

Southern Appulachiun Highlands

The SAMAB region encompasses the southern Appalachian highlands and extends approximately from Chattanooga, Tennessee, northeast to Roanoke, Virginia, crossing four states. Approximately 57% of the SAMAB region is held in small private ownerships, and U.S. Forest Service (USFS) lands account for another 20% of land ownership. Forested lands in the SAMAB region have experienced increasing demands for nonmarket services, and associated pressures to decrease timber harvests. The Great Smoky Mountains National Park is the most visited national park in the U.S. because of the tremendous human population within a l-d drive, and this recreation demand also affects adjacent national forests and private lands. The relatively small holdings of the national forests in the southern Appalachians are interspersed among many land owners and must be managed in the context of a regional mixed-ownership landscape.

Within the SAMAB region, we selected the Little Tennessee River Basin (LTRB) for intensive study. The

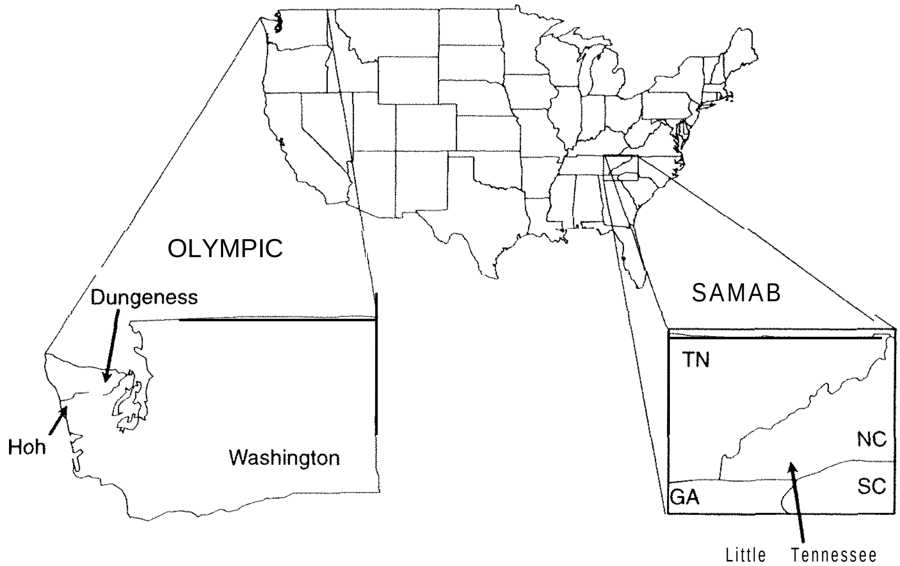

FIG. 1. Maps of the Olympic Peninsula and southern Appalachians, with locations of the Hoh, Dungeness, and Little Tennessee River Basins indicated.

116 090-ha LTRB is located primarily in western North Carolina, extending approximately from the Georgia–North Carolina border to Fontana Dam, just south of the Great Smoky Mountains National Park (Fig. 1). Although ≈ 10% the LTRB is located in north Georgia, we considered only the 103,635 ha located within North Carolina, because of limited availability of digital spatial data for the Georgia area. The LTRB is characterized by rugged topography and species-rich eastern deciduous forest. Franklin, North Carolina, the major developed area in the LTRB, is experiencing an influx of new residents. Tourism in Franklin, now a $50 million/yr business, is growing. Forest products remain an important industry in the LTRB, and the USFS is a major landholder, owning 35% of the watershed, primarily at the higher elevations (Table 1, Fig. 2). The rotation of forest cutting on the national forest lands ranges from 80 to 120 yr; harvest is primarily cove and upland hardwoods for saw timber. The USFS Coweeta Hydrological Laboratory, a Long-term Ecological Research (LTER) site, also is located within the LTRB.

Olympic Peninsula

The Olympic Peninsula, Washington, encompasses -1.6 X 10^6 ha, with the Olympic National Forest and Olympic National Park comprising nearly one-third of the land area. The pattern of land ownership on the Olympic Peninsula is quite different from that in the SAMAB region. Both public and private lands generally are held in large blocks, and the majority of the nonfederal lands are managed for timber production by the state of Washington's Department of Natural Resources (DNR) and by large private corporations. Small private ownerships comprise only ≈21% of the Olympic Peninsula, compared to ≈57% in the southern Appalachians. The controversy over the harvest of old-growth timber and conservation efforts focused on the Northern Spotted Owl (*Strix occidentalis)* in the Pacific Northwest have underscored the importance of understanding landscape dynamics on the Olympic Peninsula.

Two watersheds (Hoh and Dungeness River Basins) on the Olympic Peninsula were selected for intensive study (Fig. I), because a representative range of land-ownership classes did not occur in a single watershed. Both basins originate in the high elevations of the Olympic National Park, centrally located on the Peninsula. The 58 876-ha Dungeness River Basin (DURB) extends north from the Park to the town of Sequim. Major land-ownership classes in the DURB are the National Park Service, USFS, and small private ownerships in the Sequim area (Table 1, Fig. 2). The 78 007-ha Hoh River Basin (HORB) extends west from the Park to the Pacific Ocean. Major land-ownership classes in the HORB are the National Park Service, the Washington DNR, and large commercial private ownerships (Table 1, Fig. 2).

METHODS

Datubase development

Land-cover interpretation.-Land-cover patterns were interpreted from Landsat Multispectral Scanner (MSS) and Thematic Mapper (TM) imagery for four time periods in each region. In the LTRB, MSS imagery was dated 25 August 1975; 7 August 1980; 21 July 1986; and 7 May 1991. Dates of MSS imagery for the Olympic Peninsula, encompassing both the HORB and DURB, were 31 May 1975; 5 August 1980; and 3 August 1986. TM imagery from 16 September 1991 was

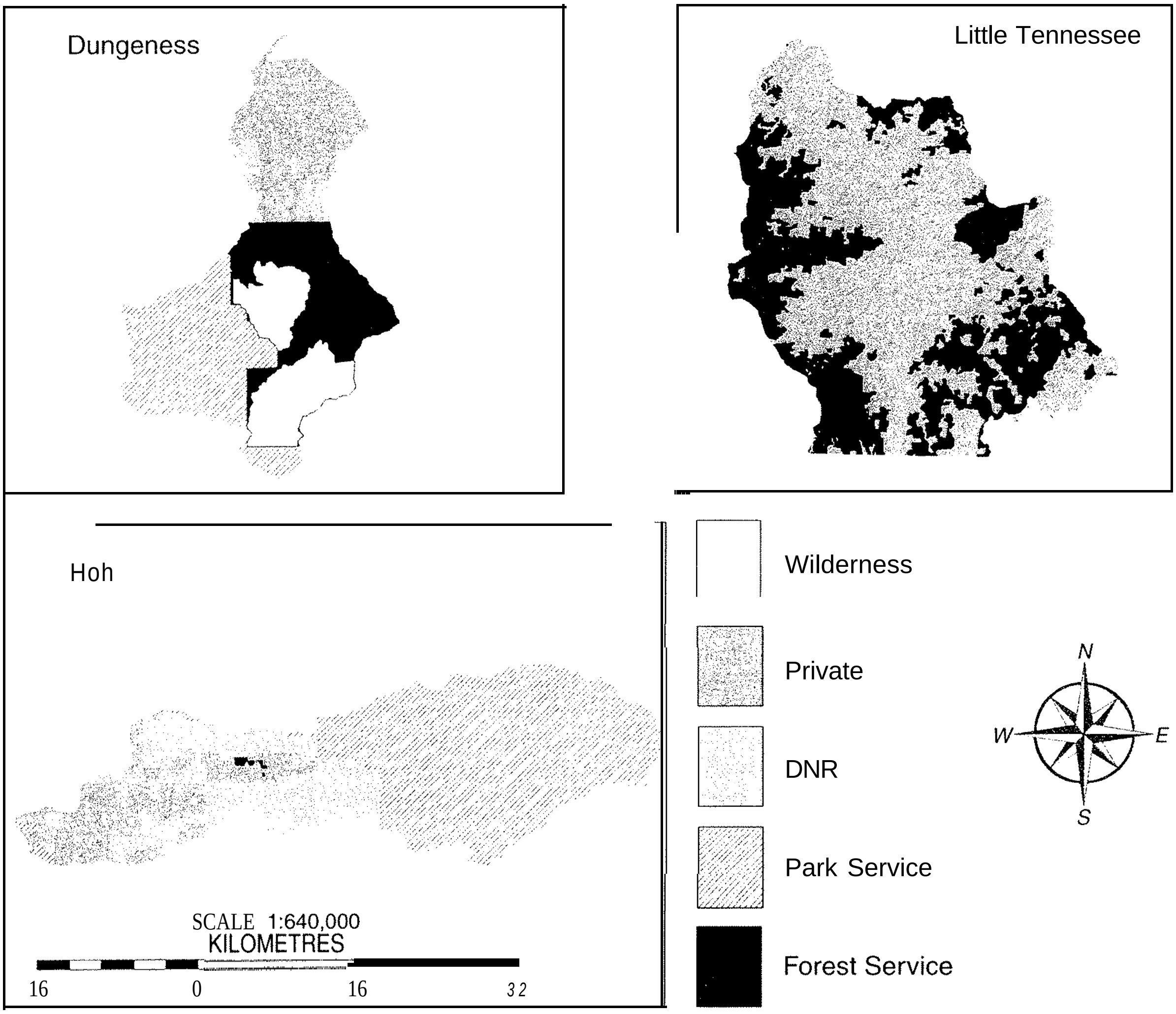

FIG. 2. General patterns of land ownership in the Hoh, Dungeness, and Little Tennessee River Basins.

used for the most recent time period on the Olympic Peninsula. Because the resolution of MSS imagery is -90 m, the landscape was represented as a 90-m grid and other data layers were resampled to a 90-m resolution, as necessary, within the Geographic Resources Analysis Support System (GRASS) geographic information system (GIS) (USA CERL 1991). Image interpretation was done similarly for each region within ERDAS (ERDAS 1994).

MSS data were classified prior to rectification and resampling. An iterative self-organizing algorithm in ERDAS (ISODATA) was run on MSS bands 2, 3, and 4 (the first band had heavy striping that could not be corrected). This resulted in 100 spectral signatures that were used by MAXCLAS, a maximum-likelihood classifier, to generate a final single-layer coverage based on spectral similarities. For the Olympic Peninsula, MSS land-cover layers were registered to the 199 1 TM scene using the image-to-image registration feature within ERDAS. Each image was rectified with a root mean square (RMS) error of less than 1 pixel (90 m).

