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Mapping Land-Use Change and Monitoring the Impacts of Hardwood-to-Pine Conversion on the Southern Cumberland Plateau in Tennessee

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ABSTRACT: Over the past two decades, forests in the southeastern United States have undergone dramatic changes as the result of urban sprawl and conversion to intensively managed pine plantations. The Cumberland Plateau, an important ecoregion in the southeastern United States, contains some of the largest remaining tracts of privately owned, native hardwood forest in North America. These ecologically important forests have been undergoing increasingly rapid rates of hardwood-to-pine conversion, much of which has gone undetected by large-scale statewide inventories. Forest conversion in Tennessee's southern Cumberland Plateau provides a case study highlighting the need for interdisciplinary and spatially explicit assessments of the impact and drivers of land-use change at smaller scales. Aerial and satellite imagery were used to create computer-generated maps of land use and forest cover for a 243 000 ha study area within a seven-county region of the southern Cumberland Plateau in Tennessee to track and document patterns of forest change and conversion between 1981 and 2000. The ecological impact of forest harvesting and hardwood-to-pine conversion was evaluated by (i) monitoring aquatic macroinvertebrate diversity, (ii) tracking breeding-bird populations, and (iii) comparing calcium (Ca) stores and cycling in a chronosequence of hardwood to first- and second-rotation loblolly pine (*Pinus taeda*) plantations. It was found that 14% of native forest cover had been lost since 1981, 74% of which resulted from hardwood-to-pine conversion. It was also found that the rate of conversion to pine doubled from 1997 to 2000. Water quality in streams, as measured by the abundance of critical macroinvertebrates, was significantly lower in recently logged sites than in undisturbed native forest. Surveys of breeding-bird populations showed that pine plantations of several age classes had lower species richness and evenness than did native oak–hickory forests. Despite similar soil concentrations of Ca in native hardwood, mature first-rotation, and early second-rotation pine, changes were found in aboveground Ca storage that suggest substantial system Ca losses that may limit productivity of second-rotation pine or regrowth of oak–hickory forest. As part of the ongoing research on the socioeconomic drivers of land-use change on the Cumberland Plateau, it was found that Tennessee's major forest conservation incentive program only delays forest conversion for a few years while subsidizing landowners who would not have converted their land in the absence of the program. These results demonstrate the need for more detailed and multidisciplinary research conducted at smaller scales so as to enhance the understanding of the impact and drivers of land-use change at larger scales.

KEYWORDS: Calcium cycling, Oak–hickory forest, Pine conversion

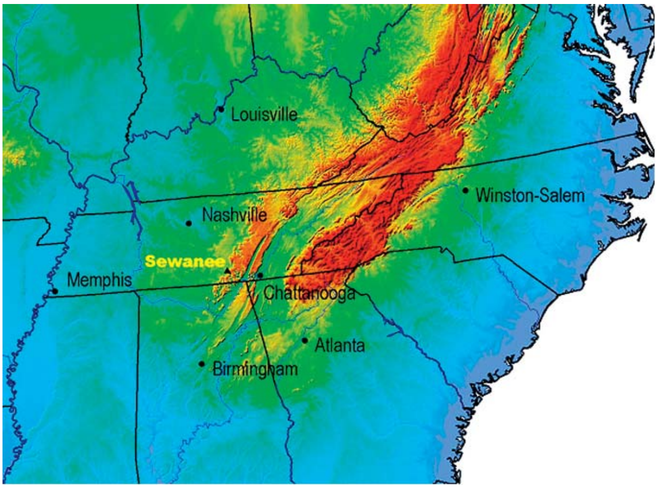

Figure 1. Location of the Cumberland Plateau (red and yellow), and study site of the SAA Forestry Demonstration Project near Sewanee, TN (35°N, 85°W).

1. Introduction

Extending from northern Alabama, through Tennessee, Kentucky, Virginia, and into West Virginia, the Cumberland Plateau covers a total surface area of 55 000 km^2 (Figure 1) that supports one of the most diverse woody plant communities in the eastern United States (Ricketts et al., 1999). These upland forests, predominated by an oak–hickory [(*Quercus* spp.)–(*Carya* spp)] complex, serve as a critical neotropical migratory songbird habitat (Haney and Lydic, 1999) and protect the headwaters to some of the most biologically diverse freshwater stream systems in the world (Ricketts et al., 1999). The region also supports some of the highest predicted herpetofaunal diversity in the southeastern United States (Durham, 1995). The hard mast (acorns) associated with the mature oak canopy of this ecosystem serves as a keystone resource within the food web that affects the survivorship of hundreds of animal species inhabiting the forest (McShea and Healy, 2002). The shallow sandstone-derived soils underlying the upland forests of the Cumberland Plateau are acidic and low in base cations (Francis and Loftus, 1977), rendering the entire system highly sensitive to nutrient removals from forest harvesting and acid precipitation (Adams et al., 2000).

Like much of the forests in the eastern United States, the native deciduous hardwood ecosystems of the Cumberland Plateau have undergone a long history of land-use change driven by agricultural conversion, timber extraction, and more recently, urban sprawl and large-scale conversion to intensively managed pine plantations (Bonnicksen and Burton, 2003; Wear and Greis, 2001). The portion of the Cumberland Plateau located in Tennessee contains some of the largest remaining tracts of privately owned hardwood forest in North America. Starting in the 1950s and up to the present, native forest on the southern Cumberland Plateau in Tennessee has been converted to loblolly pine (*Pinus taeda*) plantations at increasing rates (Hinkle et al., 1993). Evans et al. (Evans et al., 1999) analyzed 18 yr (1981–98) of forest change for Grundy County, Tennessee, and found that 5323 ha of native forest area was converted to pine plantations and 470 ha was cleared for agricultural and residential use. This overall forest conversion resulted in a 12% net loss of privately owned, native hardwood habitat, the peak of which has occurred since 1994, concurrent with the rise in chip mill activity in this general region (Draper, 1999).

Despite evidence of increasing harvest and pine conversion rates in this region, the Tennessee Division of Forestry recently reported that the state's forests were improving in condition since the 1950s, largely because of statewide increases in forest cover and growth rates. These observations were based upon a regional U.S. Forest Service Assessment [the Forest Inventory and Analysis (FIA)], which is a valuable tool for examining changes in forest cover on a statewide basis. However, as Schweitzer (Schweitzer, 2000) indicated, the sampling points were too far apart to detect significant landscape changes on a county basis. As a result, subregional land-use change was not recognized by the state. Without formal recognition that hardwood-to-pine conversion has become increasingly widespread in the southern Cumberland Plateau, there has been no attempt to monitor the ecological impact of harvesting and intensive plantation management on watershed integrity, local populations of organisms, and biogeochemical cycling. Moreover, the complex matrix of private ownership of the region's forests has made it difficult to analyze the socioeconomic drivers of this land-use change.

Because a very large proportion of forest land is privately owned in the states that compose the Cumberland Plateau, landscape-level change becomes a collective function of the decisions made by the region's myriad of small, large, and often absentee landholders; the resource professionals who advise them; and the government officials who enforce regulations and provide incentives for various land-use options. In this setting, public access to scientifically sound landscape-level information about the effect of different land uses on regional biodiversity, the local resource base, and ecosystem services become increasingly important if private landowners and public agencies are to engage in environmentally responsible land management.

