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Factors Related to Spatial Patterns of Rural Land Fragmentation in Texas

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Abstract Fragmentation of family-owned farms and ranches has been identified as the greatest single threat to wildlife habitat, water supply, and the long-term viability of agriculture in Texas. However, an integrative framework for insights into the pathways of land use change has been lacking. The specific objectives of the study are to test the hypotheses that the nonagricultural value (NAV) of rural land is a reliable indicator of trends in land fragmentation and that NAV in Texas is spatially correlated with population density, and to explore the idea that recent changes in property size patterns are better represented by a categorical model than by one that reflects incremental changes. We propose that the State-and-Transition model, developed to describe the dynamics of semi-arid ecosystems, provides an appropriate conceptual framework for characterizing categorical shifts in rural property patterns. Results suggest that changes in population density are spatially correlated with NAV and farm size, and that rural property size is spatially correlated with changes in NAV. With increasing NAV, the proportion of large properties tends to decrease while the area represented by small properties tends to increase. Although a correlation exists between NAV and population density, it is the trend in NAV that appears to be a stronger predictor of land fragmentation. The empirical relationships established herein, viewed within the conceptual framework of the State-and-Transition model, can provide a useful tool for evaluating land use policies for maintaining critical ecosystem services delivered from privately owned land in private land states, such as Texas.

Keywords Conservation easements · Land fragmentation · Landowner cooperatives · Non-agricultural value · Spatial analysis · State-and-Transition model

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Introduction

Fragmentation of private farms and ranches is becoming an issue of concern in Texas, and across much of the Western United States. The overall decline in rural property sizes appears to be a product of changes in the demand for rural land, driven by regional, social, and economic dynamics, combined with other factors such as declining agricultural returns, environmental regulations, increasing age of rural landowners, and high cost of intergenerational land transfers (Wilkins and others 2000, 2003; Cromartie and Wardwell 1999; McCann 1999; Peterson 1997; Rowan and White 1994).

The net consequence of land fragmentation can vary depending upon land cover and prevailing land use. More decision-makers can result in less integrated land management decisions that may at times lead to degradation in wildlife habitats and biodiversity (Rollins 2000; Fulbright 1997). Similarly, where landowners have rights-of-capture to groundwater (e.g., Texas), an increase in landowners can lead to depletion in water supplies (Wagner and Kreuter 2004). Land fragmentation can also be associated with loss of native plant and animal species, increase of invasive species, increased soil erosion, decline in water quality, loss of agricultural production, and increased cost of public services (Wilkins and others 2003; Collinge 1996). Furthermore, construction of new houses in areas with scenic amenities has led to the rapid transformation of rural landscapes and has destabilized local communities (Theobald 2001; Riebsame and others 1996). Conversely, many landowners purchasing smaller tracts of land tend to place a higher value on wildlife habitats and associated amenity values, and they are more likely to seek land management advice and forgo economic land use in deference to conservation (Sanders 2005).

The negative effects of ownership fragmentation are more likely to be of consequence in areas where privately owned farms and ranches dominate. Such is the case in Texas, where 84% of the land consists of privately owned farms and ranches. With a population of nearly 21 million, the second fastest population growth rate, and 3 of the 10 largest cities, Texas has become the second most populous state in the United States (Murdock and others 2002). This rapid population growth led to the conversion of over 1 million hectares (ha) of rural land to urban uses between 1982 and 1997, with annual conversion from 1992 to 1997 being nearly double that for the prior 10 years (Wilkins and others 2000). Such intensive conversion to nonagricultural land uses has a profound influence on the structure and nature of farms, ranches, and rural land markets (Pope and Goodwin 1984). There has been a marked increase in the number of rural land parcels and a decrease in average parcel size in most Texas counties (Wilkins and others 2000), with more than 80% of rural properties now being smaller than 200 ha (Wilkins and others 2003). Additionally, during the 1990s, the area of land in mid-size ''bread and butter'' farms and ranches (202–809 ha) declined by more than 100,000 ha per year in Texas (Wilkins and others 2000). As a result, the Texas Governor's Task Force on Conservation (2000) identified fragmentation of family-owned farms and ranches as the greatest single threat to wildlife habitat, water supply, and long-term viability of agriculture (Wilkins and others 2003).

