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Landscape Trends in Mid-Atlantic and Southeastern United States Ecoregions

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ABSTRACT / Landscape pattern and composition metrics are potential indicators for broad-scale monitoring of change and for relating change to human and ecological processes. We used a probability sample of 20-km × 20-km sampling blocks to characterize landscape composition and pattern in five US ecoregions: the Middle Atlantic Coastal Plain, Southeastern Plains, Northern Piedmont, Piedmont, and Blue Ridge Mountains. Land use/land cover (LULC) data for five dates between 1972 and 2000 were obtained for each sample block. Analyses focused on quantifying trends in selected landscape pattern metrics by ecoregion and comparing trends in land cover proportions and pattern metrics among ecoregions. Repeated measures analysis of the landscape pattern documented a statistically significant trend in all five ecoregions towards a more fine-grained landscape from the early 1970s through 2000. The ecologically important forest cover class also became more fine-grained with time (i.e., more numerous and smaller forest patches). Trends in LULC, forest edge, and forest percent like adjacencies differed among ecoregions. These results suggest that ecoregions provide a geographically coherent way to regionalize the story of national land use and land cover change in the United States. This study provides new information on LULC change in the southeast United States. Previous studies of the region from the 1930s to the 1980s showed a decrease in landscape fragmentation and an increase in percent forest, while this study showed an increase in forest fragmentation and a loss of forest cover.

The physical landscape of the United States is constantly changing and land use/land cover (LULC) is perhaps the most ubiquitous of all environmental change. Therefore, quantifying the degree and types of change occurring at the landscape scale is necessary to better understand its potential impact on ecological condition, such as on water quality, or on fluctuations of wildlife populations. At local and regional scales, LULC change can have profound impacts on aquatic systems due to land use practices that adversely affect water quality and sedimentation. Moreover, local LULC changes are fundamental agents of global climate change and are significant forces impacting biodiversity, water and radiation budgets, trace gas emissions, carbon cycling and, ultimately, climate at all scales (Donoghue 2002, Riebsame and others 1994).

KEY WORDS: Landscape monitoring; Ecoregions; Trends; Indicators; Fragmentation; Mid-Atlantic United States; Southeastern United States

Although scientists recognize the importance of LULC change, little comprehensive information exists on the geographic distribution and rates of LULC change throughout the United States. Even less information exists on trends in landscape pattern at this spatial scale. To address this knowledge gap, the US Geological Survey (USGS), US Environmental Protection Agency (EPA), and the National Aeronautics and Space Administration (NASA) have initiated research on strategies for documenting the spatial characteristics of LULC change. Landsat multispectral scanner (MSS) and thematic mapper (TM) satellite data are used to document the types, geographic distribution, and rates of change in the conterminous United States from the early 1970s to 2000 (Loveland and others 2002). The component of the research focusing on landscape pattern changes is the topic of this article.

Metrics characterizing aspects of landscape pattern are needed to test hypotheses and to correlate landscape pattern with important environmental attributes or processes such as water quality, avian population dynamics, large mammal movements, or nutrient flows

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(O'Neill and others 1997). Landscape pattern metrics (LPMs), such as those described in McGarigal (1999) and in Gustafson's (1998) review, also create a common language to characterize landscapes because they quantify within-site variability, compare among sites, and describe changes over time (Imbernon and Branthomme 2001). Therefore, in addition to simple statistics on change in LULC proportions, LPMs should also be considered in monitoring programs (Herzog and Lausch 2001, Petit and Lambin 2001).

The literature is rich with examples stating the need for long-term monitoring of the landscape and its spatial pattern, because LULC is oftentimes a stressor to ecosystems. Understanding the effects of LULC on the continued provision of ecosystem services to society is thus crucial and explains why landscape approaches and characterizations are important parts of federal assessments (Reed and others 1996, Bourgeron and others 1999, Dumanski and Pieri 2000, McDonald 2000, Jones and others 2001). This also helps explain why landscape monitoring is moving up the political agenda (McDonald 2000, Kammerbauer and others 2001). A long-term vision is necessary so that we can begin to evaluate trends in natural resources. Although landscape metrics are still an issue of science and policy debate, there are few other options currently available for monitoring and reporting over large areas (Fjellstad and others 2001, Dramstad and others 2002). Landscape metrics are becoming a standard assessment protocol for agroecosystems (Fjellstad and others 2001, Dramstad and others 2002), semiarid grasslands (Holm and others 2002), and tropical forests (Lambin 1999).

While landscape patterns in relatively small geographic areas have been modeled, analyses of landscape pattern changes over areas the size of ecoregions or states have been less frequent. However, developing indicators of ecosystem health at regional scales has become a priority (Griffith 1998, Bourgeron and others 1999), and landscape measures will undoubtedly be one focus of such endeavors. Riitters and others (1995) called for more research using real landscapes over real time changes. This is particularly appropriate in the mid-Atlantic and southeastern United States, as a number of socioeconomic driving forces shape the landscape here (Galehouse 1981, Bielski 1992, Greene and Benhart 1992, Hartshorne 1997, Walcott 1999). Moreover, regional planners and resource managers have been slow to deal with landscape dynamics because they are inadequately equipped to analyze both rapid change and gradual evolution (Marcucci 2000).

The primary objectives of this research are to quantify trends in landscape pattern from the early 1970s to 2000 for five ecoregions in the United States and to determine if these trends in landscape pattern differ among the five ecoregions. If differences among ecoregions are found, such a result would suggest ecoregions provide suitable, although not necessarily the only, spatial context for regional analysis and reporting of trends in LULC and LPMs.

Study Area

The five ecoregions were chosen from the set delineated by the US EPA (Omernik 1987, 1995) and include the Middle Atlantic Coastal Plain, Southeastern Plains, Piedmont, Northern Piedmont, and Blue Ridge Mountains (Figure 1). The digital boundaries of the ecoregions were downloaded from ftp://ftp.epa.gov/wed/ecoregions/us in March 2001. These ecoregions form a gradient of geographic and landscape mosaic types from the Atlantic Coast to the highest elevations in the eastern United States (Table 1). Furthermore, portions of the eastern United States are dynamic, and different socioeconomic factors drive landscape change among ecoregions. The primary characteristic of these ecoregions is the dominance of forest, whether it be eastern deciduous, southern pine, or mixed deciduous/coniferous forest. The Northern Piedmont contains significant areas of both built-up (urban) and agricultural land, but forest cover had the highest proportion of coverage by the mid-1980s to the present (Loveland and others 2002).

Methods

Essential details of the sampling design and analysis protocol are summarized here, with full description of the methodology reported in Loveland and others (2002). The national geographic framework for the research is organized by ecoregions, which are areas of relatively homogeneous environmental attributes, such as soils, geology, natural vegetation, land use, and physiography (Omernik 1987, 1995). Although other regionalization schemes exist, this particular framework incorporates human activity (in the form of dominant land use) as a factor in determining boundaries. This is useful in a geographic study such as ours, in which we desire to also understand some of the socioeconomic drivers of LULC change. We have found that several socioeconomic variables (e.g., dominant industry, clusters of small- to mid-sized cities) correlate well with ecoregion boundaries.

To estimate landscape pattern and LULC proportions, we first partitioned the ecoregion into nonoverlapping, 20-km × 20-km blocks. The choice of the 20-km × 20-km block size entailed compromising be-