For classification on the Olympic Peninsula, the TM image was cut to encompass the two study areas. The six bands used (l-5 and 7) were transformed using the TM Tasseled Cap into "brightness," "greenness," and "wetness" spectral indices (Crist and Cicone 1984), which is useful in determining structural characteristics in western hemlock and Douglas-fir forests (Cohen and Spies 1992). The "wetness" layer was minimally influenced by topographic shadowing and had the highest correlation to stand structure. Cohen and Spies (I 992) proposed that "wetness" be renamed to "maturity" to reflect the relationship to structural attributes of closed-canopy coniferous forests. The "maturity" layer is essentially a contrast between bands 5 and 7 in the mid-infrared range, whereas "greenness" expresses the difference between the first three visible bands (1, 2, 3) and the near-infrared band 4. "Brightness" is a weighted sum of all six bands together, weighing bands 3, 4, and 5 twice as much as the others.

A relative sun-incidence layer was developed from available 7.5' (1:24 000) digital elevation models (DEMs) to help reduce the effects of topographic shadowing on classification (Eby 1987). Where there were gaps in these larger scale data, smaller scale 1:250,000 DEMs, resampled to 25 X 25 m, were inserted. The ERDAS RELIEF program calculates the relative sun incidence using the sun's azimuth and elevation at the time of image acquisition (provided in the header data), resulting in a shaded relief layer that reflects conditions at the time of the satellite's overpass.

The brightness, greenness, maturity, and sun-incidence layers were combined in a separate image file, and an unsupervised classification algorithm (ISODATA) in ERDAS was used to generate 150 class signatures. A maximum-likelihood classifier (MAXCLAS) was run on the data using the ISODATA signatures as input. Based on field experience in the study areas and available aerial photos, the original 150 TM classes were condensed to 12 land-cover classes, which were used to generate random sample sites for accuracy assessment. A stratified random sample was generated using the ERDAS RANDCAT program. In total, 241 points were located in the Hoh River drainage, 179 were located in the Dungeness watershed, and 157 in the Little Tennessee watershed. These points were plotted on color prints of the original TM imagery and then were taken to the field for closer assessment. Eleven percent of the plots were visited in the field, and the remaining plots were evaluated using the most recent aerial photography available from DNR, NPS, and USFS. Results of the accuracy assessment were also used to further lump the land-cover classes to increase the overall accuracy. Despite the use of the sun-incidence layer, topographic effects were still apparent in the classified images. Final accuracies of the interpreted maps were >90%.

The final land-cover classes used in the study were as follows. In the LTRB, analyses were conducted on three classes from the final map layer (Fig. 3): (1) forest, which was primarily mixed hardwoods with occasional stands of pine; (2) grassy cover, including agricultural fields, pasture, lawns, and old fields; and (3) unvegetated, which included exposed soil, pavement, and developed areas. In the HORB and DURB (Fig. 3), the grassy and unvegetated classes were as described for the LTRB. For forest cover, however, coniferous forest was distinguished from deciduous/mixed forest (primarily alder regeneration plus some areas of cottonwood and big-leaf maple), resulting in four classes. Vegetation classes in the Olympics were similar to those developed by the Wilderness Society (Morrison 1992).

Other spatial data.—In addition to land cover, a set of spatial data layers including slope, aspect, elevation, land ownership, roads, and population density was assembled for each region and stored in the GRASS, Slope, aspect, and elevation were derived from 7.5' digital elevation model (DEM) data obtained from the U.S. Geological Survey for both regions. The DEM data were imported into GRASS at 25-m resolution; slope, aspect, and elevation layers were created within GRASS and resampled to a 90-m cell size. Land-ownership maps for part of the LTRB were obtained in digital form from the USFS (for the Wayah Ranger District), and the remainder of the ownership pattern in the LTRB was digitized manually from 1.24000 maps. For the Olympic Peninsula, data on general ownership were obtained from the Puget Sound River Basin Team, Washington DNR's GIS database, the USFS, National Park Service, and commercial owners. Primary and secondary road data were obtained for a single time period only from the 1990 TIGER lines, which were received as ARC/Info coverages then converted to GRASS. The North Carolina Center for Geographical Information and Analysis (NCGIA) provided the road data for the LTRB. Road data were used within GRASS to derive two additional data layers: (1) distance from each grid cell to the nearest road, and (2) distance from each grid cell, by road, to the nearest market center. Market centers included Franklin, North Carolina in the LTRB; Sequim, Washington in the DURB; and Highway 101 in the HORB. Finally, population density data were obtained from the 1990 census at the census tract level (irregular polygons of varying size) from TIGER/Line Census files from the NCGIA and the Washington Geographic Redistricting System. Grid cells occurring within a given census tract received the population density for that tract.

Landscape pattern analysis

Land-cover patterns were analyzed by computing indices that describe both overall landscape pattern and that of each land-cover class. Analyses for all four time periods were conducted separately for private and public ownership classes within each watershed by using the SPAN program (Turner 1990*a*, *b*). The proportion of the landscape area, p, occupied by each cover type was calculated. Nearest neighbor probabilities, $q_{i,j}$, which represent the probability of cells of land cover *i* being adjacent to cells of land cover j, were calculated by dividing the number of cells of type *i* that are adjacent to typej by the total number of cells of type *i*. The $q_{i,j}$ values were used in the contagion index (Eq. 2) and as a fine-scale measure of the degree of clumping in any cover type.

Two overall landscape indices adapted from O'Neill et al. (1988) were calculated. The first index, D, is a measure of dominance, calculated as the deviation from the maximum possible landscape or habitat diversity:

$$D = \left[H_{\max} + \sum_{i=1}^{m} (P_i)\log(P_i)\right] \Big/ H_{\max}, \qquad (1)$$

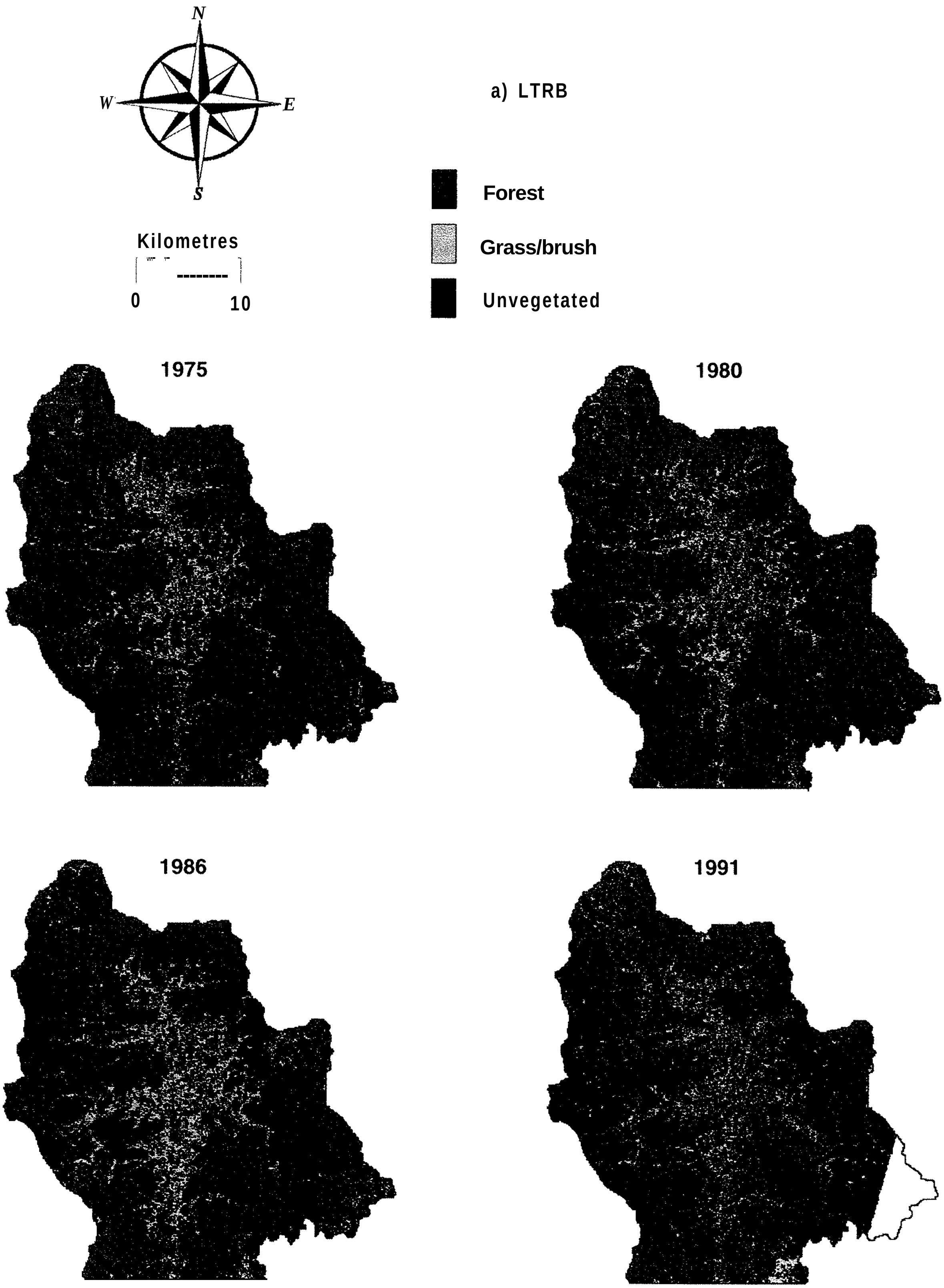

FIG. 3. Land-cover patterns interpreted from Landsat Multispectral Scanner (MSS) imagery for each of four time periods in (a) the Little Tennessee, (b) the Hoh, and (c) the Dungeness River Basins.