The decrease in midsize farms may be attributed to the high per-hectare prices being paid for smaller land parcels and the greater economies of scale afforded by land consolidation. While the average market value of rural land in Texas increased by 2.7% per year since 1992 to \$1,542 ha^{-1} in 2001, the average agricultural value of land grew by 0.4% annually to about \$198 ha^{-1} in 2001 (Wilkins and others 2000). The difference between the two values represents the nonagricultural value (NAV). Some researchers have reported that the NAV of rural land is positively correlated with population density in proximate urban areas (Shi and others 1997). Rapid population growth may thus lead to escalating NAV of rural properties, creating incentives for farm and ranch owners to sell off or subdivide their land for development purposes. Pope (1985) hypothesized that a significant increase in NAV leads to a bimodal distribution in farm size because ownership sizes tend to be either fragmented into smaller properties in response to demand from ''nonproducer'' landowners, or ownership size increases because of consolidation into larger properties by agricultural producers seeking greater economies of scale in response to inflated land values. A bimodal property size distribution has been reported by real estate brokers for some counties in Texas (Pope and Goodwin 1984).

Correlations between property size, land use, and landscape characteristics, such as plant community composition and hydrology, suggest that ecosystem services are being negatively affected by changes in land ownership size in Texas (Wilkins and others 2003). Similarly, a study in the Upper Midwest determined that parcel sizes and changes in parcel sizes are important covariates in the relationship between land use and forest-cover changes (Brown and others 2000). However, ecosystem responses to external forces, such as population-dependent land size and associated land use shifts, are often nonlinear (Westoby and others 1989).

One challenge in studying the effects of land fragmentation is the lack of a conceptual model for describing the dynamics of ownership size distributions under shifting demographic and economic conditions. Standard economic theory does not adequately handle spatial interactions (Shi and others 1997) or discontinuous environmental responses. This has led to a lack of empirical analyses of land fragmentation (Munton and Marsden 1991; Bentley 1987; King and Burton 1982). Here, we borrow the State-and-Transition model from ecology. This is a tool developed to describe a series of relatively stable assemblages of plants (states) that can persist until some environmental threshold condition occurs leading to a relatively rapid transition to another assemblage (Westoby and others 1989). The State-and-Transition model aids in strategically applying management interventions to either avoid such transitions or to stimulate transitions toward a more favored state.

We propose that this model can also provide a conceptual framework for characterizing relatively discrete assemblages of land ownership sizes along with the conditions for transition between such assemblages. By conceptualizing the dynamics of land fragmentation as a set of states and transitions, one can explore those determinants of changing ownership sizes within a framework that may more closely resemble reality. This argument is based on the observation that land fragmentation in a local area is a relatively rapid phenomenon, often resulting in a quantum change rather than an incremental change in property size and associated management objectives. In this case, effective implementation of public policies aimed at avoiding the undesirable consequences of fragmentation would require approaches that vary according to the existing assemblage of property sizes. This in turn requires knowledge about property-size thresholds with regard to management decisions and distribution patterns of property-size categories.

We have three objectives in this study: (1) To test the hypothesis that NAV of rural land is a reliable indicator of trends in land fragmentation, thereby determining whether Pope's (1985) use of economic theory to show how high demand for land can lead to bimodal property size distributions in rural areas is borne out by recent land ownership changes in Texas; (2) To test the hypothesis that NAV of land in Texas is directly correlated with population density, thereby corroborating results of an earlier study by Shi and others (1997) that focused on 13 metropolitan areas in 6 states neighboring Virginia; and (3) To explore the idea that recent shifts in land ownership patterns can be better represented by a categorical model, such as the State-and-Transition model, than by one that reflects a continuum of property sizes and incremental changes in size. First we provide additional information about land fragmentation and a conceptual framework for modeling landownership fragmentation.