Information derived from large-scale studies of land-cover change may not lend itself to this purpose because the ecological and socioeconomic conditions that govern forest resource sustainability often vary tremendously across the landscape and among ecoregions. Thus, more detailed and cost-effective assessments of smaller areas that provide spatially explicit information about the ecological impact of land-use practices on a subregional landscape are needed to help guide the decisions of local land managers and resource professionals. A small area assessment generates the information necessary to allow each land-use decision to be made within the context of what is happening to the greater landscape and provides this information at appropriate spatial and temporal scales.

In 2000, the Landscape Analysis Lab (LAL) at the University of the South in Sewanee, Tennessee, initiated a Small Area Assessment (SAA) Forestry Demonstration Project to develop and identify technologically accessible and cost-effective ways of generating landscape-level information to be used in

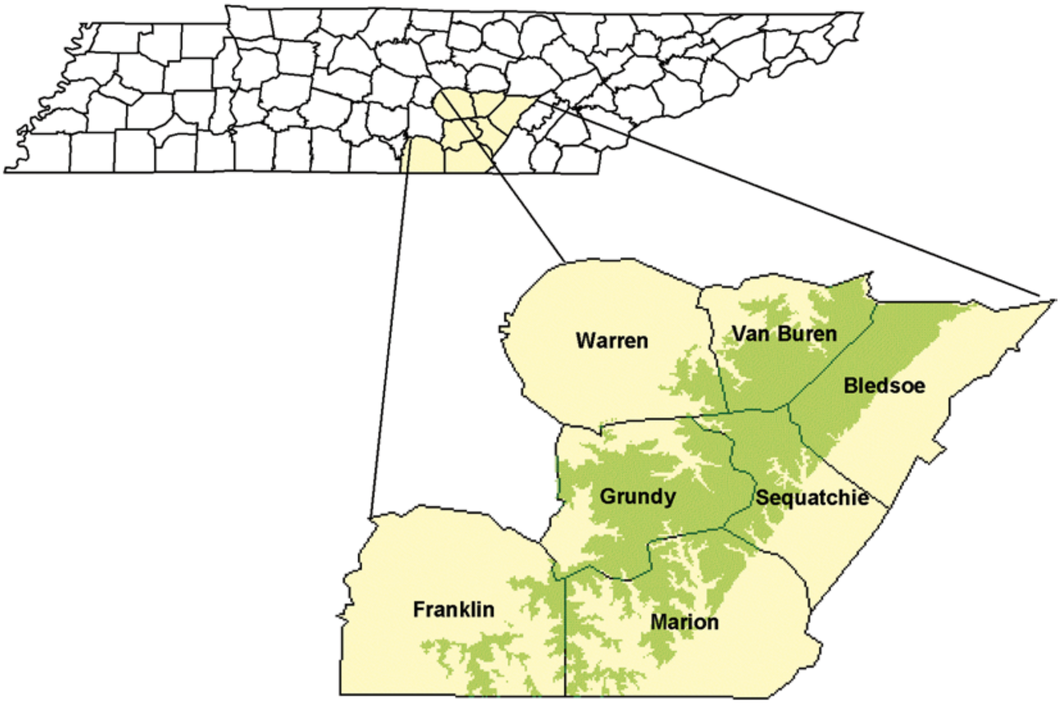

Figure 2. Study area on the southern Cumberland Plateau intersecting the seven-county region (243 000 ha) over which land-cover change was quantified from 1981 to 2000 by the SAA Forestry Demonstration Project.

subregional studies (Evans et al., 2002). This interdisciplinary assessment examined changes in forest cover from 1981 to 2000 in a 243 000 ha region located within seven counties of the Cumberland Plateau using satellite imagery and aerial photography (Figures 1 and 2). Field surveys of breeding birds were also conducted to assess bird community responses to forest change, and benthic macroinvertebrate populations were compared among intact forest and disturbed sites as a means of monitoring watershed integrity. More recently, we have initiated research to determine how hardwood-to-pine conversion affects ecosystem biogeochemistry, and in particular, the cycling and storage of calcium (Ca). We have also begun to explore landownership patterns and socioeconomic drivers behind land-use change in the seven-county region.

Our objectives for writing this paper are threefold. First, we summarize the methods and results of the mapping and biomonitoring studies undertaken by the 2000 SAA Forestry Demonstration Project. Second, we present results from ongoing studies of the impact of hardwood-to-pine conversion, and the subsequent harvest on aboveground storage and cycling of calcium. Third, we describe an ongoing examination of the socioeconomic drivers of land-use change in this region.

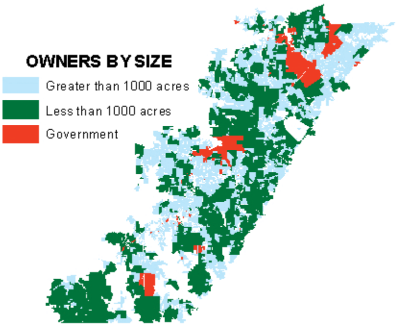

Figure 3. Distribution of landownership by landowners holding 1000 acres (405 ha) or more in the seven-county study region (243 000 ha) on the southern Cumberland Plateau, as calculated by 2000–01 county tax parcel maps digitized by the LAL.

2. Methods

2.1. Study area

The research described in this paper takes place within a 243 000 ha upland region of Tennessee's southern Cumberland Plateau, which extends into seven counties (Figures 1 and 2), near the town of Sewanee (35°N, 85°W). Despite a long history of human occupation of the Cumberland Plateau, much of this area escaped forest conversion because the region's drought-prone, nutrient-poor soils and cooler climate were less favorable for agriculture than the surrounding valleys. A portion of the Plateau's forest was once converted to pasture, but a substantial amount has been retained in forest (~80%) (Evans et al., 2002). Most of this forest was repeatedly logged (high graded) over the past 150 years and left to naturally regenerate. Currently, these forests have regenerated from the last round of widespread tree cutting that occurred in the first half of the 1900s.

The plateau forest canopy is composed predominately of a mixture of oak species (*Q. prinus, Q. coccinea, Q. velutina, Q. alba, Q. stellata*), hickory species (*C. glabra, C. pallida, C. Tomentosa*), along with sourwood (*Oxydendrum arboreum*), black gum (*Nyssa sylvatica*), and red maple (*Acer rubrum*) (Hinkle et al., 1993; Ramseur and Kelly, 1981). The soils on the plateau are also diverse, but a large percentage of this region is covered by sandstone-derived Ultisols that are inherently low in base cations and orthophosphate (Francis and Loftus, 1977). Elevations in this region generally range from 450 to 600 m.

Approximately 86% of forest in Tennessee is privately held by a matrix of large and smaller landholders (Schweitzer, 2000). Thus, ownership in the Cumberland Plateau is largely fragmented with little communication between adjacent private landowners (Figure 3). Consequently, there is no landscape-level, land-use planning, and due to private ownership, little government oversight of harvesting practices.