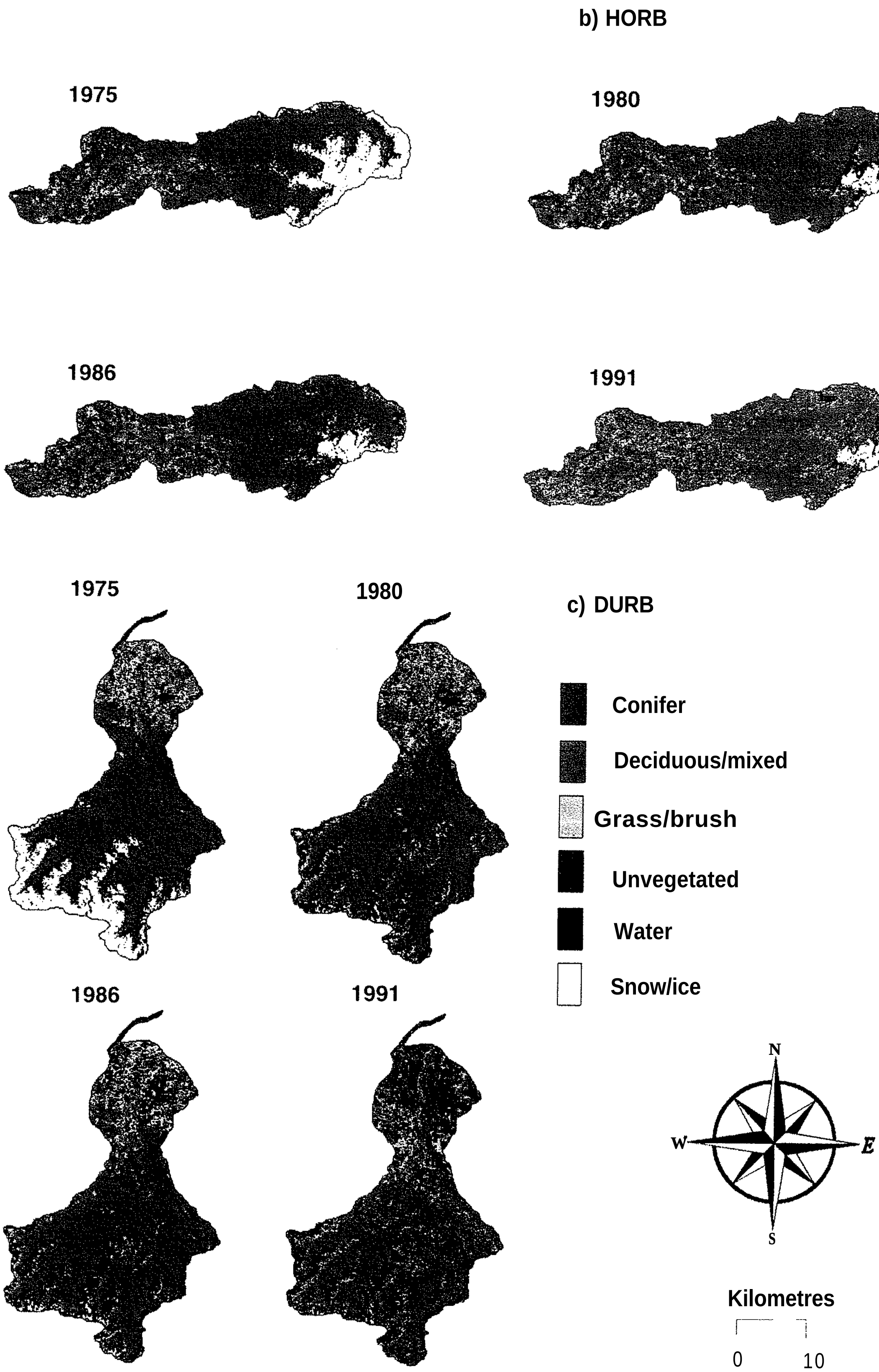

FIG. 3. Continued

where m = number of land-use types observed on the map, P_i is the proportion of the landscape in land use i, and H_{max} = log(m), the maximum diversity when all land uses are present in equal proportions. Values of D approaching one indicate a landscape that is dominated by one or a few land uses, and lower values nearing zero indicate a landscape with land uses represented in approximately equal proportions.

The second overall index, C, measures contagion, or the adjacency of land-cover types (O'Neill et al. 1988, Li and Reynolds 1993). The index is calculated from an adjacency matrix, Q, in which $q_{i,j}$ is the proportion of cells of type i that are adjacent to cells of type j, such that:

$$C = \left[K_{max} + \sum_{i=1}^{m} \sum_{j=1}^{m} (q_{i,j})\log(q_{i,j})\right] \Big/ K_{max}, \qquad (2)$$

where $K_{max} = m$ log(m) and is the absolute value of the summation of $(q_{i,j})\log(q_{i,j})$ when all possible adjacencies between land-cover types occur with equal probabilities. The index C will be zero when the pattern is dispersed and all possible adjacencies occur with equal probability. Values of C approaching one indicate a landscape with a clumped pattern of land-cover types.

The remaining metrics were computed separately for each land-cover category. For each cover type, every patch in the landscape map was identified and its area and perimeter recorded. A patch was defined as contiguous, adjacent (horizontally or vertically) cells of the same land cover; diagonal cells were not considered to be contiguous. The total number of patches, arithmetic mean patch size, area-weighted mean patch size, size of the largest patch, and number of single-cell patches were recorded for each cover type. The arithmetic mean patch size was calculated by simple division of the summation of the patch sizes, ΣS_i, by the number of patches, N,. The area-weighted mean patch size was computed by dividing the summation of the squared patch sizes, $\Sigma(S_i^2)$ by ΣS_i. Finally, an edge-to-area ratio was computed for each cover type by dividing the total number of edges by the number of grid cells of that cover type.

Models of land-cover change

Land-cover change was estimated on a per-grid-cell basis by using the time series of remote imagery. A regular grid was used to sample individual grid cells within each watershed, and the land-cover category at each of the four time periods plus the values of the five independent variables and the land-ownership class were recorded for each sampled grid cell. Sample sizes for each watershed were 4008, 8298, and 10459 grid cells for the LTRB, DURB, and HORB, respectively. The watersheds and ownership patterns were of differing size, and the spacing between individual points in the sample grids varied among watersheds (spacing: LTRB, six grid cells north-south and five grid cells east-west; DURB and HORB, three grid cells north–south and east-west). Because we focused on areas subject to human-influenced land-cover change, NPS lands and wilderness areas were excluded from the sample. Although including the "background" land-cover changes occurring in the absence (mostly) of anthropogenic factors might yield an interesting comparison, there was also substantial between-year variation for ice and snow cover in Olympic National Park, limiting its utility for this purpose.

We posited that the probability of land cover changing was related to the five spatially explicit variables: slope, elevation, distance to the nearest road, distance to the nearest market, and population density. Define X^k as the 6 X m matrix of these five independent variables plus a unitary constant for each of m sites with an initial land-cover class k. In addition, define Y_i^k as a discrete variable that defines land-cover change. If the cover class did not change, then $Y_i^k = 0$; $Y_i^k = 1$ or 2 if the cover class k changes to one of the two other cover classes. The probability of cover changing from k to j is defined as a function of site attributes:

$$\Pr(Y_i^k = j) = P_{ij}^k = F_{ij}^k(X_i^k) \qquad \text{for all } j, \qquad (3)$$

where Pr is the probability operator and F is a cumulative distribution function. The relationship in Eq. 3 can be estimated by specifying the functional form for F and fitting the equations to observed land-cover changes. We chose to examine this process using a multinomial logit formulation (e.g., Maddala 1983):

$$P_j^k = F_{ij}(X'\beta_j)$$
$$P_j^k = \frac{e^{X'\beta_j}}{Se^{X'\beta_S}}, \qquad (4)$$

where e is the base of the natural logarithms and β is a 6 X 1 vector of estimated coefficients. A multinomial choice model is appropriate because the dependent variable is discrete and takes on more than two values. Because the dependent variable is categorical (i.e., its values have meaning only as labels), either a probit model (based on a joint normal distribution) or a logit (based on the logistic distribution) may be applied. Monte Carlo analysis indicates that the logistic closely approximates the normal (Judge et al. 1985:778). Furthermore, the logistic has a closed-form solution that provides a practical advantage for predicting probabilities of land-cover change.

The multinomial logit defines F as a logistic cumulative-distribution function (CDF), which closely approximates the normal. Because the probabilities of transition must sum to one, one of the equations in Eq. 4 is redundant, so one equation was dropped from the estimation exercise. For our analysis, we estimated the equations for land-cover change (where $Y_i^k = 1$ or 2) and dropped the no-change alternative. The effects of variables on the "null transition" were then estimated using the estimated coefficients and the "summing-up" rule.

We defined models for three initial cover classes (indexed by k): forest, grassy, and unvegetated (coniferous and mixed forest were modeled separately in the HORB and DURB). Each of three different blocks of two equations defined by Eq. 4 were estimated separately by maximizing their respective likelihood functions using a nonlinear optimization algorithm implemented in the software package LIMDEP (Greene 1992). The three hypotheses were tested as follows by using estimation results.

Temporal change in transition models.—Land-cover transition models (Eq. 4) defined the probability of a transition as a function of five spatially variable factors. We hypothesized that the influence of other unmeasured factors (e.g., wood and agricultural prices, or changes in policies) would shift the average probability of transition, but would not affect the marginal effects of spatial variables on probabilities. That is, they would affect only the intercept terms in the vector $\boldsymbol{\beta}$. This hypothesis was tested by determining whether or not the relationships between the various explanatory variables and transition probabilities remained constant between periods. The test was implemented by assuming that, although the constant term might shift between periods to reflect changes in unmeasured variables that varied over time, all other coefficients would be constant between periods. This required constructing (1) a transition model for the null hypothesis, where all β coefficients except the intercept were held constant for the two periods, and (2) a transition model for the alternative hypothesis, where all β coefficients were allowed to vary between periods. The null model was, therefore, a constrained version of the alternative. Accordingly, we can test the hypothesis by comparing likelihood function values for the null and alternative models (see Judge et al. 1985: 182). The log likelihood ratio test was constructed as:

$$LR = -2 \ln(L_c/L_u), \tag{5}$$

where L_c and L_u are likelihood values for the constrained and unconstrained versions of the model. LR has a chi-squared distribution, with degrees of freedom equal to the number of constraints imposed to form the null hypothesis. The model for the alternative hypothesis is defined by using a dummy variable (D) that is equal to one for one period and equal to zero for the other:

$$F(\;) = (X'\boldsymbol{\beta} + DX'\boldsymbol{\gamma}).$$

Accordingly, the dummy variable allows for different relationships between probabilities and site attributes for the different periods. The null hypothesis is constructed by constraining all γ to zero.