Land Fragmentation

Currently, the consequences of increasing development and rural land values on land fragmentation rates are not entirely clear. Previous studies of agricultural land values found that distance of properties to roads and metropolitan areas, changes in county population, and county population density were significant explanatory variables for future development rents (Plantinga and Miller 2001). Furthermore, urban land use density and land value have been found to be positively correlated in some areas (Peiser 1989), but this is not a consistent relationship (Breslaw 1990), some having identified negative or inconsistent relationships between land values and distance to urban centers (Plantinga and Miller 2001). In addition, econometric estimates of land prices that include farm income and returns of alternative investments have been criticized for not accurately reflecting structural changes and other characteristics of agricultural land markets (Gilliland 1988).

Land fragmentation effects are generally considered to be deleterious; however, in some cases smaller property size may lead to positive outcomes (e.g., Alig and Healy 1987; Bentley 1987; Healy 1985; Bradley 1984; Clout 1984; King and Burton 1982). With regard to the negative attributes of land fragmentation, both economic and ecological components are present. For example, spillover effects associated with land fragmentation in urban fringe areas occur when the perception among rural landowners is that agricultural production has no future in their area, and they react by not investing in improvements, shifting to less labor-intensive production, and selling productive farmland (Conklin and Lesher 1977). Such land use conversions can also lead to the use of lower quality land for production at greater economic and environmental costs (Greene and Stager 2001; Nelson 1992). These types of anthropogenic land transformations have led to habitat fragmentation through the conversion of relatively undisturbed landscapes while producing a mosaic of remnant habitat patches surrounded by different land uses. Habitat fragmentation has been called ''the most serious threat to biological diversity and...the primary cause of the present extinction crisis'' (Collinge 1996; Wilcox and Murphy 1985), because it can disrupt dispersal and movements of animals, increase predation, disturb animal social structure, and diminish habitat health by inhibiting events such as migratory grazing and natural fires.

Land use changes that lead to land fragmentation may have global ramifications. The recent Millennium Ecosystem Assessment Synthesis Report revealed that about 60% of the earth's ecosystem services are being used unsustainably (MEASR 2005). Although population-related land use intensification has contributed to short-term gains in economic development, the associated long-term costs of depleting nonmarket ecosystem goods and services are seldom included in economists' benefit–cost analyses of land use change. If continued unabated, land use strategies that do not account for the depletion of such ''natural capital'' (Daily and Ellison 2002; Costanza 1997) will ultimately lead to accelerated loss in biological diversity and ecosystem function and concomitant decrease in human well-being provided by these ecosystems (Conroy and others 2003). Consequently, increased economic costs of maintaining ecosystem services have been used as a primary argument for preventing land fragmentation (Bentley 1987).

Many benefits provided by contiguous open rural areas have public good characteristics that are not priced in land markets and therefore tend to be undersupplied by private producers (Plantinga and Miller 2001). This has led some to conclude that regional planning should guide rural land use (Breslaw 1990). Nelson (1992) recommended that policies aimed at controlling land fragmentation should (1) increase the productive value of rural land, (2) stabilize the consumptive value in land, (3) eliminate speculative value of farmland, and (4) eliminate the ''impermanence syndrome.'' Some state and local governments have resorted to the purchase or transfer of development rights, establishment of agricultural districts, right-to-farm laws, large-area land use planning, and agricultural zoning (Vesterby and Heimlich 1991). Others have argued that perhaps the only effective deterrent to the subdivision of large rural landholdings and loss of associated amenities is the compensation of landowners for foregone development rents (Plantinga and Miller 2001). An increasingly popular approach to protecting the integrity of landholdings is the implementation of conservation easements (Weibe and others 1996), in which the landowner relinquishes the development rights on the property for the term of the easement in exchange for a payment or tax reduction (Plantinga and Miller 2001).

In the face of increasing population pressure in a private land state, such as Texas, rural land subdivision is unlikely to cease even if comprehensive land use planning mechanisms are implemented. Based on this, a framework for modeling land use changes in response to shifting property size patterns is an important first step for developing policy recommendations for land management institutions that maintain and enhance ecosystem services provided by rural areas.