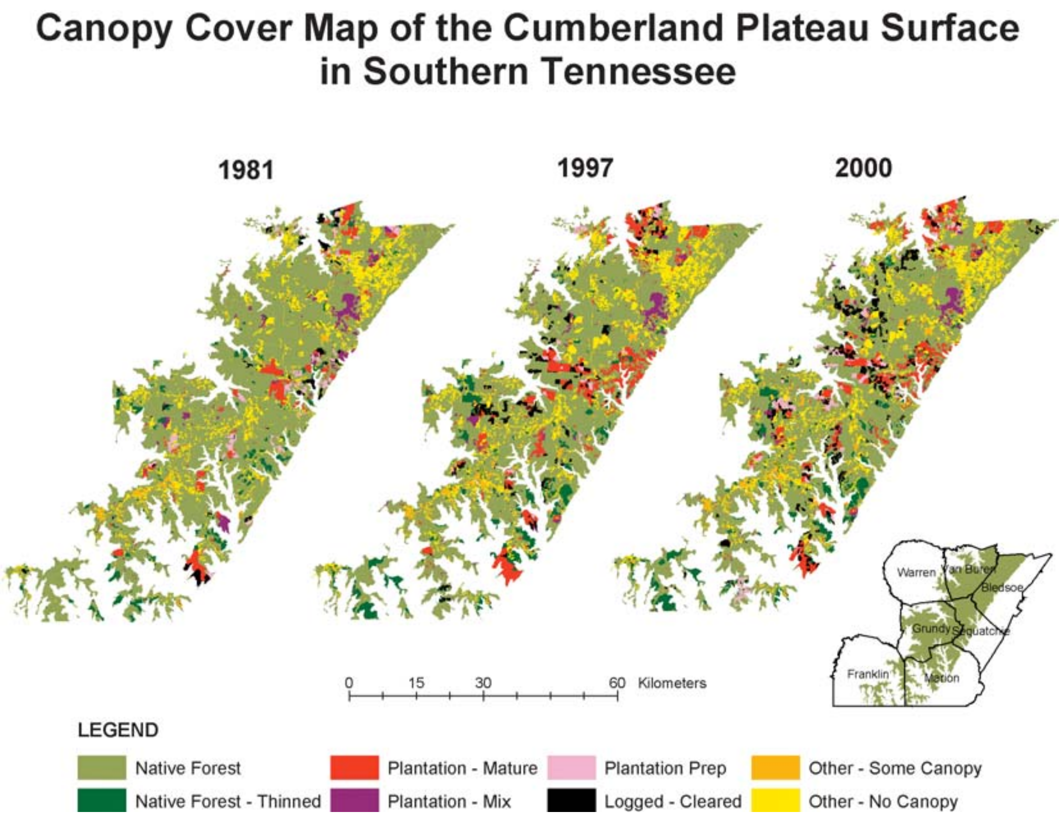

Figure 4. Land-cover change from 1981 to 2000 in the seven-county region of the southern Cumberland Plateau. Images digitized using ERDAS Stereo Analyst software or ArcView 3.2 in 2D from the following sources: 1981–82 NHAP Program Infrared Photography, 1997–99 NAPP Panchromatic Photographs, and 2000 FSA Slides and USGS Enhanced Thematic Mapper Imagery for Aug–Sep 2000 (from Evans et al., 2002; available online at www.lal.sewanee.edu).

2.2. The SAA Demonstration Project

Using aerial and satellite imagery, we created computer-generated maps of land use and forest cover for the SAA study area for the following years: 1981, 1997, and 2000. From these maps we were able to track and document patterns of forest change and conversion between 1981 and 2000 (Figure 4). The following publicly available sets of aerial imagery spanning the study area were obtained for use in this project.

- 1981: Color infrared stereo pair prints from the National High Altitude Photography (NHAP) of the U.S. Geological Survey.
- 1997: Black-and-white stereo pair prints from the National Aerial Photography Program (NAPP) of the U.S. Geological Survey.
- 2000: Small-area format color slides from the Farm Service Agency (FSA) of the U.S. Department of Agriculture. [Landsat Enhanced Thematic

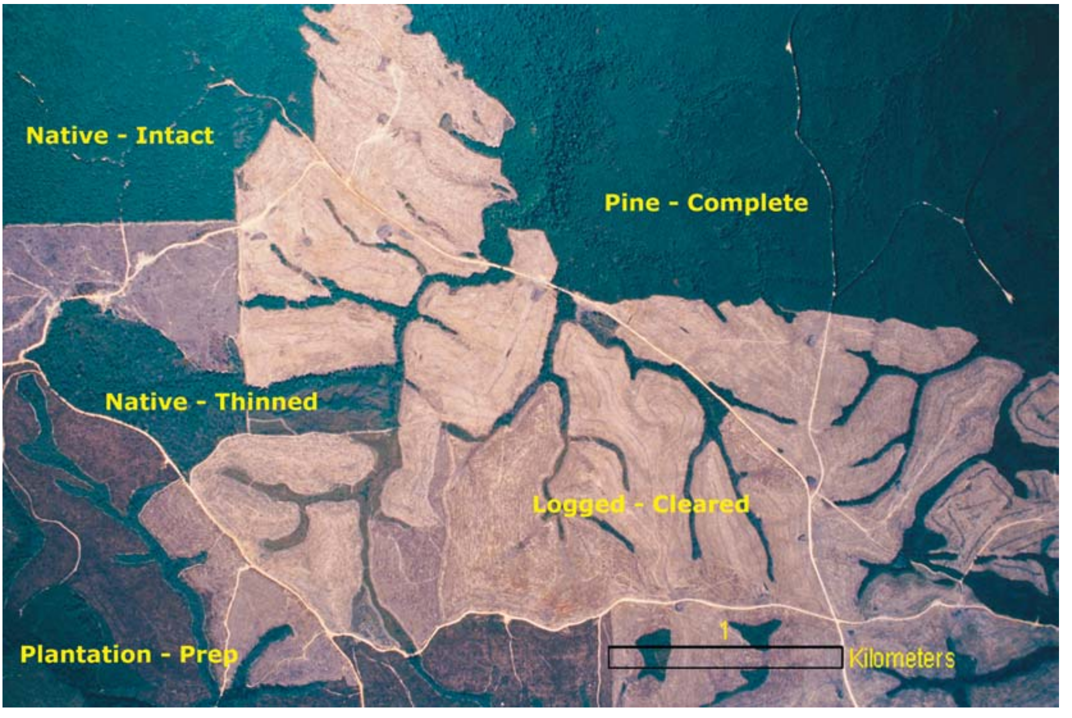

Figure 5. Aerial view from FSA slide demonstrating forest clearings in Grundy County, TN (southern Cumberland Plateau) in the fall of 2000 and representative land-use classification calls. FSA slides are available as 35-mm transparencies (resolution 1 pixel = 3 m) and are one of the six imagery sources used to map forest change by the SAA Forestry Demonstration Project.

Mapper + (ETM+) satellite imagery was also in conjunction with the FSA slides.]