To construct comparisons between periods of unequal length, we assumed that the probability of any transition occurring within a period was proportional to the length of the period. The lengths of our periods are: (1) 60 mo for 1975-1980, (2) 72 mo for 1980-1986, and (3) 58 mo for 1986–1991. We treated the periods 1975-1980 and 1986-1991 as essentially equivalent. To make the period 1980-1986 comparable to the others, we evaluated each observed transition with a draw from a uniform distribution (between 0 and 1). If the number was less than $60/72 = 0.8333$, then the transition was recorded. Otherwise, it was discarded.

Effects of ownership on transition models.—We similarly tested for identical transition models between the two ownerships. The model for the null hypothesis of identical transition models constrains all elements of β to be equal between ownerships. For the alternative hypothesis, all elements of $\boldsymbol{\beta}$ were allowed to vary between ownerships. The likelihood ratio test was used to construct the test.

Effects of spatial variables on transition models.—The general hypothesis that the spatial factors (elevation, slope, distance to a road, distance to market, and population density) explain cover transitions was examined by testing for the significance of the estimated models (we test for a significant difference from the null model defined by constraining all elements of $\boldsymbol{\beta}$, except the intercept, to zero) and defining the associated likelihood ratio test. The significance of individual variables in explaining specific land-cover transitions was also tested (hypotheses are summarized in Table 2). With a linear regression, we could simply test the significance of the estimated coefficients (β). However, with the multinomial logit model, the estimated coefficients and their variances do not necessarily correspond to the sign, relative magnitude, or significance of the referenced transition probability. Marginal effects of individual variables and their variances were calculated from the estimated CDF (e.g., $ME_i = \delta F/\delta X_j$) and variance-covariance matrix for $\boldsymbol{\beta}$. These estimates depend on the value of the independent variables (X), and we set all X to mean values. This generates a test that is conservative: i.e., a marginal effect may prove to be insignificant with independent variables set at mean values but significant for some other plausible combination of values. We tested for significance at the $P \leq 0.05$ level but, in light of the conservative nature of the test, also report the results for the $P \leq 0.20$ level.

RESULTS

Land ownership and landscape pattern

Little Tennessee River Basin.—Landscape patterns differed subtly between USFS and private lands in the LTRB. Forest was the dominant land cover in both ownerships, although forest was in lower proportions on private lands (0.78-0.86) than on USFS lands (0.96-0.98) (Table 3). Dominance and contagion were always greater on the USFS lands than on private lands (D = 0.91, 0.92, 0.86, 0.96 on USFS, and 0.66, 0.59, 0.54,

TABLE 3. Indices of landscape pattern for USFS and private land ownerships in the Little Tennessee River Basin, North Carolina, from maps derived from Landsat MSS imagery, where p_i is the proportion of the watershed in the specified cover type; q_{ii} is the probability of adjacency for grid cells of the same cover type being adjacent in the horizontal or vertical direction; and $N_{1\text{-cell}}$ is the number of single-cell patches.

Index	USFS 1975	USFS 1980	USFS 1986	USFS 1991	Private 1975	Private 1980	Private 1986	Private 1991
a) Forest cover								
p_i	0.98	0.98	0.96	0.96	0.86	0.83	0.78	0.85
q_{ii}	0.99	0.99	0.97	0.98	0.93	0.91	0.90	0.90
No. patches (N)	107	111	134	119	409	549	734	462
Normalized N†	107	111	134	119	221	297	397	250
Average patch size‡	404	391	317	350	173	124	88	145
Weighted average size‡	7670	7763	7686	7632	42868	39460	1.5923	44924
Normalized $N_{1\text{-cell}}$†	36	37	55	42	117	156	215	143
Edge/area	0.30	0.29	0.34	0.34	0.42	0.49	0.53	0.52
Largest patch size‡	14114	14273	14226	14072	54525	51189	28351	54477
b) Unvegetated cover								
p_i	0.012	0.006	0.008	0.002	0.030	0.055	0.048	0.017
q_{ii}	0.41	0.27	0.19	0.23	0.37	0.41	0.41	0.32
No. patches (N)	183	145	228	48	940	1472	1301	631
Normalized N†	183	145	228	48	508	796	703	341
Average patch size‡	2.9	1.8	1.5	1.6	2.6	3.1	3.0	2.1
Weighted average size‡	7.0	3.2	2.3	3.0	11.3	20.8	21.7	16.9
Normalized $N_{1\text{-cell}}$†	89	88	157	35	279	416	390	230
Edge/area	2.44	2.98	3.27	3.11	2.54	2.37	2.38	2.72
Largest patch size‡	29	12	7	9	71	212	193	125
c) Grassy/brushy cover								
p_i	0.011	0.014	0.031	0.036	0.105	0.115	0.167	0.129
q_{ii}	0.32	0.31	0.31	0.33	0.45	0.43	0.51	0.37
No. patches (N)	237	307	651	711	2196	2673	2545	3652
Normalized N†	237	307	651	711	1187	1444	1376	1974
Average patch size‡	2.1	2.0	2.1	2.2	3.9	3.5	5.4	2.8
Weighted average size‡	5.9	5.1	5.3	4.8	22.9	14.9	94.9	16.0
Normalized $N_{1\text{-cell}}$†	150	190	414	420	561	765	738	1106
Edge/area	2.84	2.88	2.84	2.76	2.22	2.31	1.98	2.54
Largest patch size‡	29	27	28	21	150	91	529	273

† Number of patches was normalized for differences in area of each ownership class by dividing the actual number of patches on private lands by the ratio of private : USFS lands (0.65/0.35 = 1.85); this permits the number of patches to be compared between the ownerships.

‡ Units are 90 X 90 m grid cells.

0.65 on private lands; C = 0.67, 0.64, 0.51, 0.48 on USFS, and 0.38, 0.39, 0.36, 0.36 on private lands for 1975, 1980, 1986, and 1991, respectively). These indices indicate a more even distribution and less clumped spatial pattern of land-cover classes on private lands than on USFS lands.

The spatial arrangement of land-cover classes on USFS lands remained remarkably constant from 1975 to 1991. Although the number of forest patches fluctuated somewhat, there was little change in the area-weighted average patch size, size of the largest patch, and number of single-grid-cell patches for forest cover (Table 3). Unvegetated and grassy/brushy cover types comprised a total of 2-4% of the USFS lands between 1975 and 1991, and the most notable change was an increase in the number of grassy/brushy patches and a decrease in the number of unvegetated patches. Patch sizes of these two cover types, both average and area-weighted average, showed little change (Table 3).

More variability in landscape pattern through time was observed on private lands (Table 3). Forest cover declined by 8% between 1975 and 1986, then increased in 1991 to levels comparable to those in 1975. The number of forest patches increased between 1975 and 1986, with an almost twofold increase in the number of single-cell forest patches. Average, area-weighted average, and largest patch size for forest cover decreased during the 1975–1986 period, and then increased in 1991. Unvegetated areas accounted for 2-5% of the private land cover between 1975 and 1991, and grassy/brushy cover accounted for IO-17%. Area-weighted average sizes of patches of nonforest cover varied substantially through time (e.g., sixfold for grassy/brushy cover) (Table 3).

Differences between the USFS and private lands reflect the greater abundance of nonforest cover classes and larger relative number of small patches on private lands (Table 3). Forest edge-to-area ratios were greater on private lands compared to USFS lands. The average size, area-weighted average size, and number of patches of nonforest cover were substantially larger on private lands compared to USFS lands. It is interesting to note, however, that the size of the largest patch of contiguous forest was greatest on private lands, reflecting

TABLE 4. Indices of landscape patterns for USFS and private ownership types in the Hoh River Basin, Washington, based on Landsat MSS imagery. Indices are defined as in Table 3.

Index	DNR 1975	DNR 1980	DNR 1986	DNR 1991	Private 1975	Private 1980	Private 1986	Private 1991
a) Coniferous forest cover								
p_i	0.56	0.54	0.40	0.50	0.26	0.27	0.22	0.39
q_{ii}	0.86	0.82	0.77	0.76	0.66	0.64	0.57	0.69
No. patches (N)	281	388	484	619	616	628	635	655
Normalized N†	199	275	343	439	616	628	635	655
Average patch size‡	46	33	19	19	7.0	7.1	5.9	10.1
Weighted average size‡	1631	850	480	1357	93	98	51	320
Normalized $N_{1\text{-cell}}$†	113	145	176	238	320	311	319	376
Edge/area	0.65	0.79	1.01	1.07	1.54	1.58	1.83	1.37
Largest patch size‡	2969	1762	1348	3531	372	444	187	746
b) Deciduous/mixed forest cover								
p_i	0.08	0.10	0.26	0.14	0.42	0.40	0.49	0.19
q_{ii}	0.53	0.51	0.65	0.49	0.74	0.71	0.74	0.49
No. patches (N)	460	544	694	877	407	464	399	944
Normalized N†	326	386	492	622	407	464	399	944
Average patch size‡	4.4	4.4	8.7	3.7	17.5	14.4	20.7	3.4
Weighted average size‡	17	22	121	26	580	322	486	62
Normalized $N_{1\text{-cell}}$†	159	199	228	372	205	235	175	589
Edge/area	2.06	2.00	1.49	2.15	1.18	1.25	1.14	2.17
Largest patch size‡	56	122	466	139	1750	1041	1267	332
c) Grassy/brushy cover								
p_i	0.09	0.14	0.19	0.16	0.17	0.16	0.16	0.21
q_{ii}	0.51	0.52	0.48	0.35	0.58	0.54	0.47	0.42
No. patches (N)	522	718	1029	1556	503	547	811	1124
Normalized N†	370	509	730	1103	503	547	811	1124
Average patch size‡	4.2	4.7	4.4	2.5	5.7	4.8	3.4	3.1
Weighted average size‡	18	22	25	10	64	53	26	15
Normalized $N_{1\text{-cell}}$†	193	274	396	688	268	319	501	667
Edge/area	2.08	2.00	2.11	2.66	1.82	1.95	2.26	2.41
Largest patch size‡	72	99	169	58	252	250	122	73
d) Unvegetated cover								
p_i	0.25	0.21	0.15	0.20	0.14	0.18	0.11	0.20
q_{ii}	0.72	0.66	0.63	0.67	0.66	0.67	0.62	0.66
No. patches (N)	518	583	574	634	368	397	328	426
Normalized N†	367	413	407	450	368	697	328	426
Average patch size‡	11.4	8.4	6.1	7.4	6.3	7.4	5.5	7.9
Weighted average size‡	87	82	54	64	129	138	39	53
Normalized $N_{1\text{-cell}}$†	189	218	238	264	205	227	159	217
Edge/area	1.22	1.74	1.58	1.44	1.55	1.48	1.74	1.53
Largest patch size‡	309	387	287	220	414	437	171	163

† Number of patches was normalized for differences in area of each ownership class by dividing the actual number of patches on private lands by the ratio of private : USFS lands (0.65/0.35 = 1.85); this permits the number of patches to be compared between the ownerships.
‡ Units are 90 X 90 m grid cells.

in part, the extent and spatial distribution of the ownerships themselves (Fig. 2).