A Framework for Modeling Land Ownership Fragmentation

To retain the integrity of ecosystems and the services they provide and to mitigate negative impacts of land fragmentation, it is necessary to consider rural land management from an ecosystem perspective. A good indicator of ecosystem health is determined by its sustainability, which is achieved when all ecosystem elements contribute to its overall wholeness (Krehbiel and others 1999).

A model developed to describe discontinuities in the dynamics of semiarid rangeland ecosystems is the State-and-Transition model (Westoby and others 1989). In this model, states are relatively persistent plant communities that encompass a degree of variation in composition over time and space, while transitions between discrete alternate states are triggered by certain natural or anthropogenic threshold conditions, such as persistent drought, fire, or management actions. In the United States, the Natural Resources Conservation Service has adopted the State-and-Transition model as the basis for Ecological Site Descriptions (ESD). The ESDs describe potential changes in landscapes after the occurrence of some environmental ''threshold,'' and, where possible, the management measures that are necessary to return the site to a previous condition or ''state.'' One important characteristic of the State-and-Transition model is that not all transitions are reversible, especially if key elements of the system are substantially depleted or eliminated. For example, significant soil loss can lead to long-term changes in plant community structure by disrupting nutrient cycling. Also, not all services can be restored, and those that are heavily degraded may require considerable time for restoration (MEASR 2005), with the cost of restoration being generally high in relation to the cost of preventing degradation of the ecosystem in the first place.

We propose that the State-and-Transition model can also be used as a framework for modeling discontinuous effects of shifts in property size patterns on ecosystem management, and for considering policies aimed at curbing land subdivision to maintain the supply of critical ecosystem services provided by contiguous open land. In terms of a tool for forecasting land subdivision, the State-and-Transition model can be used to represent property size states (or categories) with relatively rapid transitions after the occurrence of key market thresholds. In the four-stage model presented in Figure 1, State I represents large ranches (>800 ha, >2000 acres); State II includes medium-sized farms and ranches (200–800 ha, 500–2,000 acres); State III represents small farms and ranches (4–200 ha, 20–500 acres); and State IV represents exurban development (*sensu* Cowley and Spillete 2001) (< 4 ha, < 20 acres).

In the case of land subdivision, irreversible transitions between property size states can occur when new parcels are developed with permanent structures that make it economically intractable to reconsolidate land parcels. In addition, the increase in impervious cover resulting from building construction reduces the vegetative cover and infiltration capacity of land, increasing runoff and soil loss potential (Wilcox 2002). Also, increasing density in permanent structures may increasingly inhibit the use of some management tools necessary for maintaining healthy ecosystems. For example, the use of episodic fire is often critical for maintaining vigorous herbaceous cover to enhance infiltration or for maintaining habitat heterogeneity and biodiversity.

Notable land use trends related to shifts in rural property size have been reported for Texas. With respect to land use and land cover, ranches greater than 800 ha are more likely to remain as native rangeland, whereas midsize properties (200–800 ha) are more likely to contain a high proportion of cropland, and areas fragmented into properties less than 200 ha are most likely to be converted to nonnative pasture (Wilkins and others 2003). These land use and land cover differences reflect landowner responses to management realities associated with property size. In addition, shifts in land cover from rangelands to croplands and nonnative pastures affect ecosystem services and increase the potential for pollution externalities, such as elevated nutrient runoff leading to increased water contamination. A spatially