The NHAP, NAPP, and FSA aerial imagery were scanned and stored digitally. Stereo block models of the NHAP and NAPP imagery were created using ERDAS Orthobase software so that the imagery would be orthorectified for use in softcopy stereo. FSA slides were individually rectified using ERDAS Imagine software. A base layer was created for 1997 where land-cover calls and delineations were conducted using 3D visualization of NAPP aerial photography on the computer using ERDAS Stereo Analyst. Land-cover differences between 1997 and 1981 were also digitized using softcopy 3D visualization of the NHAP aerial photography. Land-cover differences between 1997 and 2000 were conducted in 2D using ESRI ARCVIEW but employed both satellite imagery and the high-resolution FSA color photography. The major cover categories depicted in these maps included 1) native forest with an intact canopy, 2) silviculturally thinned native forest, 3) areas that had been recently logged and cleared of trees, 4) pine plantation, and 5) areas with partial or no tree canopy in predominately agricultural or residential/urban use (Figure 5). Land-use designations made from NAPP, NHAP, and FSA aerial photographs were verified using a series of ground photographs taken at selected locations within the coverage area and through the development of cross-categorization matrices. Post hoc error assessment involved a comparison of final forest-cover calls with satellite vegetation indices and an independent post hoc field classification check (Evans et al., 2002).

2.3. Benthic macroinvertebrate surveys

To assess the potential effects of forest changes on watershed integrity and water quality on the Cumberland Plateau, we conducted field samples of stream-dwelling macroinvertebrates across the seven-county study area. Our goals were 1) to provide a preliminary assessment of patterns of diversity and abundance of these organisms in different habitats, and 2) to evaluate the feasibility of using focused and inexpensive field sampling to assess changes in biodiversity. Aquatic macroinvetebrates were sampled during the spring (March and April in 2000 and 2001) and summer (June and July in 2001). The springtime samples were taken in the southern portion of the study area (Franklin and Marion Counties) and were conducted in first-order streams whose watersheds were either entirely covered by native forest, or were entirely clearcut (except for streamside management zone buffers) within the past 12 months. The summertime samples were taken from seven undisturbed (native forest) and seven disturbed (clearcut or heavily logged) streams across the study area.

Sample collection and invertebrate identification used a reduced form of the rapid bioassessment protocols established by the U.S. Environmental Protection Agency (EPA). Collection and identification of macroinvertebrates was performed by S. and L. Hamilton (Austin Peay State University). Benthic macroinvertebrates were collected using a 500-μm opening mesh D-net [this is a change from the Quality Assurance Project Plan (QAPP) because the streams were too narrow to use the 1.0-m-wide kick-net]. Collection followed the field-sampling procedures for a single habitat established in the EPA's Rapid Bioassessment Protocols (RBPs) for Use in Wadeable Streams and Rivers. At each site, a 100-m sampling reach was delineated and five riffle areas within the reach were sampled (1 m^2 each) for macroinvertebrates. Prior to sampling, the habitat was evaluated on the EPA protocols. We also measured the total width of the streamside management zone (SMZ) buffers of uncut forest along each stream. This measurement was made from the midpoint of each sampling transect. All five samples from each reach were combined and preserved in 70% ethanol.

Once samples had been transported to the laboratory, they were processed for subsampling and macroinvertebrate identification. Subsampling followed the procedure established in the EPA RBPs. A 300-organism subsample was removed from each sample and preserved separately from the remainder of the sample. Organisms were identified to the genus level by using a variety of taxonomic keys, including Edmunds et al. (Edmunds et al., 1976), Harris et al. (Harris et al., 1987), Stehr (Stehr, 1987), Stewart and Stark (Stewart and Stark, 1988), Merritt and Cummins (Merritt and Cummins, 1996), and McCafferty (McCafferty, 1998).

We used appendix B of Barbour et al. (Barbour et al., 1999) to assign tolerance values and feeding groups to each taxon in our sample. We used values for the

Southeast, but when tolerance/feeding group behavior was not included for the Southeast we used Midwest values, and when these were missing we excluded the taxon from the analysis. To examine effects of land-use change on macroinvertebrate communities independent of the effects on abundance (i.e., based on presence/absence) we created a Normalized Differenced Benthic Index (NDBI) with the following structure:

$$\frac{\#\text{Intolerant} - \#\text{Tolerant}}{\#\text{Tolerant} + \#\text{Intolerant}}.$$

This index has a range between -1 with totality of pollution-tolerant inverts and $+1$ being a totality of intolerant inverts. Zero signifies balance between the two. Thus, a higher index value indicates higher water quality. Invertebrate taxa were assigned as tolerant if the tolerance score from the RBPs was greater than 5 and intolerant if the score was less than 5 (this corresponds to the mean tolerance for the study area, which was 5.3).

2.4. Bird community response to forest change

We conducted field surveys of breeding birds in the major habitat classes on the Cumberland Plateau in an effort to examine the effects of land-use change on breeding-bird habitat. The surveys used 5-min point counts arranged in transects. All counts were conducted by one observer to eliminate among-observer bias. Each transect had 10 counts (unless the size of the habitat patch was too small to fit 10 counts). Counts within each transect were spaced 200 m apart, and all counts were conducted within 4 h of sunrise. At each count, all birds seen or heard were recorded, and the distance between the observer and each bird was estimated using a rangefinder. Each transect was contained within just one habitat class (defined below) and counts were conducted at least 50 m from the edge of each habitat class (usually more than 150 m).

Each transect was located in one of six distinct habitat classes: (i) mature native hardwood forests with no evidence of recent logging; (i) thinned native forests with 50%–90% canopy removed, but not subjected to site preparation; (iii) mature loblolly plantations with closed canopy; (iv) middle-aged loblolly (0.5–2.0-m height, no closed canopy); (v) early loblolly ($<$ 0.5-m height and little vegetation between seedlings) pine plantations; and (vi) residential–rural areas.

We quantified species richness at three spatial scales: at the level of each point count, at the level of each transect, and at the level of each habitat class. For the two larger scales we compared richness using rarefaction curves. These curves plot species richness against the number of individual bird samples, thus allowing us to compare species richness among habitat types while controlling for the confounding effect of sampling effort and bird density (Gotelli and Graves, 1996). For the analysis at the level of individual points there were not enough observations per point to construct meaningful rarefaction curves, so we compared the number of species at each point.

We measured evenness by calculating the probability of interspecific encounter for each transect and for data pooled within each habitat class. This measure of evenness controls for variation in the number of individuals sampled. Evenness is a measure of the extent to which the bird community is dominated by a few abundant species. Such domination results in low evenness; a high evenness community has relatively equal abundance of most species.

We used three methods to check the robustness of our conclusions to the variation in the technique used to calculate bird density (which is birds per area, regardless of species). First, we produced an index of abundance by summing the numbers of birds detected at each point, regardless of distance from the point. Second, we calculated per-point densities by dividing the number of birds detected within 50 m of each point by 0.79 ha (the area of the 50-m circle). Third, we used detection functions (estimates of how the probability of detecting a bird changes with distance from the point) to calculate densities using DISTANCE 3.5 (Thomas et al., 1998).

We visualized differences in bird communities with detrended correspondence analysis (DCA). This analysis is an ordination technique that uses reciprocal averaging of species-abundance data to place samples (e.g., transects or points) in an ordination space defined by a small number of dimensions. The detrended analysis places samples in the ordination space such that distances between points are equivalent across the entire ordination space.