Hoh River Basin.-Both DNR lands and private lands (with primarily commercial owners) in the HORB showed low-to-moderate levels of fluctuation in landscape pattern through time, but some differences in landscape structure between ownerships were observed (Table 4). In general, the dominance index was greater on DNR lands than on private lands (e.g., $D = 0.36$ and 0.25, respectively, in 1975), but contagion was similar between ownerships at each time period, ranging between 0.37 and 0.48. Coniferous forest lands generally occupied ≈50% of the DNR lands but only 22-40% of the private lands (Table 4). Coniferous forest was more aggregated on the DNR lands than on private lands, as indicated by both the nearest-neighbor probabilities and the edge-to-area ratios. Average patch sizes of coniferous forest were consistently greater on DNR lands than on private lands. The area of deciduous/mixed forest increased on DNR lands but decreased on private lands (Table 4), with private lands having larger patches. The proportion of the landscape occupied by grassy/brushy cover increased on both DNR and private lands, and the spatial pattern of this cover type was similar between the two ownerships. The spatial pattern of unvegetated cover also showed few differences between ownerships, with patch characteristics being similar, and fluctuations through time relatively small.

Dungeness River Basin.—Although the USFS and

TABLE 5. Indices of landscape pattern for USFS and private ownership types in the Dungeness River Basin, Washington, based on Landsat MSS imagery. Indices are defined as in Table 3.

Index	USFS 1975	USFS 1980	USFS 1986	USFS 1991	Private 1975	Private 1980	Private 1986	Private 1991
a) Coniferous forest cover								
p_i	0.73	0.83	0.82	0.73	0.17	0.23	0.18	0.11
q_{ii}	0.88	0.92	0.92	0.88	0.64	0.73	0.62	0.56
No. patches (N)	89	66	78	107	434	301	395	404
Average patch size†	130	200	167	110	6.7	11.3	8.0	4.7
Weighted average size†	6759	7551	7202	6229	196	216	80	69
No. 1-cell patches	66	45	44	70	257	138	191	239
Edge/area	0.56	0.40	0.40	0.58	1.59	1.21	1.59	1.86
Largest patch size†	8352	9211	8871	7746	649	551	356	185
b) Deciduous/mixed forest cover								
p_i	0.10	0.06	0.06	0.01	0.31	0.16	0.17	0.02
q_{ii}	0.30	0.43	0.48	0.22	0.58	0.56	0.54	0.35
No. patches (N)	716	298	220	128	819	460	552	186
Average patch size†	2.2	3.4	4.2	1.6	6.6	5.8	5.2	2.1
Weighted average size†	6.3	11.0	14.0	3.4	88	38	46	5.9
No. 1-cell patches	448	1.58	107	99	442	215	272	129
Edge/area	2.82	2.29	2.11	3.11	1.72	1.77	1.86	2.73
Largest patch size†	36	38	45	13	330	154	190	20
c) Grassy/brushy cover								
p_i	0.005	0.04	0.03	0.11	0.29	0.29	0.30	0.16
q_{ii}	0.32	0.38	0.40	0.31	0.59	0.51	0.55	0.45
No. patches (N)	39	245	183	772	587	877	745	793
Average patch size†	2.1	2.9	3.0	2.2	8.4	5.7	6.9	3.6
Weighted average size†	5.7	6.7	8.4	6.6	120	40	60	34
No. 1-cell patches	24	126	105	520	292	400	375	477
Edge/area	2.78	2.50	2.43	2.78	1.68	1.99	1.83	2.22
Largest patch size†	17	22	27	34	462	220	200	216
d) Unvegetated cover								
p_i	0.15	0.06	0.09	0.14	0.21	0.30	0.33	0.68
q_{ii}	0.63	0.50	0.62	0.60	0.61	0.69	0.70	0.87
No. patches (N)	321	250	211	387	501	393	407	197
Average patch size†	7.4	3.9	6.7	6.0	7.2	13.2	13.9	59.6
Weighted average size†	75	36	102	47	143	554	627	8717
No. 1-cell patches	177	132	112	217	246	195	198	120
Edge/area	1.58	2.12	1.64	1.70	1.59	1.24	1.23	0.55
Largest patch size†	80	706	544	228	627	1469	1770	10105

† Units are 90 X 90 m grid ceils.

small private ownerships accounted for similar proportions of the DURB (22% and 24%, respectively), the landscape patterns observed in these ownerships differed dramatically (Table 5). The USFS lands were dominated by coniferous forest cover, ranging between 0.73 and 0.83 of the landscape between 1975 and 1991. Private lands had a much lower proportion of coniferous forest land, ranging between 0.11 and 0.23, with a net decrease through time. The dominance index was greater on USFS lands than on private lands for all time periods (D = 0.52, 0.62, 0.62, 0.55 for USFS, and D = 0.20, 0.21, 0.21, 0.46 for private lands for 1975, 1980, 1986, and 1991). Contagion increased through time and was similar between ownerships, ranging between 0.42 and 0.65.

The USFS lands were characterized by moderate changes through time and no net loss of forest cover, whereas private lands exhibited substantial loss of forest. The abundance and spatial distribution of coniferous forest cover were relatively stable on USFS lands (Table 5). Deciduous/mixed forest cover decreased on USFS lands, with associated decreases in patch sizes; grassy/brushy cover increased; and unvegetated cover showed moderate fluctuation through time (Table 5).

Private lands, however, changed dramatically. The proportions of coniferous forest and deciduous/mixed forest on private lands declined substantially between 1975 and 1991, with concomitant large declines in average and area-weighted average patch sizes (Table 5). Patch sizes of conifers were much smaller on private lands than on USFS lands, and private lands generally had four to six times the number of conifer patches (Table 5). The largest patch of contiguous coniferous forest cover was an order of magnitude greater on USFS than on private lands. The proportion of the landscape in grassy/brushy cover on private lands also declined by about half during the time interval. In contrast to these declines, the unvegetated cover on private lands increased between 1975 and 1991 from 0.21 to 0.68 (Table 5). Increased connectivity of the unvegetated cover is evident in the increase of the $q_{i,i}$ values from 0.61 to 0.87, order-of-magnitude increases in average

TABLE 6. Tests for temporal change in transition models. Each entry is the log likelihood ratio for the test of identical transition models between the referenced periods, by initial cover class. ** indicates rejection of identical transition models at the $P \leq 0.01$ level. Blank cells indicate that the test could not be constructed, due to a limited sample size relative to the number of independent variables in the alternative model.

Lands, periods	df	Initial cover class: Conifer forest	Forest?	Grassy	Unvegetated
a) Little Tennessee River Basin					
Private lands					
1975-1980 vs. 1980-1986	10		19.38	19.40	33.62**
1980-1986 vs. 1986-1991	10		48.76**	20.34	20.62
U.S. Forest Service					
1975-1980 vs. 1980-1986	10		16.65		
1980-1986 vs. 1986-1991	10		27.49**		
b) Hoh River Basin					
Private lands (commercial)					
1975-1980 vs. 1980–1986	15	50.01**	37.94**	29.01	48.47**
1980-1986 vs. 1986-1991	15	33.15**	39.58**	37.87**	28.25
Washington Department of Natural Resources					
1975-1980 vs. 1980-1986	15	73.40**	54.74**	81.01**	67.91**
1980-1986 vs. 1986-1991	15	19.54	50.16**	38.39**	68.22**
c) Dungeness River Basin					
Private lands					
1975-1980 vs. 1980-1986	18	33.15	34.18	74.14**	18.83
1980–1986 vs. 1986-1991	18	20.02	39.04**	16.98**	47.67**
U.S. Forest Service					
1975-1980 vs. 1980-1986	18	19.87	24.38	18.72	48.64**
1980-1986 vs. 1986-1991	18	27.31	31.76	23.57	34.61

† Deciduous forest with occasional nine in Little Tennessee River Basin; mixed deciduous and forest in Hoh and Dungeness Basins.

and area-weighted average patch sizes, decreases in the number of single-cell patches, and a decrease in edge-to-area ratio.

Land ownership and landscape change: Little Tennessee River Basin

Temporal change in transition models.—The hypothesis of identical transitions for forest cover between 1980-1986 and 1986-1991 on private lands was rejected (Table 6a). However, we do not reject identical transitions for forest cover between 1975-1980 and 1980-1986. There was no significant difference for grassy cover in both cases, but for unvegetated cover we found a significant difference between 1975–1980 and 1980-l 986 but not between 1980-I 986 and 1986–1991.

A similar result for forest cover was observed on the USFS lands (Table 6b). That is, the overall relationship between site features and transition probability for forest land was not significantly different between 197% 1980 and 1980– 1986, but shifted between 1980-I 986 and 1986–1991. Tests for temporal change in grassy and unvegetated cover types could not be constructed for USFS lands because of limited degrees of freedom (i.e., there were very few observations relative to estimated parameters in the alternative model).

Effects of ownership on transition models.-We tested for differences in transition models between ownerships by comparing pooled and separate-effects models based on the tests of temporal change in transition models. Accordingly, we pooled data for the 1975–1980 and 1980-I 986 periods for forest and grassy cover, as they did not differ.

In both 1975–1986 and 1986-1991, the hypothesis of identical transition models for the private and public ownerships was rejected (Table 7), indicating structural dissimilarities in the spatial relationships for forest cover changes between USFS and private lands. However, there were no significant differences between the transition models for grassy and unvegetated cover on public and private lands. This may reflect the relatively small sample size for these types of cover on national forests.