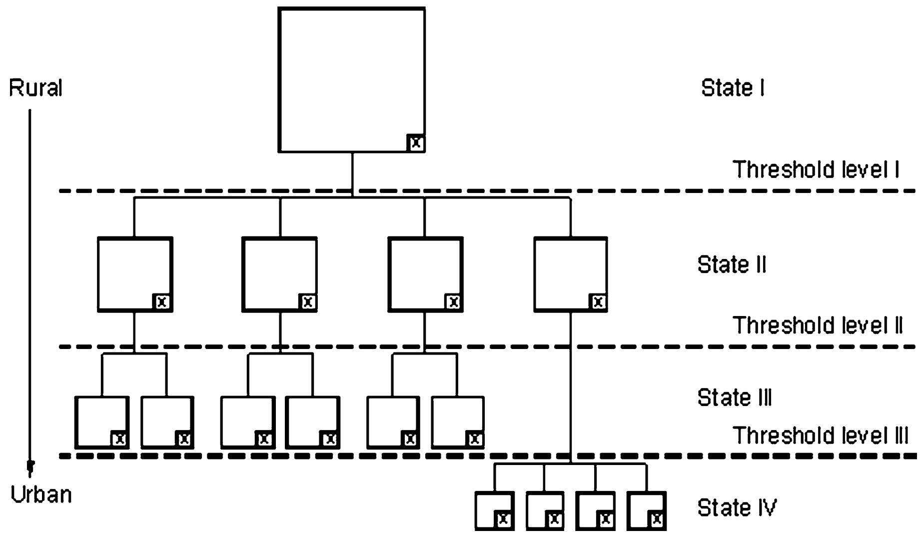

Fig. 1 Model depicting multiple thresholds of land fragmentation. State I = large ranches, State II = medium-sized farms/ranches, Stage III = small farms and ranches, and Stage IV = exurban development. ''X'' represents impervious cover of permanent structures

explicit land use model, based on the State-and-Transition conceptual framework, could be used to forecast areas at increasing risk of land fragmentation and associated ecosystem degradation.

Methodology

Analytical Variables

The market value of Texas rural lands has been identified as a function of the productive and consumptive uses of the land (Pope 1985). The productive value of rural land is the value of land for agricultural (including native rangeland) or forestry products and can be quantified as the present discounted value of expected returns from the land (Munroe and York 2003). In contrast, the consumptive value is generally the value placed on land according to aesthetic and recreation appeal, i.e., the value of land if it were ''consumed'' for nonagricultural purposes (Nelson 1992; Pope 1985). If the most profitable use of rural land is nonagricultural, or if it is purchased for nonproduction purposes, such as enjoyment of open space, then its market value tends to be higher than its agricultural use value (Shi and others 1997). This disparity in market and productive values of rural land tends to increase with proximity to urban areas because of the demand for recreation and other consumptive land values. In addition, growth in the consumptive use of rural land applies upward pressure to land prices, is a major force in increasing the number of small-scale farms and ranches, and can lead to a bimodal distribution of rural property sizes (Pope 1985). For these reasons, we examine the relationships between NAV and changes in rural property size.

Based on durable goods theory, land-price-land-characteristic functions have been referred to as the demand for land (Chicoine 1981), which tends to be positively correlated with population density (Gilliland 1988), proximity to major metropolitan areas, and recreational and aesthetic appeal (Pope 1985). Consequently, the price differential between the market value and productive use value of rural land has been linked directly to the population of proximate urban areas (Shi and others 1997; Berry 1978). As a result, some studies used population density as the sole proxy for consumptive values of rural land, but it has also been found that urban to rural land use ratios can be better explained when residential land value and industrial concentration are incorporated as explanatory variables (Munroe and York 2003). In addition, urban spatial area has been linked not only to population, but also to income and agricultural rent (productive value of land) (Brueckner and Fansler 1983). Based on these observations, we do not assume that population density is the sole determinant of change in rural property size, but rather we examine relationships among property size change, agricultural value, and NAV. We assume that changes in economic activity and income are reflected in changes in the agricultural and nonagricultural values of land over time.