To put our results into a larger context we produced indexes using Partners in Flight (PIF) priority scores (Carter et al., 2000) for all species and habitats in our samples. PIF is a planning agency, made up of scientists from government agencies, universities, and nonprofit organizations that scores all bird species in North America for their conservation priority. High-scoring birds are those whose populations are in steep decline or who face rapid habitat loss. These scores therefore put our results into a continent-wide perspective. First, we calculated the number of individuals of each species detected within 50 m of count centers per transect for all habitat classes (a measure of relative abundance), then multiplied this by priority scores derived from PIF. This procedure weighs all PIF scores by relative abundance. Second, we examined PIF scores unweighed by any measure of abundance. We also categorized species according to PIF priority ranks (extremely high priority, high priority, moderate priority) and quantified the numbers of species from each habitat class present in each of these PIF priority classes.

We used the year 2000 land-cover GIS layers to calculate landscape metrics associated with each bird sampling transect. We buffered each transect at 150 and 1000 m and computed landscape metrics using Patch Analyst 2.2. We then used linear regression to compare each landscape metric to bird species richness measured at the level of the whole transect.

2.5. The impact of conversion on soil calcium stores

Although soil nitrogen and phosphorus availability limit forest productivity worldwide, several studies have established the potential for long-term soil calcium depletion in the eastern United States as a result of short rotation timber harvests (Adams et al., 2000; Huntington, 2000; Wilson and Grigal, 1995; Federer et al., 1989). We assessed the impact of hardwood-to-pine conversion on calcium stores and cycling by (i) comparing aboveground Ca stores in an oak–hickory forest and a mature pine stand, (ii) quantifying Ca export with three first-rotation pine harvests, and (iii) analyzing foliar, forest floor, and mineral soil samples collected from nine sites representing a chronosequence of hardwood to second-rotation pine plantation. We studied three specific stages in this continuum: (i) mature native oak-mixed hardwood forest, (ii) recently harvested mature (>30 yr) first-rotation pine, and (iii) early (3 yr old) second-rotation pine plantations. Each land use was replicated on three noncontiguous upland sites located within a 24 000 ha area in Franklin County, Tennessee (Figure 2). The hardwood forest stands sampled were representative of the mixed oak–hickory complex native to the upland areas in this region described by Kelly (Kelly, 1979). The first-rotation-harvested pine sites were originally converted from native hardwood to loblolly pine plantations in the early 1960s, cleared of pine in 2000, and left unburned to regenerate naturally. On these sites, harvests included only merchantable stems, as is the operational practice. The size of these clearcuts ranged from 14 to 20 ha. The young second-rotation sites were originally hardwood forest that had been cleared in the mid-1960s and replanted in loblolly pine. Merchantable stems were harvested from these sites in 1999, followed by site preparation that included burning slash, replanting with loblolly and shortleaf (*P. echinata*) pine seedlings, and herbicide application. The size of these harvests ranged from 20 to 80 ha.

On each of the land-use sites, three composite samples (each composed of five randomly collected subsamples) of forest floor (0.25 m^2), surface mineral soil (0–15 cm), and mid-to-upper-canopy foliar biomass were collected. The soil, plant, and litter samples were dried at 60°C, weighed, and sent to A & L Analytical Laboratories, Inc. (Memphis, Tennessee) for standard soil and tissue nutrient concentration analyses. Total soil concentrations of Ca were analyzed using inductively coupled argon plasma (ICAP) spectroscopy after a perchloric acid digest conducted in our lab (USEPA 1997). Calcium stores in the surface soil and forest floor were calculated as the product of mass (Mg ha^{-1}) and Ca concentration (%). Because the forest floor of the mature pine sites was considerably disturbed following the timber harvest, we used the samples taken from surrounding similar-aged unharvested pine to estimate forest floor mass on the harvested sites. On the hardwood and pine sites, nine wood cores were collected using an increment borer and were then dry ashed prior to elemental assay using ICAP. Bole Ca concentrations of wood cores were multiplied by the dry mass of pine and hardwood stems removed from each first-rotation-harvested site.

On one hardwood site and one mature pine site, we measured the diameter at breast height (dbh) of all trees in three 0.10-ha plots. These measurements were used in allometric equations developed by Harris et al. (Harris et al., 1973) and Pastor et al. (Pastor et al., 1984) to estimate aboveground biomass of boles plus branches and leaves or needles for hardwoods and mature pine plantations. Aboveground calcium stores on these two sites were estimated as the product of dry biomass and Ca concentrations in bole, branch, and leaf or needle tissues. Surface soil (0–15 cm) Ca mass was calculated using mean bulk densities for total Ca concentrations for hardwood and first-rotation pine sites.

After determining that replicate sites within a land use did not differ statistically, we used single-factor analysis of variances (ANOVAs) to determine if mean concentrations and contents of soil, forest floor, and foliar Ca differed among the three land uses (hardwood forest, mature first-rotation-harvested pine, and early second-rotation pine; $n = 3$ sites per land use).

2.6. Socioeconomic drivers of land-use change

The first step taken in our attempt to understand the socioeconomic drivers behind forest conservation in the region involved studying the impacts of the Tennessee Forest Greenbelt law. Like most states, Tennessee offers landowners a use value taxation program that permits enrollees to have the property taxed on the basis of its value as agricultural or timber land as opposed to its market value. In this way the state hopes that landowners will maintain land as farm or forest as opposed to developing it or selling to a developer. In Tennessee's case, the Greenbelt programs are voluntary. We administered mail surveys to landowners in a single county of the SAA and analyzed tax data from the county tax assessors. Our questionnaires attempted to uncover landowners' motives for holding onto land and their attitude toward the Greenbelt Program, plans for harvesting timber and/or developing their land, and general attitudes toward the environment and government policies. The tax data were analyzed to determine how much the program raised landowners' reservation prices (minimum prices at which they would sell) for their land. Under varying assumptions about the (monetized) intangible values they derived from holding onto land and about rates of land appreciation, reservation prices were compared with market values to determine 1) whether or not the program would affect owners' willingness to convert or to sell to a developer, and 2) how long land would remain forested.

We also digitized the tax maps (for parcels of 4 ha or more) for the surface of the plateau in our seven-county area so as to be able to examine land-ownership patterns based upon data for 2000–01 (Figure 3).

3. Results and discussion

3.1. Forest-cover change

Native forests of the southern United States are currently undergoing dramatic changes due to shifting patterns in land use. In recent years, the creation of industrial pine plantations has emerged as a dominant force of change and has been predicted to be a major cause of native forest loss in the future (Wear and Greis, 2001). We found that the recent accelerated conversion of hardwood forest to pine monocultures has resulted in the massive alteration of habitat at the landscape level on the Cumberland Plateau in Tennessee (Evans et al., 2002).

We found that 26 592 ha (14%) of native forest cover have been lost since 1981 and that most (74%) of this loss was caused by conversion of native forests to plantations consisting of nonnative loblolly pine. The rate of conversion from native forest to pine plantation doubled during the last 3 years of the study period, from 1220 ha yr^{-1} between 1981 and 1997 to 2358 ha yr^{-1} between 1997 and 2000.