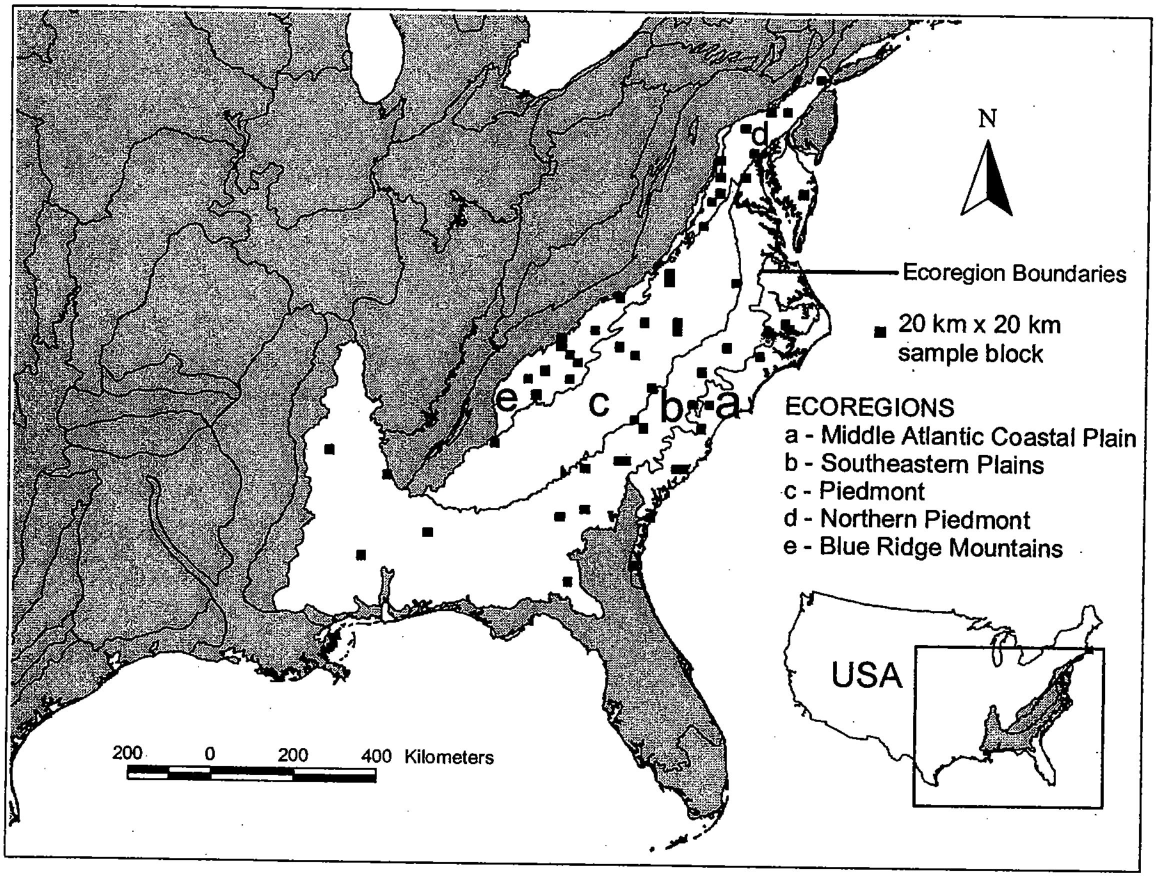

Figure 1. The five ecoregions comprising the study area in the mid-Atlantic and southeastern United States. Instances where the sample blocks appear wider or taller are those in which the blocks are adjacent to each other.

Table 1. Physical geography of five study ecoregions[a]

Ecoregion	Land-surface form	Potential natural vegetation	Land use	Soils
Middle Atlantic Coastal Plain	Flat plains	Oak/hickory/pine, pocosin, southern floodplain forest, southern mixed forest	woodland and forest, with some cropland, pasture and swamp	Aquults
Southeastern Plains	Irregular plains	Oak/hickory/pine, Southern Mixed forest	mosaic of cropland, pasture, woodland, and forest	Tertiary sands, silts, and clays
Northern Piedmont	Irregular plains, plains with low to high hills	Appalachian oak	cropland with pasture, woodland and forest, urban	Mesic Udalfs and Udults
Piedmont	Smooth to irregular plains	oak/hickory/pine, southern mixed forest	mosaic of cropland, pasture, woodland/forest, urban	Ultisols derived from igneous and metamorphic rocks
Blue Ridge Mountains	Low mountains, open, low mountains	Appalachian oak	forest and woodland mostly ungrazed, woodland and forest with some cropland and pasture	Hapludults, Dystrochrepts

[a] From Omernik (1987) except for the Piedmont, which is from metadata downloaded from ftp://ftp.epa.gov/wed/ecoregions/us in March 2001.

tween two competing objectives: estimating change in LULC, which favors a small block size for precision (Harrison and Dunn 1993, Stehman and others 2003) and evaluating change and trend in land-scape pattern, which favors a larger block size (O'Neill and others 1996). To maintain equal-area sampling units and to avoid introducing differences in the landscape pattern metrics attributable to area differences, we overlaid each ecoregion with full-size blocks (i.e., we did not create blocks smaller than 20-km × 20-km at ecoregion boundaries to fit blocks inside the ecoregion). Consequently, the boundary of the region (population) covered by the blocks eligible for sampling deviates slightly from the ecoregion boundary. In each ecoregion, a simple random sample (without replacement) of 20-km × 20-km blocks was selected. Ten or eleven sample blocks (depending on the size of the ecoregion) were selected to satisfy an *a priori* planning criterion of estimating overall gross change in LULC for each ecoregion with a margin of error of ± 1% based on an 85% confidence interval (Loveland and others 2002, p. 1093). Once the sample blocks were selected, we then assembled our image database.

The National Land Cover Data (NLCD) from 1992 (Vogelmann and others 2001) served as the baseline for deriving land cover change information. The 21 LULC classes in the NLCD were collapsed to 11 classes (a modified Anderson level I classification) deemed important for examining patterns of LULC change. The overall accuracy for the original 30-m NLCD data in the eastern United States was 80.5% for the Anderson level I classification level (Yang and others 2001). The chosen classification scheme was also based on amenability to accurately identify these classes from the available imagery and ancillary interpretation resources. The 11 classes selected were: urban or built-up, mining, agriculture, forests and woodlands, shrubland/grassland, wetlands, water, snow/ice, natural barren, mechanical-disturbed (primarily timber harvesting in the study region), and nonmechanical disturbed (e.g., from storms, fire, or pest infestations). The imagery used as reference data to detect LULC change consisted of five dates of MSS and TM data geocoded to a common Albers equal area projection. Some of the imagery was obtained from the North American Landscape Characterization (NALC) Program (Sohl and Dwyer 1998). In the NALC program, all MSS images (from the 1970s, 1980s, and 1990s) were resampled to a 60-m cell size. In addition, 30-m TM images were obtained for the 1990s and 2000 for aiding the identification of LULC changes. The imagery dates were generally within ± 2 years of 1973, 1980, 1986, 1992, and 2000. The dates (with the exception of 1980) were chosen based on available, low-cost, cloud-free imagery from the NALC program.

The first step in the LULC interpretation protocol was intensive manual re-interpretation of the NLCD data within each sample block. Because the NLCD is intended for regional to national scale studies, it does not necessarily achieve the accuracy desired at the sample block level required for our analyses of change and trend. Consequently, the "editing" of the NLCD was implemented to improve upon the accuracy of the baseline LULC data. The manual editing of the NLCD data employed onscreen interpretation methods with the 1992 Landsat TM data and aerial photography serving as additional materials to aid interpretation. This intensive reinterpretation effort likely significantly improves upon the accuracy of the NLCD in our sample blocks. Loveland and others (2002, pp. 1094–1095) discuss the in-progress validation of the interpreted LULC change. A major problem associated with assessing the accuracy of the interpreted change is identifying adequate historical materials that would produce the higher quality reference change data required for such an assessment.

The next step in the protocol was to perform manual interpretation of the remaining sets of Landsat MSS and/or TM imagery, and to visually identify and code areas that underwent LULC change in each sample block. LULC for the 1972, 1980, 1986, and 2000 periods were then "back or forward classified" using the 1992 LULC data as the template. The analyst searched through, for example, the edited 1992 LULC map, examining the 1986 and 1992 Landsat images and aerial photographs for any valid LULC changes that occurred between 1986 and 1992. Any identified change in land cover was manually digitized on-screen, and the resultant edits were incorporated to produce a 1986 map. After the 1986 LULC interpretation was completed the same procedures were used to create the 1980, 1972, and 2000 LULC maps. The minimum spatial resolution of the LULC change database is 60-m × 60-m, and all final LULC maps for each time period had this spatial resolution.

Upon completion of the interpretation process, a suite of standard landscape pattern metrics (LPMs) describing the number, size, shape, and spatial relationship of patches of land cover types was calculated for each sample block in each time period. We used FRAGSTATS 3.01.02 to derive the LPMs (McGarigal 1999). Landscape patches were defined using the eight adjacent and diagonal cells, and a landscape border was used in the calculation of metrics. Seven metrics were chosen: number of patches, number of forest patches, forest edge, forest area-weighted mean patch size, forest percentage of like adjacencies (the percentage of all cell adjacencies involving the forest class that are forest-to-forest adjacencies), perimeter/area fractal dimension, and area-weighted mean area-perimeter ratio. Definitions of these metrics are found in McGarigal (1999). These metrics were chosen because they repre-

Table 2. Within-ecoregion analysis of the linear trend component of LPMs and LULC[a]

Landscape metric	Linear trend	MACP	SE Plains	N. Pied.	Pied.	B.R. Mtns.
% Urban	↑	< 0.01	< 0.01	< 0.01	< 0.01	0.01
% Forest	↓	< 0.01	0.29	< 0.01	< 0.01	< 0.01
% Agriculture	↓	0.49	< 0.01	< 0.01	0.04	0.13
No. patches	↑	< 0.01	< 0.01	0.02	< 0.01	< 0.01
No. forest patches	↑	0.02	0.03	0.01	0.04	< 0.01
Total forest edge	↑	0.05	< 0.01	< 0.01	0.03	< 0.01
Forest A/W MPS	↓	0.09	0.32	< 0.01	0.1	< 0.01
P/A fractal dimension	↑	0.04	0.06	0.07	0.45	0.01
A/W perimeter–area ratio	↑	0.01	< 0.01	< 0.01	0.32	< 0.01
Forest % like adjacencies	↓	< 0.01	0.01	< 0.01	< 0.01	< 0.01

[a] Values shown are *P* values (one-sided alternative hypothesis). The arrows under the "Linear trend" column show the direction of the trend for each metric. The trends for each metric were consistent for all five ecoregions, even if the trend for an ecoregion was not statistically significant. MACP, Middle Atlantic Coastal Plain; SE Plains, Southeastern Plains; N. Pied, Northern Piedmont; Pied, Piedmont; B.R. Mtns., Blue Ridge Mountains; MPS, mean patch size; P/A, perimeter–area; A/W, area-weighted.

sent the main aspects of landscape pattern such as patch size, texture, and shape (e.g., see Li and Reynolds 1995, Riitters and others 1995, Cain and others 1997, Griffith and others 2000). Moreover, mean patch size, number of forest patches, interspersion, and forest edge amounts were useful in discriminating contiguous from fragmented forests in Virginia (Trani and Giles 1999). We also analyzed trends in percent agriculture, urban, and forest. It is important to note that all metrics for all dates were calculated on maps with a 60-m cell size. While MSS, TM, and enhanced thematic mapper+ images were used as interpretation aids, classification of spectral values into LULC classes was not performed on these images.