Effects of spatial variables on transition models.—All models estimated for the LTRB were significant. On private lands, nine of the 30 marginal effects were significantly different from zero for the period 1975–1980 (Table 8). Slope was especially important in explaining these transitions. For the transition from forest to grassy cover, slope was negative, consistent with our expectations (see Table 2) of the effect of cost on timber harvest or development. Slope was also a significant

TABLE 7. Tests for differences in transition models between ownership classes in each watershed. Each entry is the log likelihood ratio (with a chi-squared distribution) for the test of identical transition models between ownerships. The degrees of freedom for each test differ for pooling periods 1975-1980 and 1980-1986 in some cases. ** indicates rejection of identical transition models at the $P \leq 0.01$ level. Blank cells indicate that the test could not be constructed due to a limited sample size relative to the number of independent variables in the alternative model.

Lands, periods	df	Initial cover class: Conifer forest	Forest†	Grassy	Unvegetated
a) Little Tennessee River Basin					
U.S. Forest Service vs. private lands					
1975-1986	14		48.23**	24.66	
1986-1991	12		50.63**	22.17	7.50
b) Hoh River Basin					
Washington Department of Natural Resources vs. private lands (commercial)					
1975-1980	15	42.05**	42.12**	44.60**	75.31**
1980-1986	15	47.68**	25.26	26.71	56.95**
1986-1991	15	35.28**	47.81**	24.46	19.46
c) Dungeness River Basin					
U.S. Forest Service vs. private lands					
1975-1980	15	34.63	51.39**		55.66**
1980-1986	15	40.43**	46.02**	67.92**	84.61**
1986-1991	15	45.56**	49.22**	44.60**	38.71**

† Deciduous forest with occasional pine in Little Tennessee River Basin; mixed deciduous forest in Hoh and Dungeness Basins.

factor for the transition from forest to unvegetated cover. For grassy cover on private lands in the period 1975-1980, transition back to forest was positively related to slope at the $P \leq 0.20$ but not the $P \leq 0.05$ level (Table 8), consistent with our expectations reflecting increasing costs of development. The same relationship was reflected in the negative effect of slope and distance to market on the grassy-to-unvegetated transition. However, the relationship between the grassy–unvegetated transition and elevation was significantly positive, whereas we expected a negative marginal effect. Transition from unvegetated to forest land was positively related to slope, consistent with expectations. Transition from unvegetated cover was positively related to elevation but negatively related to slope. The latter result is counterintuitive.

In contrast to results for the period 1975-1980, only three of 30 effects on private lands were significantly different from zero during the period 1986-1991 (Table 8). None of the spatially explicit variables had a significant effect on transitions from forest cover or on transitions from grassy and unvegetated cover to forest. For transitions from grassy to unvegetated cover, we found a negative relationship with elevation and slope and a positive relationship with population (the latter at only the $P \leq 0.20$ level). All three results were consistent with our expectations. The only significant relationship for the unvegetated-to-grassy transitions was reflected in a negative marginal effect for population.

Changes in marginal effects between periods may provide insights into the structural change observed in forest cover transitions between 1975-1986 and 1986-1991. In general, fewer spatial variables produced significant marginal 'effects in the period 1986-1991. To examine the resulting implications for forest-cover change, we plotted the probabilities of a forest-grassy transition for the three periods, relative to distance to roads, slope, and elevation (three variables producing significant marginal effects in 1975–1980). Probabilities were calculated with all variables held at sample means and the referenced variable varied over its observed range (Fig. 4). All relationships had the anticipated downward-sloping effect of the variable on a

TABLE 8. Tests of hypotheses regarding marginal effects of independent variables on specific land-cover transitions for private lands in the Little Tennessee River Basin. For each entry, the first sign represents the expected sign of the marginal effect on the referenced transition (from Table 2). The sign in parentheses is the marginal effect, calculated at the means of the other independent variables. * indicates that the calculated marginal effect is significant at the $P \leq 0.05$ confidence level; absence of asterisk indicates significance at the $P \leq 0.20$ level. NS indicates the result was not significant.

Variable	Forest to grass	Forest to unvegetated	Grass to forest	Grass to unvegetated	Unvegetated to forest	Unvegetated to grass
1975-1986						
Elevation	−(−)	NS	NS	−(+)*	NS	+(+)*
Slope	−(−)*	−(−)*	+(+)	−(−)*	+(+)*	+(−)*
Distance to road	NS	NS	NS	NS	NS	NS
Distance to market	−(−)	NS	NS	−(−)*	NS	−(−)*
Population	NS	+(+)	NS	NS	NS	NS
1986-1991						
Elevation	NS	NS	NS	−(−)*	NS	NS
Slope	NS	NS	NS	−(−)*	NS	NS
Distance to road	NS	NS	NS	NS	NS	NS
Distance to market	NS	NS	NS	NS	NS	NS
Population	NS	NS	NS	+(+)	NS	−(−)*

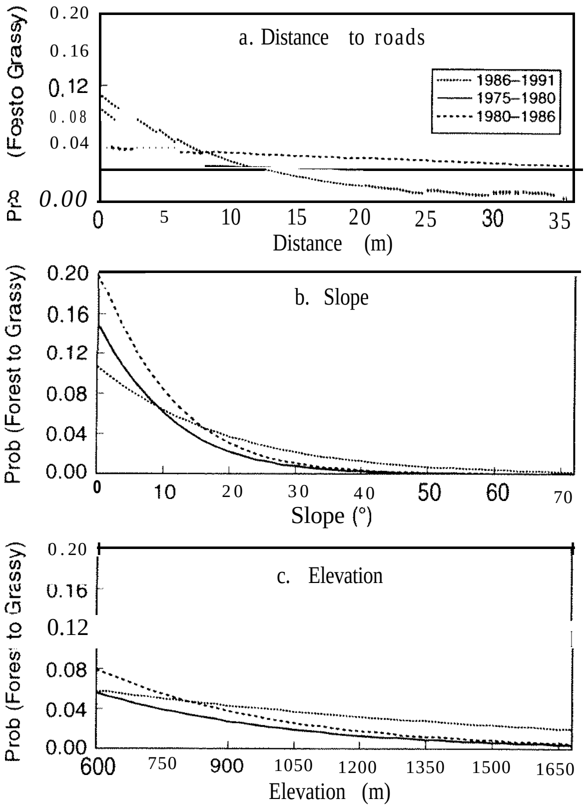

FIG. 4. Comparison of the probability of transition from forest cover to grassy cover in the Little Tennessee River Basin, as related to independent variables between time periods.

forest-to-grassy transition. However, these relationships changed substantially between periods. Probability of transition was concentrated much closer to roads for the period 1986-1991 (Fig. 4a). In contrast, forest-to-grassy transitions were less influenced by slope and elevation in this later period (Figs. 4b and c). Accordingly, the probability of disturbance was greater at higher elevations and on steeper slopes in 1986–1991 than in the previous two periods.

Effects of spatial variables on transition models were substantially different for the USFS lands. In contrast to the private lands, no spatial variable produced a significant marginal effect on any transition probability, even at the $P \leq 0.20$ level. Furthermore, only eight of 30 coefficients were significant in constructing the cumulative distribution function for transitions.

Land ownership and landscape chunge: Hoh River Basin

Temporal chunge in transition models-Tests for temporal change in transition models indicate rejection of stable models on private lands in the HORB (Table 6b). With the exception of grassy cover between the periods 1975-l 980 and 1980-l 986, all transition models for private lands shifted significantly through time. On DNR lands, transitions for the coniferous forest type (which applies to the largest share of the landscape) were constant between 1980-1986 and 1986-1991. However, all other temporal comparisons of models indicate dynamic transition relationships for DNR lands.

Effects of ownership on transition models.—Differences between private and DNR transition models were tested for all periods and for all initial cover types in the Hoh Basin (i.e., no data were pooled between periods). For coniferous forest, there were significant differences between owners in the transition models for all three periods (Table 7b). Identical transition models for all other cover types in 1975-1980 were also rejected. For 1980–1986, we could not distinguish between private and DNR transition models on mixed forest and grassy cover types. For 1986-l 99 1, we could not distinguish between private and DNR transition models for grassy and unvegetated cover types (Table 7b).

Effects of spatial variables on transition models.-In contrast to the LTRB, individual spatial variables provided little explanation of private cover transitions in the HORB. Only one of the marginal effects coefficients (elevation, Table 9a) was significant at the $P \leq 0.20$ level for private lands for 1975-I 980, showing the expected positive relationship. Only four of 24 marginal effects coefficients were significant at the $P \leq 0.05$ level for 1986–1991, primarily within the grass-to-forest and grass-to-unvegetated transitions. Population was not included because the reported values did not vary across this sparsely populated drainage.

Effects of spatial variables on the DNR lands in the HORB were quite different from those on the private lands (Table 9b). For the period 1975–1980, three of 24 marginal effects coefficients were significant at the $P \leq 0.05$ level (Table 9b); 10 of 24 were significant at the $P \leq 0.20$ level. Five of 24 were significant for 1986-1991 (Table 9b) (eight at $P \leq 0.20$), but several of these significant factors had signs opposite to our expectations. The strongest set of spatial relationships was found for transition from coniferous forest to grassy cover in 1975-1980, where all four of the spatial variables had significant marginal effects. The signs for the effects of elevation and distance to roads on harvest probability were consistent with our expectations (i.e., that increasing costs are inversely related to harvest probability). However, the signs of the marginal effects for slope and distance to market were not consistent with our expectations, indicating positive relationships with harvest probability.

Land ownership and landscape change: Dungeness River Basin

Temporal change in transition models.-In contrast to findings for the HORB, transition models for the DURB were relatively stable. Transition models for the conifer land-cover type did not differ significantly on

TABLE 9. Tests of hypotheses regarding marginal effects of independent variables on specific land-cover transitions for the Hoh River Basin, by ownership class. For each entry, the first sign represents the expected sign of the marginal effect on the referenced transition. The sign in parentheses is the marginal effect calculated at the means of the independent variables. Significance levels indicated are as in Table 8.