The total area in pine plantation (area with planted trees) increased by 170% (10 104 ha) from 1981 to 2000. Total area under intensive pine plantation management in 2000 (includes a proportion of recently cleared area plus area with

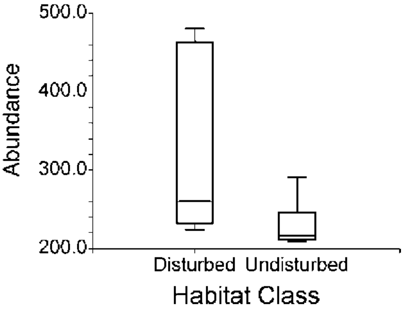

Figure 6. Macroinvertebrate abundance per sample in disturbed and undisturbed streams in the seven-county study region on the southern Cumberland Plateau. Boxplots indicate medians and interquartile ranges (from Evans et al., 2002; available online at www.lal.sewanee.edu).

planted trees) was determined to be 35 724 ha. Total area of native forest converted to agriculture, residential, and other nonsilvicultural uses increased by 18% between 1981 and 2000, accounting for only 26% of hardwood conversion.

About 80% of all newly created pine plantations that appeared in the study area between 1981 and 2000 were derived from either intact or thinned native forests. Less than 3% were derived from lands associated with agriculture. Between 1981 and 2000, most existing or recently converted pine plantations remained as pine plantations and did not transition to other uses.

Pine conversion activity was highly clustered, causing a concentration of impact in certain counties and watersheds. All counties showed a net loss of native forest, with Van Buren County being the highest at 18% (6427 ha). From 1997 to 2000, 90% of all native forest removal resulted from clearings that were greater than 16 ha in size [Forest Stewardship Council (FSC) certification limit]. In addition, 70% of this native forest removal resulted from clearings that were greater than 49 ha in size (Sustainable Forestry Initiative certification average clearcut size limit).

3.2. Macroinvertebrate surveys

Disturbed sites had a statistically significant higher abundance of macroinvertebrates (Mann–Whitney *U* Test: $Z = 2.044$, $p = 0.04$, $n = 14$; Figure 6). The NDBI index was significantly higher for undisturbed than disturbed sites (ANOVA: $F = 16.23$, $p = 0.0017$, power = 0.96; Kruskal–Wallis chi-square = 7.9, $p = 0.005$; Figure 7). The width of SMZ in logged areas also had a significant effect on NDBI, independent of the disturbed/undisturbed effect [linear regression of disturbance against NBDI with SMZ width as interaction term: $p = 0.0003$, $R^2 = 0.64$, F ratio = 24, coefficient = 0.013 (this is approximately a 1.3% increase in NDBI for every meter of SMZ)].

The finding that overall abundance of macroinvertebrates was higher in disturbed streams is consistent with previous research. For example, Kedzierski and Smock (Kedzierski and Smock, 2001) studied streams in Virginia and found that whole-stream annual macroinvertebrate production was greater in sections of

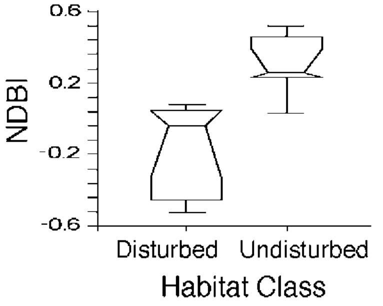

Figure 7. NDBI for disturbed and undisturbed watershed sites in the seven-county study region on the southern Cumberland Plateau. Higher values indicate higher water quality. (Nonoverlapping notches of the two boxplots indicate that medians are significantly different at the 0.05 level.)

streams that were surrounded by logging activities than in undisturbed sections of stream. Likewise, Stone and Wallace (Stone and Wallace, 1998) found higher benthic invertebrate abundance in disturbed streams. This increased productivity is likely due to increased sediment in the streams (Kedzierski and Smock, 2001). We did find more sand in streams from disturbed sites, which suggests that these streams have received more sediment than undisturbed streams.

The NDBI indicated that the invertebrate community on undisturbed sites had a higher proportion of intolerant taxa than did the disturbed sites that were dominated by more tolerant taxa. NDBI also increased with SMZ width. The SMZ widths in our analysis varied from 16.7 to 60 m (total width; stream-to-edge width would be half this amount). Some streams in our study area were not buffered with any SMZs, but macroinvertebrate samples were not taken from these streams. Our data indicates both that SMZs help provide increased water quality, and that some SMZs in our study area may be too narrow to provide maximal protection. Further analysis with a larger sample size would be required to determine exactly which SMZ width offers maximal protection of water quality. Watersheds that had any logging or pine plantations in them had lower NDBI values than watersheds without these activities. This data suggests that water quality at the disturbed sites was lower than at the undisturbed sites.

3.3. Bird community response

The field surveys included 503 individual point counts in 52 transects, and 82 species were detected. The six habitat classes differed significantly in species richness. At the scales of habitat classes and transects, residential–rural areas had highest richness, followed by thinned native forests, then native forests, then all age classes of pine plantation. All ages of pine plantations had lower species richness and evenness than did oak–hickory forests (Figure 8). Plantations had either similar beta diversity or slightly lower beta diversity than did oak–hickory forests.

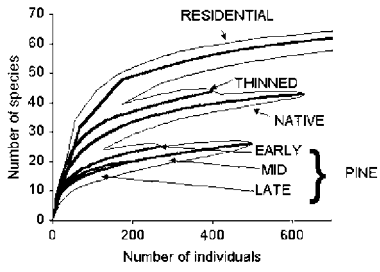

Figure 8. Bird species richness rarefaction curves calculated at the scale of six habitat classes (native mature forest; native thinned forest; early, mid, and late loblolly pine plantations; and residential–rural areas) on the southern Cumberland Plateau. Thick lines indicate the mean richness at each abundance, thin lines indicate 95% confidence intervals (from Evans et al. 2002; available online at www.lal.sewanee.edu).

Abundance of birds in plantations was either lower or the same as oak–hickory forests. Plantations had fewer cavity- and tree-nesting birds, a loss of neotropical migrant birds proportional to that of oak–hickory forests, and an increase in birds that specialize on early successional habitats.

The bird community found in residential–rural areas had higher species richness, evenness, beta diversity, and abundance than that found in oak–hickory forests. Residential–rural areas provided habitat for some birds that occurred nowhere else on the landscape, but many forest-dwelling species were less abundant in this habitat class. Thinned forests had higher richness, evenness, and abundance of birds than did mature oak–hickory forests. Residential–rural areas obtained the highest PIF scores, followed by thinned forests, oak–hickory forests, then pine plantations.

We found very high species richness in the native oak–hickory forests (43 species within 50 m of count centers). When compared to other studies of birds in southern forests, the bird community in our region had by far the highest species richness, with only the Great Smoky Mountains coming close. The direction of landscape effects on breeding-bird richness depended on the spatial scale at which we calculated landscape metrics. At a small scale, increases in edginess and fragmentation increased bird diversity, but larger-scale edginess and fragmentation were associated with decreased bird diversity.