For each ecoregion, metrics for each of the sample blocks were used in statistical analysis. Preserving metrics for each block allowed the calculation of means and variances necessary for the statistical tests. A repeated measures analysis (Meredith and Stehman 1991, Zar 1999) was used to determine whether statistically significant trends in LPMs and LULC proportions existed within each ecoregion. The analysis is based on fitting a quadratic polynomial to approximate the response of each metric over time. For each sample block, the least-squares estimates of the orthogonal polynomial linear and quadratic terms were computed using the LPM or LULC proportion as the dependent variable and time as the independent variable. To evaluate the statistical significance of the linear component of the trend over time, a one-sample *t* test was conducted with the estimated linear term derived for each sample block as the variable. That is, the slope estimated for each block was the variable analyzed, and the test determined if the population mean linear trend component was nonzero. A test for nonlinearity in the trend was obtained by conducting a one-sample *t* test using the estimates of the quadratic term computed for each sample block (i.e., determine if the mean quadratic term was nonzero). We further evaluated whether the linear trend differed among ecoregions. A standard one-way analysis of variance, with the ecoregions identified as the treatment groups, was performed using the estimated slope for each sample block as the response variable. Fisher's protected least significant difference was used to assess the statistical significance of differences in the linear trend between pairs of ecoregions.

Results

Landscape Pattern Metrics

In all five ecoregions, most of the seven pattern metrics showed statistically significant ($\alpha = 0.05$) trends toward a more fine-grained landscape, and in particular a more fine-grained forest cover (i.e., more numerous and smaller forest patches) (Table 2, Figures 2, 3, 4, 5, 6). The only metrics not showing a significant trend were forest area-weighted mean patch size in the Middle Atlantic Coastal Plain, Southeastern Plains, and Piedmont, and perimeter-area fractal dimension in the Southeastern Plains. The only ecoregion in which a more fine-grained forest landscape was not clearly evident was the Southeastern Plains, because forest area-weighted mean patch size did not show a significant decreasing trend. However, regeneration of harvested areas may connect patches in a time period to disrupt an overall declining trend (e.g., forest area-weighted mean patch size in 1992 in the Southeastern Plains) (Figure 3). The opposite situation may occur in the Middle Atlantic Coastal Plain (Figure 2), where the cycle of timber harvesting might break up several

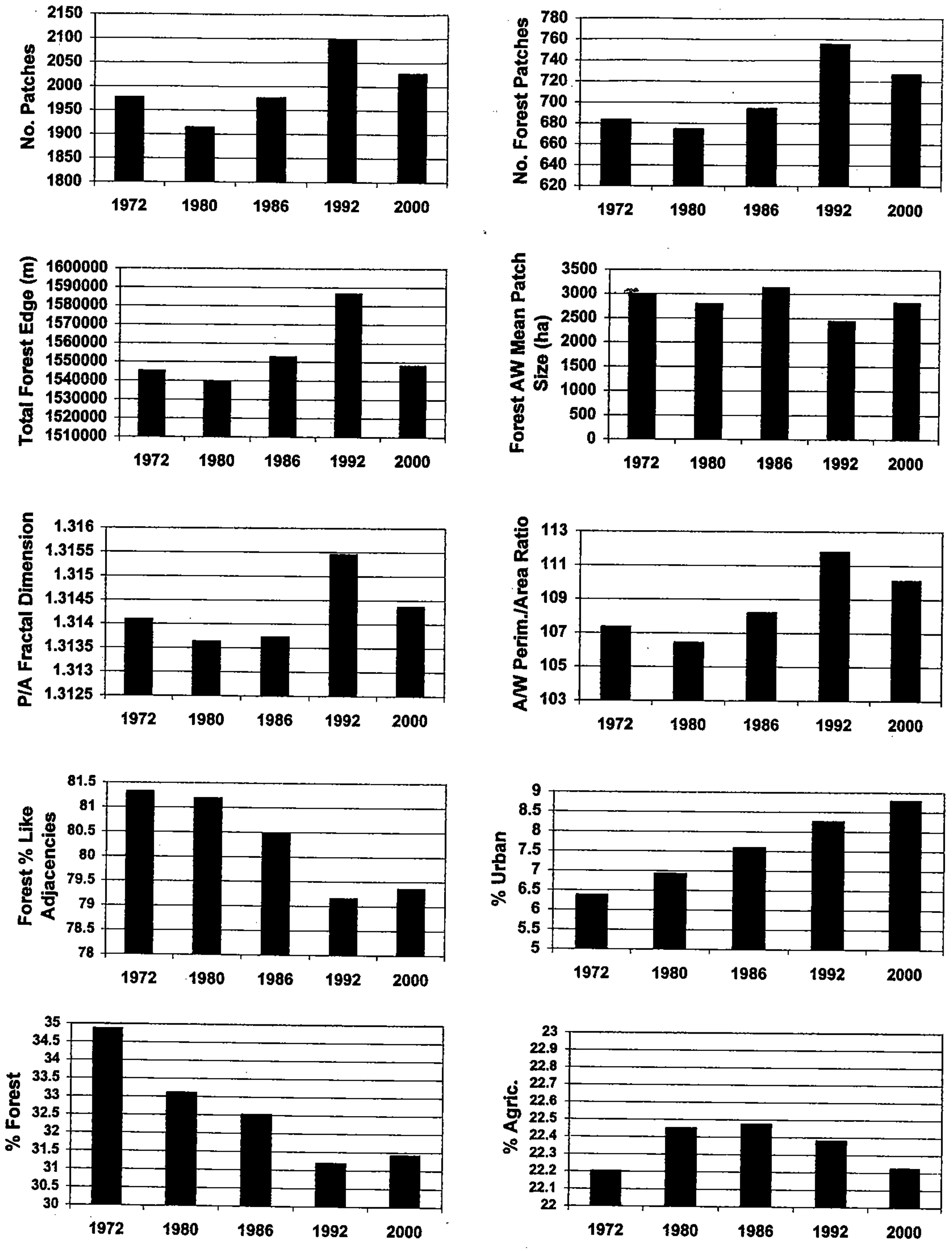

Figure 2. Trends in landscape pattern in the Middle Atlantic Coastal Plain.

patches during a time period (e.g., see number of patches and forest edge in 1992). In four of the five ecoregions, there was a statistically significant increase in perimeter-area fractal dimension with time, indicating a landscape with patches that have more convoluted boundaries. This result is counter to frequent observations in the literature that anthropogenic landscape changes generally result in patches with less complex-shaped boundaries.

For some of the LPMs, there was also a significant nonlinear component to the trend, which can take one of several forms: a leveling off, an exponential-like in-