Variable	Forest to grass	Forest to unvegetated	Grass to forest	Grass to unvegetated	Unvegetated to forest	Unvegetated to grass
a) Private lands						
1975-1980						
Elevation	NS	NS	NS	NS	+(+)	NS
Slope	NS	NS	NS	NS	NS	NS
Distance to road	NS	NS	NS	NS	NS	NS
Distance to market	NS	NS	NS	NS	NS	NS
1986-1991						
Elevation	NS	NS	+(+)*	NS	NS	NS
Slope	NS	NS	+(−)*	−(+)	NS	NS
Distance to road	NS	NS	+(+)	-(-)*	NS	NS
Distance to market	NS	NS	NS	-(+)*	NS	NS
b) Department of Natural Resources						
1975-1980						
Elevation	−(−)	NS	NS	−(−)	NS	NS
Slope	−(+)*	NS	NS	NS	+(+)	NS
Distance to road	−(−)	NS	NS	−(−)	NS	NS
Distance to market	−(+)	NS	+(+)*	−(+)*	NS	−(+)
1986-1991						
Elevation	NS	−(−)	+(+)*	NS	NS	NS
Slope	NS	NS	NS	NS	NS	NS
Distance to road	NS	−(+)*	+(+)*	−(−)	+(+)	NS
Distance to market	NS	NS	+(−)*	−(+)*	NS	NS

either ownership type between any periods (Table 6c). On private lands, some shifts in transition relationships for other cover types were observed, suggesting changes in lands dedicated to agriculture and other developed uses. On USFS lands, however, the transition models were generally stable, with only unvegetated cover showing significant change between 1975–1980 and 1980-I 986.

Effects of ownership on transition models.—Although transition relationships were more stable through time in the DURB than in the HORB, differences between ownership types were pronounced (Table 7c). With the exception of coniferous forests in 1975-1980, all transition models for public and private lands in all periods were significantly different. As in the HORB and LTRB, there were structural dissimilarities in land-cover dynamics between public and private lands.

Effects of spatial variables on transition models.-On private lands in the DURB, the significant temporal changes in transition models were reflected in the effects of the spatial variables. For the period 1975– 1980 (Table 10a), no spatial variables influenced transitions from coniferous forest cover. However, for the period 1986-l 99 1, nine of 10 marginal effects coefficients for these transitions were significant. Population density had the strongest influence in the transition models on private lands. For the period 19751980, three of the six transition models displayed in Table 10a were significantly influenced by population density at the $P \leq 0.20$ level, and two were significant at the $P \leq 0.05$ level. For the period 1986-1991, all six models indicated significant effects of population on transition probabilities.

Population density also had a significant influence on USFS transitions (Table 10b), indicating that, for example, timber harvesting was less likely where, ceteris paribus, population density was higher. In general, spatial factors had much more influence on USFS transition probabilities in the DURB than in the LTRB. While no spatial variable yielded a significant marginal effect on USFS lands in the LTRB, 11 of the 40 marginal effects (Table 10b) were significant at the $P \leq 0.05$ level, and 22 were significant at the $P \leq 0.20$ level. There were differences in the marginal effects on transitions from forest cover between periods, especially in the effects of slope: both were insignificant for 1975–1980 but negative for 1986-1991.

DISCUSSION

Land ownership and landscape pattern

Land ownership clearly influenced landscape pattern, despite differences between the two study regions. Private lands contained less forest cover but a greater number of small forest patches than did public lands, indicating greater forest fragmentation. Lands that were actively managed for timber harvest, however, showed

TABLE 10. Tests of hypotheses regarding marginal effects of independent variables on specific land-cover transitions for the Dungeness River Basin, by ownership class. For each entry, the first sign represents the expected sign of the marginal effect on the referenced transition. The sign in parentheses is the marginal effect calculated at the means of the independent variables. Significance levels indicated are as in Table 8.

Variable	Forest to grass	Forest to unvegetated	Grass to forest	Grass to unvegetated	Unvegetated to forest	Unvegetated to grass
a) Private lands						
1975–1980						
Elevation	NS	NS	+(+)	NS	+(+)*	NS
Slope	NS	NS	+(−)*	NS	+(+)	NS
Distance to road	NS	NS	+(−)	NS	+(+)	NS
Distance to market	NS	NS	+(−)*	NS	NS	NS
Population	NS	NS	−(−)*	NS	−(−)*	−(−)
1986–1991						
Elevation	−(+)*	NS	+(−)*	NS	NS	NS
Slope	−(−)*	−(+)*	+(+)*	−(−)*	NS	NS
Distance to road	−(−)*	−(−)*	+(−)	NS	NS	NS
Distance to market	−(−)*	−(−)*	+(+)*	−(−)*	NS	NS
Population	+(+)*	+(+)*	−(−)*	+(+)*	−(−)	−(−)
b) U.S. Forest Service						
1975–1980						
Elevation	−(−)	−(+)			+(+)	+(−)
Slope	NS	NS			NS	NS
Distance to road	−(+)	NS			+(−)*	NS
Distance to market	NS	NS			NS	+(+)
Population	NS	NS			−(−)*	+(+)*
1986–1991						
Elevation	NS	−(−)	NS	NS		
Slope	−(−)	−(−)*	+(+)	−(−)		
Distance to road	−(−)*	−(+)	NS	NS		
Distance to market	NS	−(−)*	+(−)*	−(+)*		
Population	NS	−(−)*	−(−)*	+(+)*		

little difference in landscape pattern between ownerships (e.g., private and DNR lands in the HORB). Differences in landscape structure between ownerships reflect the management and land-use decisions of the owners. When owners share similar objectives (e.g., commercial forests and state DNR both emphasize timber production), then landscape patterns may be similar.

Differences in landscape pattern observed between the two regions probably reflect differences in the importance of forest harvesting. Forest cover was greatest in the LTRB on both ownership classes as compared to any land-ownership category in the HORB and DURB. Although the LTRB was almost completely cut over during the early decades of the 20th century, timber extraction is no longer dominant. Private lands in the LTRB were characterized by only 10% less forest cover than USFS lands, and connectivity of the forest tbroughout the watershed was high. Small forest patches occurred primarily near the town of Franklin, and not in the higher, less accessible areas.

Greater variability in landscape structure through time and between ownership categories was observed on the Olympic Peninsula. The proportion of the ownership in coniferous forest followed a consistent rank order: USFS > DNR > HORB private > DURB private. In contrast, the proportion of the landscape in unvegetated cover was greatest in DURB private lands and lowest in USFS lands; private commercial and DNR lands had comparable intermediate levels of unvegetated cover. These trends reflect land use within the watersheds. The DNR and private commercial lands in the HORB are both managed for timber production; spatial patterns differed between these ownerships primarily due to the relative abundance of coniferous (higher on DNR lands) vs. deciduous/mixed forest cover (higher on private commercial lands). Patterns on the small private ownerships that occurred in the DURB reflected rapid development in and around the town of Sequim, resulting in substantial increases in unvegetated cover. In contrast, USFS lands in the DURB showed relatively small changes through time, although changes were greater than those observed in the LTRB.

Land ownership und landscape change

Temporal change in transition models.-The analyses of landscape change revealed differences in land-cover change through time. In the LTRB, the rate of forest transition decreased between the 1975-1986 period and the 1986-1991 period for both the USFS and private ownership classes. In the HORB, transition models were also dynamic in time for both ownership types. These results suggest the need to account for additional factors that cause individuals or institutions to change land management strategies. For example, shifts in the LTRB models may reflect a transition from land use focused on forest management to one focused on expanding residential development, along with an emphasis on public lands of in situ value (e.g., water quality maintenance and biodiversity). In the HORB, where active timber management is the dominant use, land-cover transitions may be strongly influenced by timber prices.

Temporal stability in the transition relationships for coniferous forest on USFS lands in the DURB was somewhat surprising, given the dynamics of wood products markets during this period. Between 1975 and 1980, timber markets were especially strong, with timber prices for West Coast species increasing at unprecedented rates (e.g., Mattey 1990). In contrast, stumpage prices dropped substantially in the early 1980s and then recovered beginning in 1986. For example, the average price of Douglas-fir sawtimber sold from the national forests in Washington and Oregon peaked in 1980 at $432/mbf (thousand board feet), and had fallen to $11 8/mbf by 1982. Prices remained low through the early 1980s but began to climb in 1986. By 1991, Douglas-fir prices were $395/mbf (Warren 1992). Our focus on the details of a smaller area for three periods allows us to examine spatially variable factors, but does not support direct analysis of the effects of prices and other temporally variable factors. A more frequent, perhaps annual, sampling over a larger area could allow a direct analysis of price effects on harvest behavior.

Harvesting behavior and landscape dynamics on private forest lands in the Dungeness apparently were not substantially influenced by these strong market forces, and remained relatively stable. In contrast, forest-cover dynamics changed substantially between these periods on the large commercial private lands in the HORB. In this situation, land-cover dynamics on commercial private lands were more volatile than dynamics on small private holdings. Transition models were also stable between periods on USFS lands in the Dungeness. However, the scale of our analysis may not be sufficiently large to accurately address the USFS response to markets. That is, timber harvesting may shift among drainages over time, so that the better unit of observation may be an entire national forest.

Effects of ownership on transition models.—Differences between ownerships in the models of forest-cover change were observed in all three watersheds. In the LTRB, little commercial forestry is practiced on private lands, but residential development has increased during the study period. The HORB is dominated by forestry uses, and significant differences in transition models for coniferous forest cover probably reflect differences in the forestry practiced by the Washington DNR and the commercial private owners. The similarity of transition models for grassy and unvegetated cover types in the HORB may reflect similar patterns of forest stand regeneration and regrowth. In the DURB, coniferous forest transition models did not differ between the USFS and private lands in the 1975–1980 period, but all other models differed between public and private lands. As in the LTRB, this difference probably reflects different influences on forestry practices, or multiple-use management on the USFS lands and increased residential development on the private lands.

Effects of spatial variables on transition models.-The importance of independent variables in explaining land-cover change generally varied between ownerships in each watershed. In the LTRB, spatial variables did influence land-cover change on private lands, although effects were stronger during the 1975-1986 period than in the 1986-1991 period. Most of the marginal effects were in the hypothesized directions (Table 2). However, the positive relationship between elevation and grassy-to-unvegetated transitions on private lands in the LTRB was counter to the hypothesized relationship. Increasing development at higher elevations is inconsistent with the hypothesis derived from an argument based on cost. This finding may reflect preferences for scenic views, which would encourage residential development at higher elevations, consistent with anecdotal observations on recent developments in the Southern Appalachians. Overall, land-cover changes on private lands in the LTRB were consistent with a shift from land use focused on forest management to land use focused on expanding residential development. Indeed, population has grown steadily in the LTRB, and timber production has declined. In sum, a structural shift in the pattern of disturbance was indicated on private lands, and forest disturbance on private lands was more strongly influenced by location relative to the road network than by other site factors, such as elevation and slope.