3.4. Impact on upland calcium stores

Soils collected from all nine sites were all classified as sandy loams, with neither proportions of clay nor sand differing among land uses (Table 1). Mean soil pH was equally low on all three land uses. Soil bulk density was lower on the upland forest sites, indicating greater soil compaction at the harvested and second-rotation pine sites. However, these higher bulk densities did not render surface soil total calcium stocks significantly different from those in hardwood stands (Tables 1 and 2). Predictably, forest floor mass was lowest on the second-rotation pine stands that were burned prior to replanting. However, soil organic matter was greater on the

Table 1. Soil properties of native upland oak–mixed hardwood forest, mature (>30 yr) first-rotation harvested pine and early (3 yr) second-rotation pine plantations on the Cumberland Plateau. Data are means ± 1 standard error ($n = 3$ noncontiguous upland sites per land use).

	Upland mixed oak–hickory forest	Harvested pine; sites unburned	Three-year second-rotation pine; site burned	ANOVA p value
Sand (%)	42 ± 1	50 ± 3	44 ± 1	0.394
Clay (%)	27 ± 1	22 ± 3	24 ± 1	0.442
PH	4.6 ± 0.1	4.6 ± 0.1	4.7 ± 0.4	0.791
OM%	2.9 ± 0.4	1.3 ± 0.2	4.7 ± 0.2	0.0001
CEC	7.8 ± 1.2	6.2 ± 0.1	5.3 ± 0.5	0.051
Ext. P (mg kg^{-1})	9.8 ± 2.0	6.6 ± 0.9	4.3 ± 0.9	0.03
Soil bulk density (g cm^{-3})	1.0 ± 0.1	1.3 ± 0.1	1.3 ± 0.1	0.023
Forest floor (Mg ha^{-1})	15.1 ± 2.8	23.0 ± 2.8	4.9 ± 1.1	0.005

Ext. P = Mehlich III extractable P.

second-rotation pine sites, perhaps due to incomplete burning of slash during site preparation. Although cation exchange capacity was higher in upland hardwood soils, neither exchangeable nor total Ca differed among land uses (Table 2). In particular, exchangeable Ca on the second-rotation pine sites did not differ from the older sites, despite the fact that the soils on these plantations had received a pulse of nutrients 3 years earlier when postlogging slash was burned prior to planting seedlings. Mean total soil calcium was low across land uses and did not differ from the exchangeable Ca pool on any of the sites (Table 2).

Calcium concentrations in hardwood leaves and forest floor were significantly higher than those in first- and second-rotation pine needles and litter (Table 2). Our estimates of aboveground Ca stores in a hardwood and unharvested pine stand were similar to values (307–1463 kg Ca ha^{-1}) reported in other studies (Adams et al., 2000; Kelly, 1979) and also demonstrated higher Ca retention in hardwood biomass (Table 3). While total aboveground biomass appeared similar between the hardwood and mature (unharvested) pine stands, Ca contents in foliage and wood

Table 2. Calcium concentrations and stores (kg ha^{-1}) in mineral surface soil (0–15 cm), forest floor, and tree tissues collected from a chronosequence of upland hardwood to early second-rotation pine plantations. Data are means ± 1 standard error ($n = 3$ noncontiguous sites).

	Upland mixed oak–hickory forest	Harvested mature pine; sites unburned	Three-year second-rotation pine; sites burned	ANOVA p value
Exchange soil Ca (mg kg^{-1})	447 ± 295	187 ± 14	390 ± 146	0.15
Total soil Ca (mg kg^{-1})	435 ± 155	331 ± 52	640 ± 248	0.21
Soil Ca store (kg ha^{-1})	652 ± 117	646 ± 40	1282 ± 495	0.35
Forest floor Ca (%)	1.48 ± 0.29	0.64 ± 0.12	0.50 ± 1.5	0.02
Forest floor Ca store (kg ha^{-1})	224 ± 42	147 ± 18	24.7 ± 5.4	0.005
Foliar Ca (%)	1.01 ± 0.22*	0.51 ± 0.04	0.30 ± 0.04	0.03
Bole Ca (%)	0.23 ± 0.08*	0.07 ± 0.01	No data	0.14

*Includes Kuers unpublished data from Split Creek Watershed site.

Table 3. Estimated Ca stores in aboveground biomass and surface soil (0–15 cm) of an upland oak–mixed hardwood forest and an adjacent 30-yr-old loblolly plantation on the Cumberland Plateau.

	Dry biomass (Mg ha^{-1})		Ca store (kg ha^{-1})	
	Hardwood	Pine plantation	Hardwood	Pine plantation
Forest floor	15	23	224	147
Bole + branches*	158	165	363	116
Foliage	5	4	46	21
Total aboveground	178	192	633	284
Surface soil**	1500	1950	652	646
Total system (aboveground + soil)	1678	2142	1285	930

*Bole + branches and foliar mass estimated using dbh measurements in allometric equations developed by Harris et al. (Harris et al., 1973) and Pastor et al. (Pastor et al., 1983) for oak–hickory and white pine (*P. strobus*) stands, respectively.
**Hardwood and pine surface soil (0–15 cm) mass based upon mean bulk densities of 1.0 and 1.3 g cm^{-3} (as in Table 1).

stores were twice that of first-rotation pine due to higher Ca concentrations in hardwood tissues. Thus, despite similar soil contents of both exchangeable and total Ca, our estimates of aboveground Ca stores indicate greater Ca storage in native upland hardwood stands than in mature first-rotation pine (Table 3). These data also suggest a total system Ca loss as a result of hardwood-to-pine conversion that is not evident from our soil data alone. The difference in system Ca stores between hardwood and early second-rotation pine would presumably be even greater because of considerably lower aboveground biomass in the 2-yr-old pine saplings compared to mature hardwood trees.

Our data demonstrate that following hardwood-to-pine conversion, the Ca distribution within the system shifted from a concentration in aboveground tissues and forest floor in hardwood to largely mineral soil stores in second-rotation pine. For example, mean forest floor Ca contents in hardwoods represented nearly 35% of the Ca stored in the top 15 cm of soil, while in mature pine stands, the organic horizon stored only 22% of the Ca contained in the surface mineral soil. In the second-rotation sites, forest floor represented less than 2% of the surface soil Ca store (Table 2). With similar concentrations of total and exchangeable soil Ca soils underlying all three land uses, and significantly less Ca in the second-rotation forest floor, it seems clear that soil Ca in the second-rotation pine surface soil may be insufficient to support future aboveground stores comparable to those of the mature first-rotation pine. The estimated Ca content in aboveground biomass of mature first-rotation pine was nearly 300 kg Ca ha^{-1}, representing roughly 75% of the surface soil stock of exchangeable Ca in the early second-rotation pine (Tables 2 and 3). The lack of difference in surface soil exchangeable Ca between the second-rotation pine and mature sites suggests that any post–site preparation pulse of calcium from the ash of burned slash is no longer evident in surface soil Ca storage, although residual slash from an incomplete burn may provide a source of mineralizable Ca to the young second-rotation stand. The fact that mean total and exchangeable soil Ca pools do not differ on any land-use site suggests that there is

Table 4. Calcium removal with merchantable stem harvest from pine plantations and adjacent oak–mixed forests on upland sites of the Cumberland Plateau. Mass estimates based upon records from each harvested site.