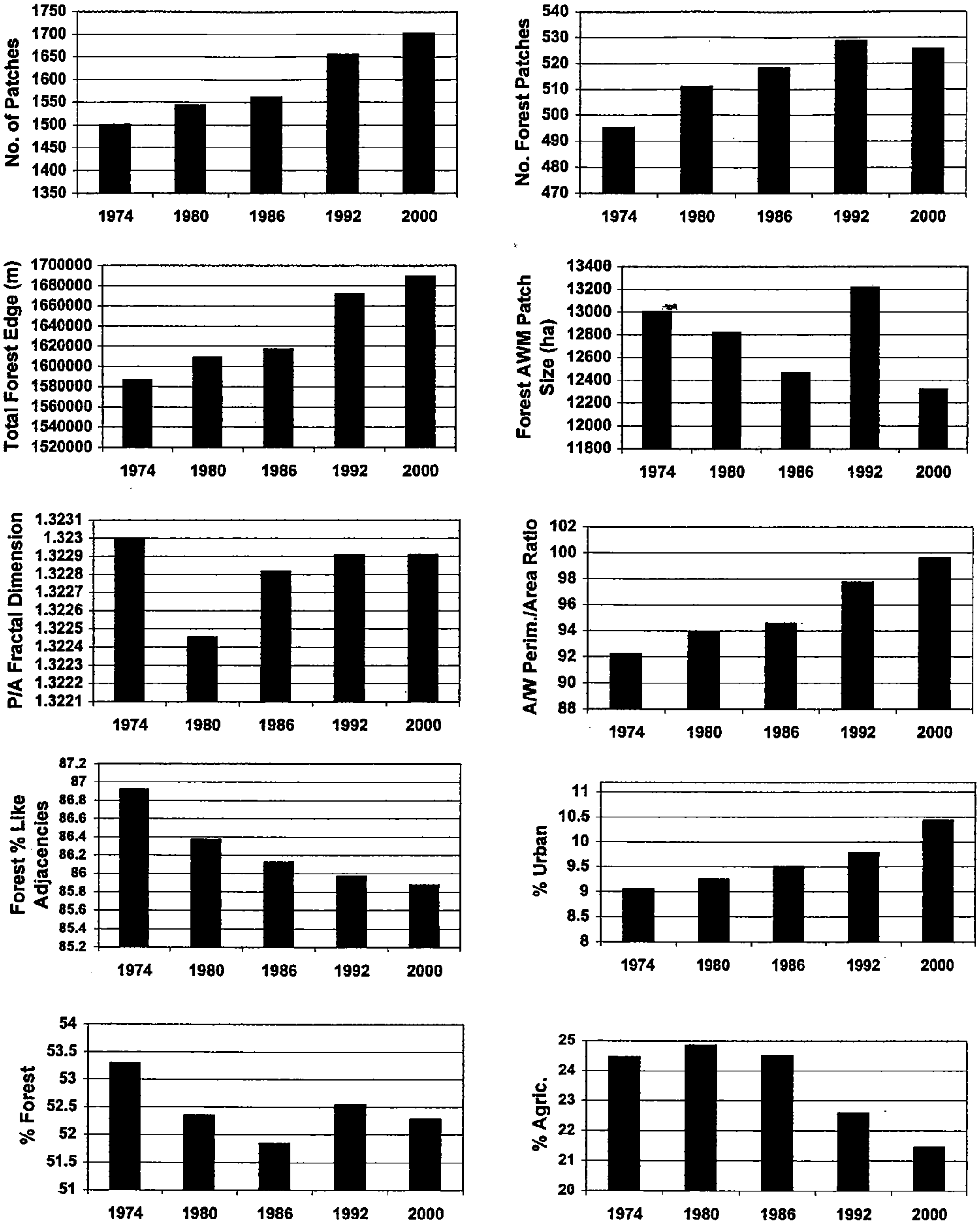

Figure 3. Trends in landscape pattern in the Southeastern Plains.

crease or decrease of the trend, or a convex or concave shaped trend (Table 3, Figures 2, 3, 4, 5, 6). Equations and coefficients for the fitted linear or quadratic models are shown in Appendix I. In the Middle Atlantic Coastal Plain, in particular, such nonlinear trends exist for six of the seven LPMs. The nonlinearity in this ecoregion is likely caused by spikes (e.g., forest total edge, Figure 2) or a concave shape (e.g., percent agriculture in Figure 2). Accelerating trends (i.e., exponential-like) are clearer in other ecoregions. In the Piedmont, for example, the trend in number of patches is accelerating (Figure 5), as is the decrease in the percentage of forest pixels adjacent to other forest pixels.

Statistically significant ecoregional differences were found for the trends for total forest edge and forest percent like adjacencies (Table 4, Figure 7). The increasing trend in forest edge was significantly higher in the Northern Piedmont than in other ecoregions, and

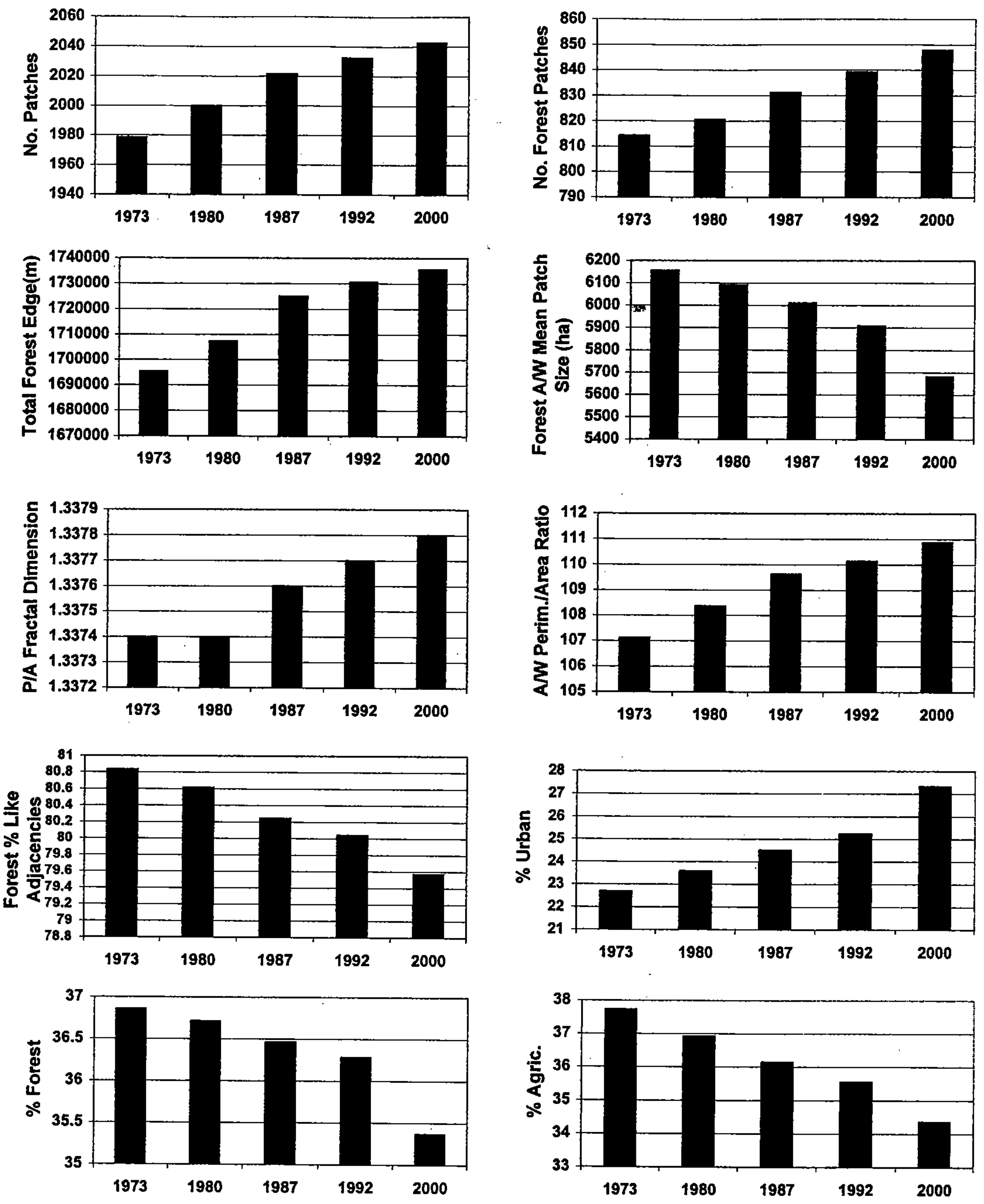

Figure 4. Trends in landscape pattern in the Northern Piedmont.

the decreasing trend in forest percent like adjacencies was significantly higher in the Middle Atlantic Coastal Plain than other ecoregions (Figure 7).

Land Cover Proportions

In all ecoregions, percent agriculture and percent forest declined, while percent urban increased (Table 2). The decline was statistically significant for percent agriculture in the Southeastern Plains, Northern Piedmont and Piedmont, and the decline in percent forest was significant in all but the Southeastern Plains. The increase in percent urban was significant in all ecoregions. The clearest examples of nonlinear trends in LULC proportions are for percent agriculture and percent urban in the Southeastern Plains, and for percent forest in the Northern Piedmont (Table 3, Figures 3, 4). The Southeastern Plains had a significant positive quadratic term for percent urban, and there is evi-