On USFS lands in the LTRB, no spatial variable had a significant marginal effect on any transition probability, and the average probability of change applied across the USFS lands was the best predictor of change. Apparently, rules that are not correlated with the spatial variables defined here have guided the management of USFS lands during the study periods, although private owners were strongly influenced by cost factors associated with development, timber harvest, or transportation. Results for USFS lands may be consistent with multiple-use management that mitigates the negative effects of timber sales on wildlife habitat and scenic views by spreading harvest activities over broad areas.

In the HORB, individual spatial variables provided little explanation of land-cover transitions on private lands, but did explain cover transitions on DNR lands. The lack of effects on private lands in the HORB suggests that factors other than those represented by the spatial variables measured here explain the probability of harvesting timber. It may be that, in an area such as the Olympic Peninsula, where both timber volume per acre (per 0.405 ha) and timber values are very high, variable costs of logging and transport have relatively little impact on harvesting decisions. That is, the high value of timber may make the spatially variable costs of timber extraction unimportant for timbering decisions. Rather, the harvesting plan may be more sensitive to temporal change in the relative prices of wood products, optimal depletion schedules, or forestry policies.

Although the DNR lands are also managed for timber production in the HORB, the sign of the effects of slope and distance on harvest probability was not consistent with our expectations. The difference in the relationship between slope and distance to market for coniferous forest transitions on DNR lands in the HORB might be consistent with the aggregation of harvest units. That is, as the basin has been developed for timber production, initial harvests may have been conducted on level and accessible sites. Accordingly, large openings in these areas may constrain further harvest, leading to a subsequent bias towards more remote and steeper sites. The availability of harvestable timber may constrain timbering choices, with the less accessible lands being harvested while the more accessible lands regenerate from past harvest, but this issue clearly needs additional investigation.

The findings for these large ownerships in the HORB again raise the issue of the appropriate scale of analysis for the types of questions addressed here. Unlike small private landowners, whose holdings are focused within a small area, large firms or public institutions may respond to regional, national, or international factors. For example, a timber corporation may alter harvest plans in response to capital requirements for milling facilities. A large government agency may focus on even-How harvesting from a broad region. This suggests that local conditions may hold less influence over the land-use choices of larger owners.

In many ways, the social settings of the DURB and LTRB are very similar. Both areas have experienced population growth and expansion in residential development in recent years. However, public and private lands are much less intermingled in the DURB, and private lands tend to be concentrated in areas that are less steep and less remote than public lands. The differences between ownerships observed in the LTRB were not found in the DURB. No significant relationships between spatial variables and land-cover change on USFS lands were observed in the LTRB, but several spatial variables had a significant influence on forest-cover transitions on USFS lands in the DURB.

For private lands in the DURB, no spatial variable influenced transitions between 1975 and 1980, but many had significant effects during the period 1986-1991. One possible explanation relates to differences in timber markets. That is, when prices are temporarily high, marginal cost factors become less critical as forest owners attempt to capture ephemeral revenues. Another potential explanation is that the basin has become more influenced by population growth pressures and less by timber harvesting on private lands. This is supported by the significant effects of population on nine of the 12 transition relationships displayed in Table 10.

Conclusions

The analysis presented here demonstrated that different broad ownership groups produce qualitatively distinct landscape patterns. Furthermore, it demonstrated that different types of owners interact with similar lands in distinct ways (Table 7). The way in which human endeavors are organized through the institutions and scale of land ownership significantly influences the dynamics of land cover. Land ownership produces distinct signatures on landscapes, creating patterns that, in turn, will influence a variety of ecological processes. Thus, understanding and predicting land cover requires knowledge about land ownership. Purely biophysical models will provide limited insight into land-cover dynamics, as some explanatory variables are likely to be socioeconomic and political (Lee et al. 1992, Machlis 1992). There remains a tremendous need for work that integrates ecological and socioeconomic dynamics at landscape scales.

We know of only one other study in which landscape pattern and rates of change in forest cover were evaluated as a function of land ownership: Spies et al. (1994) examined an area including part of the Willamette National Forest, Oregon, from 1972 to 1988, similar to the 1975-1991 period of our study. Ownership patterns in the Willamette study area were most similar to those in the HORB in this study. Public land-ownership classes occupied -70% of the study area and included USFS, Bureau of Land Management, and the State of Oregon. Private lands consisted primarily of industrial land ownerships. As in our three watersheds, Spies et al. (1994) also reported that a greater proportion of forest cover in the Willamette study area occurred from 1984 to 1988 on public lands and from 1981 to 1984 on private lands. In the HORB, public and private lands both experienced the most rapid loss of coniferous forest cover between 1980 and 1986 (Table 4), similar to the Willamette. In contrast to the Willamette study area, however, coniferous forest cover on the public lands in HORB increased subsequently. Greater amounts of edge on private vs. public lands were reported for both the Willamette and HORB study areas. Although we did not compute interior forest habitat, the 8 to 16-fold difference in weighted-average patch size and lower edge-to-area ratio for coniferous forest on public vs. private lands in the HORB (Table 4) is consistent with a greater abundance of interior habitat on public lands, as reported by Spies et al. (1994). The amount of interior coniferous forest in the HORB (based on weighted-average patch size and edge-to-area ratio) probably reached a low in 1986 on both public and private lands, and then increased between 1986 and 1991.

Patterns of change in coniferous forest were similar at low and high elevations in the Willamette study area (Spies et al. 1994), consistent with the lack of a significant effect of elevation for private lands in the HORB during any time period. However, on DNR lands in the HORB, we observed a negative relationship between elevation and forest transition to grassy cover during the 1975-1980 period, and to unvegetated cover during the 1980-1986 period, suggesting that forest-cutting rates were higher at lower elevations. In sum, forest-cover patterns on public and private lands in the Willamette study area and HORB were generally similar, but the Willamette did not show the increasing trend in coniferous forest cover observed during the lattermost time period in the HORB.

Spatially referenced physical (slope, elevation) and cultural (distance to roads and markets, as well as population density) features had measurable influences over the probability of land-cover change on public and private lands. Transition models for all river basins, ownerships, and cover types provided significant explanation of observed transitions. Thus, spatially referenced data, in comparison to simple averages, can improve the estimation of transition probabilities. Spatially explicit estimates of land-cover change can indicate to land managers what portions of the landscape may be most subject to rapid change. For example, the statistical models developed in this study can be extrapolated spatially across the landscape to map the probability of any land-cover change as a function of the attributes of each grid cell (e.g., Wear and Flamm 1993). Such maps can provide a graphical summary of where either desirable or undesirable land-cover changes are most likely to occur.

Transition probabilities generally were not stable through time, suggesting that simple Markovian models of land-cover change are not likely to represent future landscape conditions in anthropogenic landscapes. Rather, transition probabilities are likely to vary through time, as the responses of individuals and institutions to social and economic conditions change. Shifts in conditions (e.g., timber prices), trends in recreational preferences in the population, and variable rates of residential development may all lead, individually or in concert, to substantial shifts in rates of land-cover transition.

The length of time over which the imprint of land-ownership patterns will remain on the landscape is not known. Wallin et al. (1994) demonstrated that landscape patterns created by dispersed disturbances are difficult to erase, and time lags may be considerable, even with substantial reductions in disturbance rates. Thus, the imprint of current land-ownership patterns on the landscape is likely to persist for some time, even if land ownership changes. As noted by Spies et al. (1994), existing conditions are important for the design of future landscape patterns geared toward maintenance of particular species or ecosystem functions. Existing patterns will constrain future conditions for some time, and managers will continue to face the challenge of integrating present patterns with desired future conditions.

The results obtained in this study were affected by the spatial scale of the data and the land-cover categories selected for analysis. Direct comparisons of the numerical results (e.g., landscape metrics or estimated coefficients) with other studies must be done with care. Measures of landscape pattern are strongly influenced by both the grain (e.g., spatial resolution, or grid-cell size) and extent (total area considered) of the data (e.g., Allen and Starr 1982, Turner et al. 1989). In addition, selection of the categories used in the analysis constrains the results. For example, stand age was not included in this study as a modifier of forest cover; therefore, the results reported here do not distinguish between old- and secondary-growth forest cover. In addition, the use of land-cover classes based on canopy characteristics cannot be used to infer land use in the absence of additional data sources. For example, low-density residential development that does not result in canopy breakup is not likely to be detected.

Extrapolation of these analyses to other locations can be considered in two ways. First, it is of interest to determine whether or not the qualitative differences between ownership classes observed in this study are applicable to other river basins within the same regions (i.e., southern Appalachian highlands and Olympic Peninsula), or perhaps even to other river basins in forested landscapes. The results for the LTRB, HORB, and DURB should be compared with other river basins to search for generalities that might be broadly applicable. Second, the methodology demonstrated here could be applied in other systems. Data availability is often the primary limiting factor, but this constraint is diminishing rapidly with the widespread development of GIS databases for many regions.

The strong influence of land ownership on both landscape pattern and land-cover change has important implications for the future landscape mosaic. Ownership class must be considered when potential changes within a river basin or landscape are predicted. If transition probabilities are to be used to simulate future conditions (e.g., Flamm and Turner 1994*a*, b), separate models should be developed for different ownership classes. The transition models developed in this study may prove especially useful in a simulation framework that can forecast the effects of various ownership scenarios. In such an exercise, the potential implications of, e.g., changes in USFS policy, can be examined at a whole-landscape scale. For example, the models described here for the LTRB were applied in a factorial simulation experiment to project both the effects of extrapolating observed rates of change into the future, and of imposing some constraints, or rule changes, on future transitions (Wear et al. 1996).

Land ownership may have strong effects on numerous ecological processes because the influence of ownership on landscape pattern is so strong. Natural disturbance dynamics (e.g., Turner 1987, Roland 1993), stability of the vegetation mosaic (e.g., Turner et al. 1993), persistence of species (e.g., Lamberson et al. 1992, Pulliam et al. 1992, Turner et al. 1994), and quality of water resources (e.g., Peterjohn and Correll 1984, Kesner and Meentemeyer 1989) can all be affected by landscape structure. Indeed, this study could be expanded to identify linkages with additional selected ecological end points, such as species persistence or water quality. Our results suggest that land-ownership effects must be considered explicitly when the future vitality of ecological systems is considered. If paradigms such as sustainability and ecosystem management are truly to inform land management, efforts to seek generality and to understand and predict the dynamics of mixed-ownership landscapes must continue.

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