	Hardwood		Pine		Total	
Harvest site	Dry mass Mg ha^{-1}	Ca kg ha^{-1}	Dry mass Mg ha^{-1}	Ca kg ha^{-1}	Dry mass Mg ha^{-1}	Ca export kg ha^{-1}
Armfield Bluff	89	222	101	71	190	292
Equestrian	3	7	139	97	142	104
Smith Tract	28	71	159	112	188	182

Calcium contents (kg ha^{-1}) derived using hardwood and pine Ca concentrations of 0.24% and 0.07%, respectively.

little inorganic Ca to buffer the exchangeable pool as it is taken up by plants or leached down the soil profile.

Huntington et al. (Huntington et al., 2000) predicted long-term soil Ca depletion after two to three pine rotations, or 120 years, in the southeastern Unites States because of little weatherable Ca in soil parent material and negligible Ca inputs from atmospheric deposition. While the total dry mass harvested from the mature first-rotation pine was similar to values reported in the southeastern United States (Huntington et al., 2000), Ca export from combined pine and hardwood stem removal was lower, ranging from 104 to 292 kg Ca ha^{-1} (Table 4). In contrast, Ca removal from only pine stem harvests on our sites appeared higher than that reported by Thompson et al. (Thompson et al., 1986) for merchantable stems of loblolly (52 kg Ca ha^{-1}) in growing in Ultisols of the Piedmont region of North Carolina. Numerous studies demonstrate that whole tree harvests double or nearly triple Ca removal (Federer et al., 1989; Thompson et al., 1986; Johnson et al., 1988) with reported removal rates ranging from 200 to 1100 kg Ca ha^{-1} for oak–hickory forests (Huntington et al., 2000; Johnson and Van Hook, 1989). Despite the fact that only merchantable stems were harvested from our three first-rotation sites, our highest estimate of total Ca export represented roughly 40%–45% of total Ca stores in both hardwood and pine surface soil, and 20%–30% of calcium stored in surface soil and aboveground biomass combined (Tables 2, 3, 4). The combined forest floor and surface soil Ca stores in second-rotation pine appear just large enough to support our estimate of mature standing pine biomass and Ca storage, and may be insufficient to support regrowth of oak–hickory-dominated forest in the near future.

3.5. Socioeconomic drivers of forest change

The results of the initial one-county study, and subsequent three-county study (Brockett et al., 2003; Williams et al. 2003) indicated that landowners received tax relief for doing what they would have done in the absence of an incentive program. In summary, few landowners reported that Tennessee's Greenbelt Program affected their behavior, while the tax analyses showed that most recipients of the tax breaks would not have converted their forest in any case because land prices were too low for development. Those owners in areas with the highest land values would have found it economically advantageous to refrain from converting as a result of the program only until higher land prices overwhelmed the benefits of staying in the program. Consequently, at best the program delayed forest conversion slightly for a few years in high-priced zones of the area (Brockett et al., 2003).

Our current project examines the drivers of the land-use change measured by the SAA Demonstration Project. As it is in its initial phases, we have concentrated thus far on determining the land ownership patterns in the seven-county study area. Preliminary results indicate that private owners dominate this region, owning 93% of the land (Figure 3). Individuals own about 53% of the surface area, with timber companies and other businesses owning less than 20% each. In the entire study area, 10% of owners own 78% of the land while the smallest 10% of owners own only 0.6% of the land.

We also attempted to identify where local landowners reside. Our analyses show that 43% of the land is held by owners with local mailing addresses (addresses within the county of the parcel or an adjoining county). People or firms residing in Tennessee but not locally hold 19% of the land, while owners residing outside of Tennessee control the remaining 38%. Consequently, policies aimed at effecting changes in land use, land management, or environmental quality in the region must take into account that they must address the motivations and concerns of large landowners who frequently are timber companies or other businesses, and owners who often reside outside of the local area.

4. Summary and conclusions

The Small Area Assessment and related studies have highlighted important subregional changes in the landscape of the southern Cumberland Plateau. Our research demonstrates that large-scale, hardwood-to-pine conversion occurred in this region over the past 20 years but was unaccounted for by state forest inventories. We found that 26 592 ha (14%) of native forest cover was lost between 1981 and 2000, occurring largely as a result of native forest conversion to pine plantations, with a doubling in the rate of conversion during the last 3 years of the study period. Pine conversion activity was highly clustered, causing a concentration of impact in certain counties and watersheds. From 1997 to 2000, 70% of forest removal on the Cumberland Plateau resulted from clearcuts that were greater than 120 acres (48.6 ha) in size.

Water quality in streams, as measured by the abundance of critical macroinvertebrates, was significantly lower in recently logged sites than in undisturbed native forest. However, the index of water quality generally improved as streamside management zone width increased, highlighting the importance of wide vegetated buffers in maintaining water quality. These findings are important because this area serves as the headwaters to some of the most biologically diverse, freshwater stream systems in the world.

Neither pine plantations nor residential areas can provide habitat for most of the native birds of the Cumberland Plateau. The transformation of the forest has therefore reduced bird diversity in our region, particularly in pine plantations. This erosion of the quality and quantity of breeding-bird habitat is particularly problematic for many of these bird species because the Cumberland Plateau offered one of the few relatively unfragmented areas of forest in our region. Populations of many forest-dwelling birds require such core areas in order to persist. If current trends of forest change continue, such persistence may be compromised.

Our comparison of aboveground and surface soil Ca stores in native hardwood and mature pine on converted hardwood sites demonstrates a substantial loss of system Ca within one pine rotation. Despite only moderate Ca removal with merchantable stems of first-rotation pine, harvest export of Ca from our sites represented a large proportion of both the exchangeable and total surface soil pool of Ca. The combined forest floor and surface soil Ca stores in second-rotation pine appeared just large enough to support our estimate of mature standing pine biomass and Ca storage, and may be insufficient to support the vigorous regrowth of a mature oak–hickory forest in the near future. These findings highlight the need to study the effects of repeated pine rotations and harvests not only on productivity and health of subsequent plantations, but also on the structure, function, and composition of regenerating secondary native hardwood ecosystems.

The socioeconomic research indicates that Tennessee's major forest conservation incentive program, the Forest Greenbelt Program, at best only delays forest conversion for a few years while subsidizing landowners who would not have converted their land in the absence of the program. Because of the highly skewed land distribution in the region, policies aimed at effecting changes in land use, land management, or environmental quality in the region must take into account the motivations and concerns of absentee owners and of large landowners who frequently are timber companies or other businesses. We are currently developing a spatial socioeconomic model of change in land use/land cover (LULC) for the southern Cumberland Plateau for the period 1980–2000. We will use this model to simulate future impacts of various trends in, for example, timber prices or interest rates, and explore policy options, such as reforming the Greenbelt Program or giving payments to landowners for improving water quality (environmental service payments).

By detecting significant forest change previously unaccounted for by official statewide inventories, this project clearly demonstrates the critical role that more detailed studies conducted at smaller scales play in the analysis of land-use change at larger scales. Moreover, this type of interdisciplinary work highlights a critical niche that smaller institutions may fill by providing finescaled data that can be used to validate the findings of studies that examine land-use change over much larger scales. Understanding the economic and social factors that drive land-use change in these smaller landscapes will help improve natural resource policy decision making at the state level as well as uncover root causes of land-use change that may be of importance at larger scales.

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