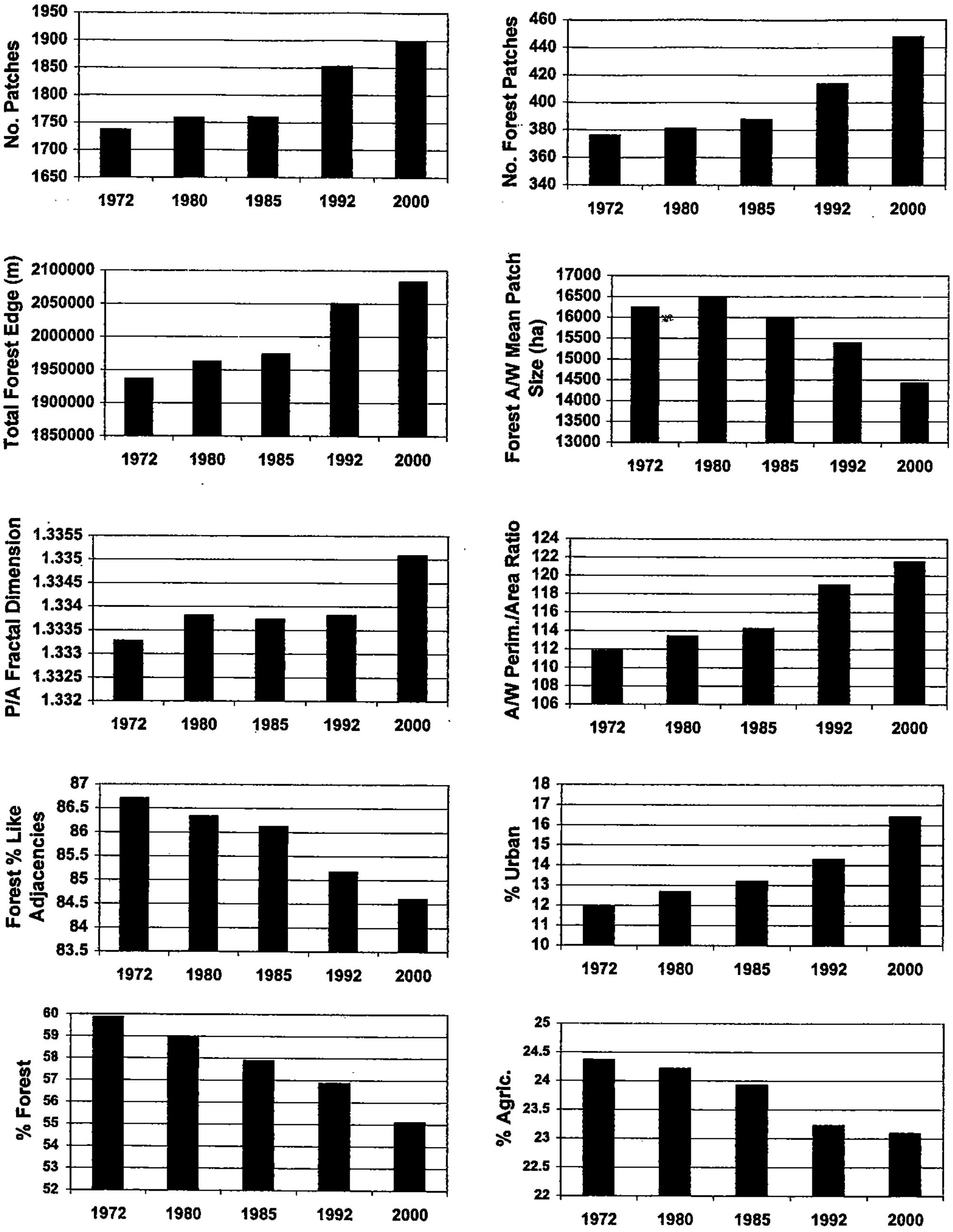

Figure 5. Trends in landscape pattern in the Piedmont.

dence, although less strong, of a similar phenomenon in the Northern Piedmont, Piedmont, and Blue Ridge Mountains (Table 3, Figures 3, 4, 5, 6). This illustrates an exponential-like growth characterizing trend in percent urban in these ecoregions. Loveland and others (2002) identified methodological issues that might have contributed to the observed exponential pattern—the difficulty of mapping change in the earlier dates (for which there was less aerial photography to use as ancillary data) and identifying small changes (such as small urbanized areas) on the coarser spatial resolution MSS imagery relative to the TM imagery used for the latter dates.

For all three of the LULC classes analyzed, the linear trend differed among ecoregions (Table 4, Figure 7). Both the Northern Piedmont and Piedmont showed statistically higher rates of urbanization than the Southeastern Plains and Blue Ridge Mountains, while the Piedmont had a significantly higher rate of forest decline than other ecoregions. The rate of agricultural

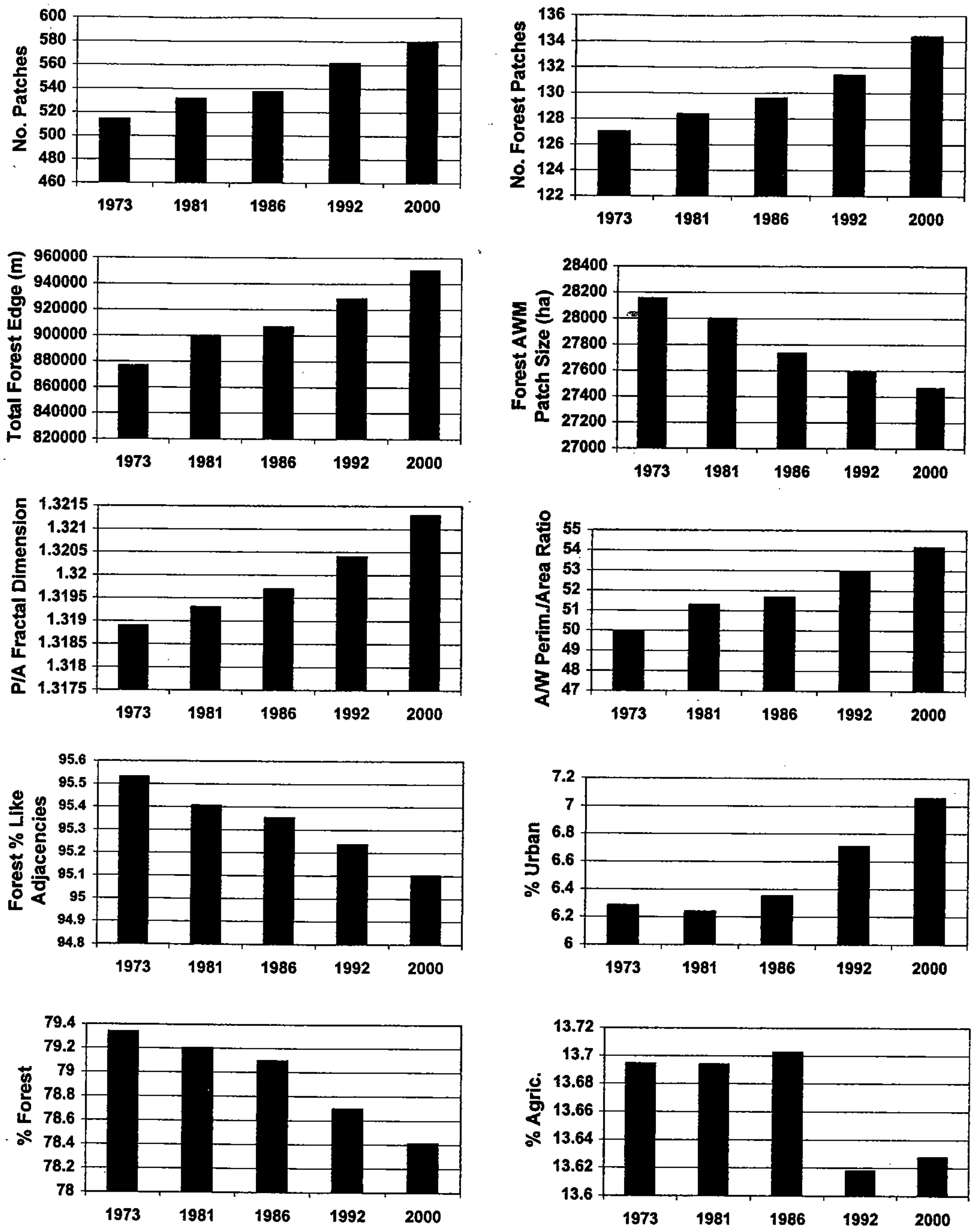

Figure 6. Trends in landscape pattern in the Blue Ridge Mountains.

decline was greatest in the Southeastern Plains and Northern Piedmont, and statistically separable from the lower rates of decline in the Blue Ridge Mountains, Middle Atlantic Coastal Plain, and Piedmont.

Discussion

Our analyses suggest that the landscape in the five study ecoregions became more fine-grained for the time period 1973–2000. This trend can be interpreted as resulting from an increasingly fragmented landscape, characterized by a greater number of patches, and patches having a greater perimeter-to-area ratio. The forest class contributes significantly to this increase in fragmentation of the landscape. There were more forest patches over time, smaller patches, greater total forest edge, and forest cells increasingly adjacent to nonforest cells (lower forest percent like adjacencies),

Table 3. Within ecoregion analysis of the quadratic trend component of LPMs and LULC[a]

Landscape metric	MACP	SE Plains	N. Pied.	Pied.	B.R. Mtns
% Urban		↑ 0.037	↑ 0.085	↑ 0.083	↑ 0.091
% Forest			↓ 0.026		
% Agriculture	↓ 0.045	↓ 0.024			
No. patches	↓ 0.023			↑ < 0.01	
No. forest patches	↓ 0.018			↑ 0.057	
Total forest edge	↓ 0.023		↓ < 0.01		
Forest A/W MPS	↑ 0.049			↓ 0.1	
P/A fractal dimension	↓ 0.05			↑ 0.078	
A/W perimeter–area ratio	↓ 0.021			↑ 0.061	
Forest % like adjacencies	↑ 0.088			↓ 0.047	

[a]Values shown are P values (two-sided alternative hypothesis). Arrows show the direction (sign) of the quadratic term (i.e., down is negative sign for the coefficient, up is a positive sign for the coefficient). No value means no significant quadratic trend.

Pied, Piedmont; MACP, Middle Atlantic Coastal Plain; N. Pied, Northern Piedmont; SE Plains, Southeastern Plains; B.R. Mtns, Blue Ridge Mountains; MPS, mean patch size; P/A, perimeter–area; A/W, Area-weighted.

Table 4. Comparisons among ecoregions of linear trend component for three LULC classes and various LPMs (i.e., does the linear trend in % urban cover differ among ecoregions)

		Ecoregion[a]				
Metric	Overall P value	MACP	SE Plains	N. Pied	Pied.	B.R. Mtns.
% Urban	< 0.01	ab3	a2	b5	b4	a1
% Forest	0.03	a4	a1	a3	b5	a2
% Agriculture	< 0.01	a1	b5	b4	a3	a2
No. Patches (ln)	0.12			NS		
P/A Fractal dimension	0.28			NS		
A/W P/A Ratio	0.22			NS		
No. Forest Patches (ln)	0.23			NS		
Total Forest Edge (km)	< 0.01	a1	a3	b5	a4	a2
Forest AW MPS (ha)	0.68			NS		
Forest % Like Adj.	0.02	b5	a2	a3	a2	a1

[a]Ecoregions sharing the same letter for a given metric (within a row) are not statistically significantly different based on Fisher's protected least significant difference. The numbers indicate the rank of the ecoregion for that metric in terms of the strength of the increase or decrease of the linear component (1 = weakest trend, 5 = strongest trend). The overall P value is derived for the null hypothesis of no differences in trend among all ecoregions versus the alternative hypothesis that at least two ecoregions differ (NS = not statistically significant at $\alpha = 0.05$).

indicating intrusion of other LULC classes into the forest cover. Interpreting the significant nonlinear components of these trends yields additional understanding of geographic variation in landscape trends. For example, in the Southeastern Plains, the significant negative quadratic term for percent agriculture (Table 3, Figure 3) suggests an accelerating conversion of agricultural land to industrial forest and/or the successional change of abandoned farms (Loveland and others 2002). That is, not only is percent agriculture declining, it is declining at an accelerated rate. Similarly, the significant negative quadratic term for percent forest in the Northern Piedmont indicates a decline more extreme than just linear.