Data and Analysis

The database used in this study included population density, property size, land use, and land value statistics for Texas from 1987 to 2000, which were available either for Independent School Districts (ISDs) or at the county level. Specifically, we obtained the following information: (1) Census data for each of the 254 Texas counties for the years 1990 and 2000 from the Texas State Data Center (U.S. Census Bureau); (2) Farm and ranch ownership sizes for each county in 1987, 1992, and 1997 from the USDA Agricultural Statistics Service; (3) Land use coverage for each of the 1072 Texas ISDs in 1992, 1997, and 2000 from the Texas State Comptroller of Public Accounts; and (4) Land values for each ISD in 1992, or 1997, and 2000 from the Texas State Comptroller of Public Accounts. We also used Geographic Information Systems (GIS) base maps obtained from the Texas Natural Resources Information System to aggregate from the ISD to the county level of analysis. The 38 ISDs (ca. 3.5% of total) that did not report data for 1992 or 1997 were excluded from the analysis. Of these nonreporting ISDs, 36 are located in three metropolitan counties, whereas the remaining 2 represent a small portion of the area in their respective counties and likely have values similar to the surrounding ISDs.

Property size data consisted of the number of and area in each of four size classes: <40, <202, 202–809, and >809 ha (<100, <500, 500–2000, and >2000 acres). Land use data consisted of area in each of nine land use categories: irrigated cropland, dry cropland, barren wasteland, orchards, improved pasture, native pasture, wildlife management areas, timberland, and ''other.'' Land value data consisted of market and agricultural productive values for each of the nine land use types. Agricultural productive value is defined as the present discounted value of expected returns to the land, whereas market values are appraisals made by a Central Appraisal District, i.e., the productive value plus the NAV of the land (Pope 1985). In this study, it is assumed that variations in arable land (topography, water availability, and soil quality) are implicitly captured in the agricultural and nonagricultural market values of land.

The average productive value (PV) and market value (MV), weighted by the proportions of area in each of the nine land use categories, and the associated NAV (MV-PV) were calculated for each ISD. These three sets of average values were then joined to a shapefile of the 1072 Texas ISDs and converted to a grid theme (100 m $\times$ 100 m pixel size) using ArcView GIS 3.2 with the Spatial Analyst extension. The grid was then summarized based on the zones (counties) of a shapefile of the 254 Texas counties to obtain the weighted averages for each county.

Changes in (1) population density (1990–2000), (2) proportion of properties in each size class (1992–1997), (3) proportion of rural land area in each size class (1987–1997, and 1992–1997), and (4) NAV per ha of land (1992–1997, and 1992–2000) were evaluated and mapped using ArcView GIS. In addition, the presence of spatial autocorrelation or spatial structure of these variables was assessed using Mantel tests (Fortin and Gurevitch 1993). The (simple) Mantel test evaluates the correlation between two distance matrices (Fortin and Gurevitch 1993) —a variable distance matrix (**A**) with the absolute differences for the variable of concern in pairs of locations as its elements and a spatial distance matrix (**B**) with the Euclidian distances between corresponding pairs of locations as its elements. The Mantel's r statistic is used to measure the correlation between the elements of the two matrices, based on the Pearson correlation coefficient:

$$r = \frac{1}{N-1}\sum_{i=1}^{n}\sum_{j=1}^{n}\left[\frac{(\mathrm{A}_{ij}-\overline{\mathrm{A}})}{S_{\mathrm{A}}}\right]\left[\frac{(\mathrm{B}_{ij}-\overline{\mathrm{B}})}{S_{\mathrm{B}}}\right]$$

where A_{ij} and B_{ij} are elements of the triangular distance matrices **A** and **B**, respectively; i and j are locations; and N is the number of elements in each triangular matrix. Mantel tests are conducted using a randomization test procedure that compares the Mantel's r for the variable to a reference distribution generated by randomly shifting the elements of one of the matrices and recalculating the Mantel's r many times. If a strong spatial pattern is present in the data for the variable, the random shuffling of the data points should eliminate the pattern (Fortin and Gurevitch 1993). In the case of a strong correlation between the two distance matrices, either positive or negative, the majority of the reference distribution's values will be either higher or lower than the Mantel's r for the variable (Fortin and Gurevitch 1993). The R Package 4.0, a multivariate and spatial analysis program (Casgrain and Legendre 2001), was used for the Mantel tests, using 1000 randomizations and $\alpha = 0.05$ level of significance.