The potential natural vegetation of the study region is predominantly forest, so landscape fragmentation is an important issue. As our results show, evidence of forest fragmentation has clearly occurred, particularly in the Piedmont and Northern Piedmont. Even the Blue Ridge Mountains showed evidence of fragmentation, although the trends were less strong than in other ecoregions. The ramifications of fragmented forests are commonly discussed in the literature (Trani and Giles 1999, Griffiths and others 2000, Marzluff and Ewing 2001, Carsjens and van Lier 2002, Olff and Ritchie 2002, Riitters and others 2002). Fragmentation is important not only to vertebrate distributions (Williams 1990, Saunders 1991, Forman 1995), but also to changes in surface energy balance and forest microclimates (Fuller 2001). In addition, some species require extensive, undisturbed tracts of land and are affected by fragmentation. This is of particular concern in the

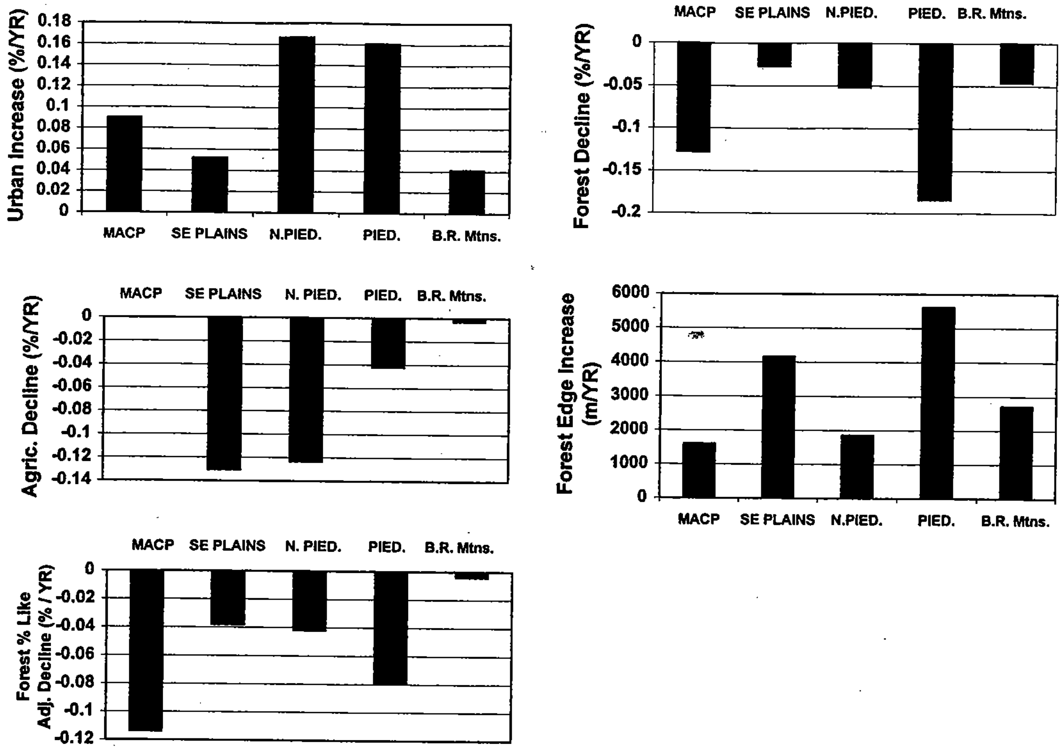

Figure 7. Ecoregional differences between trends in percent agriculture, urban, and forest from 1972 to 2000.

southeastern United States, where the long-leaf pine ecosystem, which once occupied 28 million ha, now contains only 600 ha of old-growth (Simberloff 1999).

In the Southeastern Plains, a statistically significant trend in forest mean patch size was not identified, although a spike in 1992 seems to disrupt an overall declining trend. However, this ecoregion was the only one that did not have a statistically significant decline in percent forest. This result may derive from the observed change of agriculture to forest second only to conversions related to forest harvesting (Loveland and others 2002). This is useful information, as it may shed light on the oft-cited eastern U.S. carbon sink (Turner 1990, Hu and others 1997, Houghton and Hackler 2000) by more precisely locating regions where agricultural land conversion to forest (through succession or industrial forestry practices) is occurring. Barlow (1998) has noted the effects of urban sprawl on reduction of timber harvesting, and our results have shown that urbanization is significantly increasing in the Southeastern Plains, even though the overall percentage of urban land is still relatively low.

Besides landscape fragmentation, another frequently cited landscape change resulting from human modification is straighter patch boundaries, as expressed by a lower value of fractal dimension. The slight increase in fractal dimension observed in this study likely arose from our analysis scale and the nature of change in this region. At our analysis scale, we were able to detect the character of residential development, e.g., many developments with circular drives and "spokes" of cul-de-sacs. There was scattered rural and roadside residential development that formed urban patches with convoluted shapes. Also, the timber harvest patches included many clear-cuts, which, although having straight edges, had polygonal shapes with many line segments.

The ability to discriminate among ecoregions on the basis of trends in LULC and LPMs (Table 4, Figure 7), supports the use of ecoregions as an appropriate unit upon which to document LULC change. Our findings corroborate those of Bourgeroun and others (2000), who showed that landscape structure derived from AVHRR data could be used to differentiate ecoregion patterns in the US Pacific Northwest. Ecoregions have also been found to offer distinct advantages in monitoring and assessments (Mladenoff and others 1997, Hughes and others 2000).

Several other examples illuminating ecoregional differences in landscape pattern emerged from our results. For example, the southern United States, often referred to as the Sun Belt, is rapidly growing in population and economic activity (Hartshorne 1997, Pandit 1997). However, certain areas are changing faster than others (Barkley and others 1996), as illustrated by our finding that the increasing trend in percent urban was statistically greater in the Northern Piedmont than in the Southeastern Plains (Table 4, Figure 7). Interestingly, the Northern Piedmont, as part of the decaying Rust Belt, also had a statistically stronger linear trend component in percent urban than the Southeastern Plains, which might be inappropriately grouped with other, more economically prosperous and growing regions of the south. Moreover, although forests are now being removed for residential and commercial purposes in the far more urbanized Northern Piedmont, the forest decline rate is greater in the Piedmont. The forest decline rate in the Piedmont also showed a stronger trend than other southern ecoregions such as the Blue Ridge Mountains and Southeastern Plains, the latter typically thought of as characterized by heavy timber harvesting activity. Although the Northern Piedmont has the highest percentage of urban land among the five ecoregions, agriculture remains an important part of its scenery and economy. The rate of decline in percent agricultural is higher in the Northern Piedmont than in the Piedmont, consistent with literature on the area describing exurban growth here (Bielski 1992, Greene and Benhart 1992) (Table 2, Figure 7). Finally, the rate of decline in percent agriculture is significantly higher in the Southeastern Plains than in the Middle Atlantic Coastal Plain (Table 2, Figure 7). Hart (2000) discussed the movement of the Cotton Belt from areas in the Southeastern Plains to the inland edge of the Middle Atlantic Coastal Plain, consistent with a movement away from agriculture in the Southeastern Plains.

Comparison with Previous LULC Studies

The present study reveals some interesting differences compared to two studies that examined landscape changes in the southeast United States. Turner (1990) examined landscape changes in selected counties in different physiographic provinces of Georgia. Using historic aerial photography from the 1930s through the late 1970s and early 1980s, she found that the Georgia landscape was less fragmented in the latter time periods than in the 1930s. Forest area increased in overall abundance and mean patch size increased. Henderson and Walsh (1995) also used aerial photography to examine landscape change in a Piedmont county in North Carolina from 1938 to 1982. They observed an increase in both coniferous and deciduous forest, and noted the forest class becoming more connected. There are several possible explanations for the differences we observed in our study, such as increasing forest fragmentation, a decrease in forest cover, and a decrease in area-weighted mean forest patch size. The primary reason may be that the time frame of their studies basically ended when ours began. Our study has included the recent economic boom that lasted virtually uninterrupted from the mid-1980s to 2000. This economic growth has undoubtedly had significant impacts on both the physical landscape and its geography. Our results, therefore, indicate an important directional change in landscape trends and shows the importance of continued landscape monitoring. Riitters and others (2002) also found a highly fragmented forest landscape in the United States, with the exception of a few regions including the Southern Appalachians. Our study documented significant increasing trends in the number of forest patches and forest edge in this region, but more studies are needed to show if this small overall change has significant ecological impact.

There are similarities in the studies as well. Their studies showed that diversification of the landscape increased as did percent urban land. Our studies also showed a decreasing trend in the Simpson's diversity index (not shown), which indicates a more diverse landscape, as well as statistically significant increases in percent urban in all ecoregions. Both studies documented a decrease in agricultural land in the Piedmont, and this decrease has continued, as evidenced by data from the present study. Although our analysis focused more on forests, the methods presented here could be applied to other LULC types as well, such as agriculture in the Midwest and Central United States.

Conclusions

It is important to start examining long-term landscape trends using the existing and growing archive of satellite imagery. Documenting the trends in LULC and LPMs contributes insights to understanding ecological and human processes. To further elucidate the human–ecological process linkages, our research will also examine human geography–landscape ecology interactions, such as studying socioeconomic drivers of LULC change, an integral component to understanding why spatial patterns change (Vaitkus 2002). This paper documents trends in landscape pattern in a quantitative manner, as opposed to qualitative comparison of LULC proportions or metrics. Our study contributes to the methodological development and understanding of monitoring trends in landscape pattern. In order to plan landscapes, they must be understood within their spatial and temporal contexts, as they change both ecologically and culturally (Marcucci

2000). The essence of monitoring is to function over a period of time (Mol 2001), and it is important to establish a baseline to enable the detection of development trends in a rapidly changing environment (Kammerbauer and others 2001). Lundquist and others (2001) similarly stress the importance of using landscape metrics and creating landscape "profiles." Our results establish a baseline of landscape profiles that may serve the needs of some monitoring objectives. Although our results detected few ecoregional differences in trends of LPMs, early analyses indicate ecoregional differences in LPMs at individual points in time (i.e., ecoregion differences in LPM status). Continued testing of landscape status in addition to trends will further elucidate the usefulness of ecoregions for landscape monitoring. These preliminary results indicate that ecoregions are useful because they work in discriminating certain differences in trends among different geographic areas.

The sampling and analysis protocol detected statistically significant trends within ecoregions in several LPMs as well as the major LULC types. The general trend was toward a more fine-grained landscape. The ability to detect such trends, despite the relatively high variation in both the LULC proportions and LPMs among sample blocks, suggests that a sampling based approach based on intensive manual interpretation of each sampled block is potentially a viable, practical alternative to using complete coverage of LULC information for detecting trends in LPMs. This is true not only for relatively dynamic landscapes such as the Piedmont and Middle Atlantic Coastal Plain, but also for less dynamic regions such as the Blue Ridge Mountains. Our approach is also amenable to comparing trends in LULC proportions and LPMs among ecoregions. Ecoregions are useful because they work. We were able to discriminate among the ecoregions, particularly on the basis of trends in the LULC proportions, supporting the potential use of ecoregions as a regionalization framework. However, we acknowledge that similar results may be obtained from other regionalization units that incorporate physical and anthropogenic landscape attributes. Future work will extend the analyses to additional regions, compare ecoregions at single points in time, and investigate relationships between trends in landscape pattern and various environmental phenomena such as vertebrate population dynamics and water quality.

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References

Barkley, D. L., M. S. Henry, and S. Bao. 1996. Identifying "spread" versus "backwash" effects in regional economic areas: A density functions approach. *Land Economics* 72(3):336–357.

Barlow, S., I. Munn, D. Cleaves, and D. Evans. 1998. The effect of urban sprawl on timber harvesting. A look at two southern states. *Journal of Forestry* 96(12):10–14.

Bielski, F. R. 1992. Population and housing trends. Chester County: the new urban frontier 1980-1990. *Pennsylvania Geographer* 30(1):47–61.

Bourgeron, P. S., H. C. Humphries, J. A. Barber, S. T. Turner, M. E. Jensen, and I. A. Goodman. 1999. Impact of broad- and fine-scale patterns on regional landscape characterization using AVHRR-derived land cover data. *Ecosystem Health* 5(4):234–258.

Cain, D., K. Riitters, and K. Orvis. 1997. A multi-scale analysis of landscape statistics. *Landscape Ecology* 7:291–302.

Carsjens, G. J., and H. N. van Lier. 2002. Fragmentation and land-use planning—an introduction. *Landscape and Urban Planning* 58:79–82.

Donoghue, D. M. 2002. Remote sensing: environmental change. *Progress in Physical Geography* 26(1):144–151.

Dramstad, W. E., W. J. Fjellstad, G. H. Strand, H. F. Mathiesen, G. Engan, and J. N. Stokland. 2002. Development and implementation of the Norwegian monitoring programme for agricultural landscapes. *Journal of Environmental Management* 64:49–63.

Dumanski, J., and C. Pieri. 2000. Land quality indicators: research plan. *Agriculture, Ecosystems, and Environment* 81:93–102.

Fjellstad, W. J., E. Dramstad W., H. Strand G., and G. L. A. Fry. 2001. Heterogeneity as a measure of spatial pattern for monitoring agricultural landscapes. *Norsk Geografisk Tidsskrift* 55:71–76.

Forman, R. T. T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge University Press, Cambridge 632.

Fuller, D. O. 2001. Forest fragmentation in Loudon County, Virginia, USA evaluated with multitemporal Landsat imagery. *Landscape Ecology* 16:627–642.

Galehouse, R. F. 1981. Land planning for large-scale residential development. *Urban Land* 40(9):12–18.

Greene, R. A., and J. E. Benhart. 1992. The encroachment of

Megalopolis into the Great Valley: evidence from the Cumberland Valley. *Pennsylvania Geographer* 30(1):30–46.

Griffith, J. A. 1998. Connecting ecological monitoring and ecological indicators: a review of the literature. *Journal of Environmental Systems* 26(4):325–363.

Griffith, J. A., E. A. Martinko, and K. P. Price. 2000. Landscape structure analysis of Kansas at three scales. *Landscape and Urban Planning* 52:45–61.

Griffiths, G. H., J. Lee, and B. C. Eversham. 2000. Landscape pattern and species richness: regional scale analysis from remote sensing. *International Journal of Remote Sensing* 21(13/14):2685–2704.

Gustafson, E. 1998. Quantifying landscape spatial pattern: what is the state of the art? *Ecosystems* 1:143–156.

Harrison, A. R., and R. Dunn. 1993. Problems of sampling the landscape. Pages 101–109. Page 288 *in* R. Haines-Young, D. R. Green, and S. Cousins. Eds, Landscape ecology and geographic information systems. Taylor and Francis, London.

Hart, J. F. 2000. The metempsychosis of King Cotton. *Southeastern Geographer* 00:93–105.

Hartshorne, T. A. 1997. The changed south, 1947–1997. *Southeastern Geographer* 37(2):122–139.

Henderson, B. M., and S. J. Walsh. 1995. "Plowed, paved, or in succession": land-cover change on the North Carolina piedmont. *Southeastern Geographer* 35(2):132–149.

Herzog, F., and A. Lausch. 2001. Supplementing land-use statistics with landscape metrics: some methodological considerations. *Environmental Monitoring and Assessment* 72:37–50.

Holm, A., L. Bennett, W. Loneragan, and M. A. Adams. 2002. Relationships between empirical and nominal indices of landscape function in the arid shrubland of Western Australia. *Journal of Arid Environments* 50:1–21.

Houghton, R. A., and J. L. Hackler. 2000. Changes in terrestrial carbon storage in the United States. 1. The roles of agriculture and forestry. *Global Ecology and Biogeography* 9(2):125–144.

Hu, S., P. F. Hendrix, M. H. Beare, D. C. Coleman, and C. R. Carroll. 1997. Labile soil carbon pools in subtropical forest and agricultural ecosystems as influenced by management practices and vegetation types. *Agriculture, Ecosystems and Environment* 65(1):69–78.

Hughes, R. M., S. G. Paulsen, and J. L. Stoddard. 2000. EMAP-surface waters: a multiassemblage, probability survey of ecological integrity in the U.S.A. *Hydrobiologia* 422/423:429–443.

Imbernon, J., and A. Branthomme. 2001. Characterization of landscape patterns of deforestation in tropical rainforests. *International Journal of Remote Sensing* 22(9):1753–1765.

Jones, K. B., A. C. Neale, M. S. Nash, R. D. Van Remortel, J. D. Wickham, K. H. Riitters, and R. V. O'Neill. 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: A multiple watershed study from the United States Mid-Atlantic Region. *Landscape Ecology* 16:301–312.

Kammerbauer, J., B. Cordoba, R. Escolan, S. Flores, V. Ramirez, and J. Zeledon. 2001. Identification of development indicators in tropical mountainous regions and some implications for natural resource policy designs: an integrated community case study. *Ecological Economics* 36:45–60.

Lambin, E. F. 1999. Monitoring forest degradation in tropical regions by remote sensing: some methodological issues. *Global Ecology and Biogeography* 8:191–198.

Li, H., and J. Reynolds. 1995. On definition and quantification of heterogeneity. *Oikos* 73:280–284.

Loveland, T. R., T. L. Sohl, S. V. Stehman, A. L. Gallant, K. L. Sayler, and D. E. Napton. 2002. A strategy for estimating the rates of recent United Stated land-cover changes. *Photogrammetric Engineering and Remote Sensing* 68(10):1091–1099.

Lundquist, J. E., L. Lindner, and J. Popp. 2001. Using landscape metrics to measure suitability of a forested watershed: a case study for old-growth. *Canadian Journal of Forest Research* 31(10):1786–1792.

Marcucci, D. J. 2000. Landscape history as a planning tool. *Landscape and Urban Planning* 49(1&2):67–81.

Marzluff, J. M., and K. Ewing. 2001. Restoration of fragmented landscapes for the conservation of birds: A general framework and specific recommendations for urbanizing landscapes. *Restoration Ecology* 9(3):280–292.

McDonald, M. E. 2000. EMAP Overview: Objectives, approaches, and achievements. *Environmental Monitoring and Assessment* 64:3–8.

McGarigal, K. 1999. FRAGSTATS 3.01.02. Software and user's manual. http://www-unix-oit.umass.edu/|P5fragstat.

Meredith, M. P., and S. V. Stehman. 1991. Repeated measures experiments in forestry: focus on analysis of response curves. *Canadian Journal of Forest Research* 21:957–965.

Mladenoff, D. J., G. J. Niemi, and M. A. White. 1997. Effects of changing landscape pattern and U.S.G.S. land cover data variability on ecoregion discrimination across a forest-agriculture gradient. *Landscape Ecology* 12:379–396.

Mol, G., S. P. Vriend, and P. F. M. Van Gaans. 2001. Environmental monitoring in The Netherlands: Past developments and future challenges. *Environmental Monitoring and Assessment* 68:313–335.

Olff, H., and M. E. Ritchie. 2002. Fragmented nature: consequences for biodiversity. *Landscape and Urban Planning* 58:83–92.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* 77: 118–125, (and map supplement).

Omernik, J. M. 1995. Ecoregions—a framework for environmental management. Pages 49–62. Page 432 *in* W. Davis, and T. Simon. Eds, Biological assessment and criteria. Lewis Publishers, Boca Raton.

O'Neill, R. V., C. T. Hunsaker, K. B. Jones, K. Riitters, J. Wickham, P. Schwartz, I. Goodman, B. Jackson, and W. Baillargeon. 1997. Monitoring environmental quality at the landscape scale. *Bioscience* 47(8):513–519.

O'Neill, R. V., C. T. Hunsaker, S. P. Timmins, B. Jackson, K. B. Jones, K. Riitters, and J. Wickham. 1996. Scale problems in reporting landscape pattern at the regional scale. *Landscape Ecology* 113:169–180.

O'Neill, R. V., K. B. Jones, K. H. Riitters, J. D. Wickham, and I. A. Goodman. 1994. Landscape monitoring and assessment research plan. EPA/620/R-94/009. US Environmental Protection Agency, Las Vegas, Nevada.

Pandit, K. 1997. The southeastern migration turnaround and current patterns. *Southeastern Geographer* 37(2):238–250.

Petit, C. C., and E. F. Lambin. 2001. Integration of multi-source remote sensing data for land cover change detection. *International Journal of Geographical Information Science* 15(8)):785–803.

Reed, R. A., J. Johnson-Barnard, and W. L. Baker. 1996. Fragmentation of a forested Rocky Mountain landscape. *Biological Conservation* 75(3):267–276.

Riebsame, W. J., W. J. Parton, I. C. Galvin, L. Burke, L. Bohren, R. Yung, and E. Knop. 1994. Integrated modeling of land use and cover change. *Bioscience* 44(5):350–356.

Riitters, K. H., R. V. O'Neill, C. T. Hunsaker, J. Wickham, D. Yankee, S. P. Timmins, K. B. Jones, and B. Jackson. 1995. A factor analysis of landscape pattern and structure metrics. *Landscape Ecology* 10(1):23–39.

Riitters, K. H., J. D. Wickham, R. V. O'Neill, K. B. Jones, E. R. Smith, J. W. Coulston, T. G. Wade, and J. H. Smith. 2002. Fragmentation of continental United States forests. *Ecosystems* 5:815–822.

Saunders, D. A., R. J. Hobbs, and C. R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. *Conservation Biology* 5:18–32.

Simberloff, D. 1999. The role of science in the preservation of forest biodiversity. *Forest Ecology and Management* 115:101–111.

Sohl, T. L., and J. L. Dwyer. 1998. North American landscape characterization project: the production of a continental scale three-decade Landsat data set. *Geocarto International* 13:43–51.

Stehman, S. V., T. L. Sohl, and T. R. Loveland. 2003. Statistical sampling to characterize land-cover change in the U.S. Geological Survey land-cover trends project. *Remote Sensing of Environment* 86: 517–529.

Trani, M. K., and R. H. Jr. Giles. 1999. An analysis of deforestation: metrics used to describe pattern change. *Forest Ecology and Management* 114:459–470.

Turner, M. G. 1990. Landscape changes in nine rural counties in Georgia. *Photogrammetric Engineering and Remote Sensing* 56(3):379–386.

Vaitkus, M. R., ed. 2002. Final program and abstracts. 17th annual symposium of the International Association for Landscape Ecology—United States Regional Association (US-IALE). Landscapes in Transition: Cultural Drivers and Natural Constraints. University of Nebraska, Lincoln, USA.

Vogelmann, J. E., S. M. Howard, L. Yang, C. R. Larson, B. Wylie, and N. Van Driel. 2001. Completion of the 1990s national land cover data set for the conterminous United States from Landsat Thematic Mapper data and ancillary sources. *Photogrammetric Engineering & Remote Sensing* 67(6):650–662.

Walcott, S. M. 1999. High tech in the deep south: biomedical firm clusters in Atlanta. *Growth and Change* 30(1):48–74.

Williams, M. 1990. Forests. Pages 179–201. Page 713 *in* B. L. Turner, W. C. Clark, R. W. Kates, J. W. Richards, J. T. Matthews, and W. B. Meyer. Eds, The earth as transformed by human action: global and regional changes in the biosphere over the past 300 years. Cambridge University Press, Cambridge.

Yang, L., S. V. Stehman, J. H. Smith, and J. D. Wickham. 2001. Thematic accuracy of MRLC land cover for the eastern United States. *Remote Sensing of Environment* 76:418–422.

Zar, J. H. 1999. Biostatistical Analysis, 4th ed. Prentice-Hall, Upper Saddle River, New Jersey 663.

Appendix 1: Regression prediction equations for each landscape metric and ecoregion for fitting up to a quadratic polynomial model of the relationship over time

Table A1 Regression prediction equations for each landscape metric and ecoregion for fitting up to a quadratic polynomial model of the relationship over time[a]

Landscape metric	Middle Atlantic Coastal Plain	Southeastern Plains	Northern Piedmont	Piedmont	Blue Ridge Mountains
% Urban	6.36 + 0.93t	$9.06 + 0.016t + 0.0012t^2$	$22.74 + 0.088t + .0029t^2$	$11.99 + 0.038t + 0.0046t^2$	$6.26\text{-}0.008t + 0.0014t^2$
% Forest	34.36-0.132t		$36.82 + 0.0056t\text{-}0.0021t^2$	60.08-0.177t	79.42-0.036t
% Agriculture	22.35-0.0006t	$24.62 + 0.047t\text{-}0.0064t^2$	37.77-0.124t	24.46-0.053t	
No. patches	1873 + 7.69t	1489.5 + 7.82t	1983 + 2.41t	$1735 + 1.5t + 0.177t^2$	513 + 2.44t
No. forest patches	651 + 3.64t	500 + 1.77t	813.6 + 1.29t	$376\text{-}0.232t + 0.108t^2$	126.6 + 0.271t
Forest total edge	1523713 + 1788t	1581686 + 4027t	$996828 + 114419t\text{-}3013t^2$	1925329 + 5765t	877102 + 2673t
Forest area weighted mean patch size	3443-34.0t		6198-17.31t	$16316 + 26.55t\text{-}3.645t^2$	28143-26.84t

Landscape metric	Middle Atlantic Coastal Plain	Southeastern Plains	Northern Piedmont	Piedmont	Blue Ridge Mountains
Perimeter area fractal dimension	$1.313 + 0.00021t$	$1.323 + 0.0000037646t$	$1.337 + 0.0000165t$	$1.333 - 0.000009451t + 0.000002447t^2$	$1.319 + 0.0000443t$
Area weighted perimeter–area	$104.7 + 0.258t$	$91.9 + 0.281t$	$107.4 + 0.139t$	$111.6 + 0.192t + 0.0069t^2$	$50.0 + 0.154t$
Forest % Like adjacencies	$82.1 - 0.118t$	$86.7 - 0.038t$	$80.9 - 0.047t$	$86.7 - 0.045t - 0.0013t^2$	$95.5 - 0.0135t$

[a] The explanatory variable t is coded as $t = 0$ for 1973 through $t = 27$ for 2000. The degree of the polynomial reported is determined by the statistical significance of the linear and quadratic components of trend evaluated in the repeated measures analyses (Tables 2 and 3). The linear and quadratic terms were retained if their *P* value was below 0.10. In cases for which the repeated measures analysis showed a significant quadratic but not a significant linear component, the regression prediction equation includes both the linear and quadratic terms. The estimated regression coefficients shown in this appendix table are "partial regression coefficients" and therefore the explanatory variables t and t^2 are not orthogonal. When both t and t^2 are included in the model, the sign of the coefficient of the linear term may not match the sign reported in Table 2 in which orthogonal polynomials were analyzed. The partial regression coefficients reflect an adjustment for any shared explanatory information contributed by all other explanatory variables in the model, whereas the linear component in the orthogonal polynomial analysis is not adjusted for the shared contribution with the quadratic component.