Relationships among the change variables for population density, relative number and area of properties in different size classes, and NAV were evaluated using both direct and spatial correlations. Pearson correlation coefficients were used to measure the direct correlation between the change variables. The presence of spatial autocorrelation in most of these variables violates the assumption of independence of the observations (Legendre and others 1990; Cliff and Ord 1981) and leads to biased estimation of error variance and t-test significance levels (Overmars and others 2003; Anselin and Griffith 1988). Therefore, Dutilleul's modified t-test that corrects the degrees of freedom according to the level of autocorrelation in the data (Dutilleul 1993) was used to assess the significance of the direct correlations using the PASSAGE software (Rosenberg 2001). The spatial correlations between the changes among given variables were evaluated using cross Mantel tests, which are Mantel tests using two variable distance matrices based on data for the two variables (or for the same variable measured at two different times) in the same set of locations (Wu and Mitsch 1998). The cross Mantel test is used to assess the correlation between the spatial patterns, or spatial variation, of two variables over the landscape of interest.

Results

A trend is readily observed from the 10-year analyses of changes in both the proportion of properties and land area in the four property sizes. Of the 254 Texas counties, the proportion of properties and area in property size class >809 ha increased in 236 and 146 counties, respectively. The proportion of properties and area in the medium property size class, 202–809 ha, decreased in 236 and 144 counties, respectively. Concerning the proportion of properties and area in property size class <202 ha, there was a decrease in 221 counties and an increase in 166 counties, respectively. Meanwhile, the proportion of properties and area in property size class <40 ha increased in 198 and 160 counties, respectively. Furthermore, in every county where the proportion of properties in the medium property size class decreased, there was an increase in the proportion of properties in the large property size class. Notably, of the 236 counties where the proportion of properties in the medium property size class decreased, there was an increase in the proportion of properties in the small property size class (<40 ha) for 191 (81%) of the counties.

Results of Mantel tests showed significant spatial autocorrelation in most of the variables involving temporal change (Table 1). The 10-year change in population density was spatially autocorrelated ($P < 0.05$). Similarly, the 10-year changes in the proportions of land area in each of the four property size classes were spatially autocorrelated (<40 ha size class marginal with $P = 0.054$). In the 5-year analysis (Table 1, Figure 2), changes in the proportions of land area in property size categories were spatially autocorrelated for the 202–809 ha and <40 ha categories ($P < 0.05$) but not for the >809 ha and <202 ha size classes. The 5-year changes in the proportion of properties were spatially autocorrelated ($P < 0.05$) for all but the largest property size class. Agricultural value in 2000 was spatially autocorrelated ($P < 0.01$), as was the 5-year change in nonagricultural land value ($P < 0.05$) (Figure 3), but the 8-year change was not.

Results of the Dutilleul's modified *t*-tests and cross Mantel tests showed spatial correlation between changes in population density and changes in relative number and area of the property size classes, whereas there was little direct correlation between them (Table 2). The 10-year changes in population density are spatially correlated with changes in the proportion of land area in each of the four property size classes ($P < 0.05$), although there were no direct correlations between them. The pattern of relationships between the changes in population density and proportion of properties in the four property size classes, however, appears somewhat different, with significant spatial correlations for all but the medium property size class (marginally significant for >809 ha size class) and no significant direct correlation except for the medium property size class. Moreover, although the spatial correlations were positive for the three smaller property size classes, the spatial correlation was negative for large properties (>809 ha). In addition, population density and NAV in 2000 had significant positive correlation, directly and spatially, whereas the population density and agricultural value in 2000 had no correlation (Table 2). Changes in population density (1990–2000) and NAV (1992–2000) also had significant positive spatial correlation.

Based on 5-year (1992–1997) analyses, changes in the proportion of the number of properties in each of the four property size classes and changes in NAV had significant spatial correlations (<202 ha size class marginal with $P = 0.050$) but had significant direct correlation only for the <40 ha and 202–809 ha size classes (Table 2). The correlations for the three smaller property size classes were positive, but that for the largest property size class was negative. Interestingly, the proportion of properties >809 ha has tended to decrease, whereas the proportions of properties <202 ha have tended to increase in areas where the NAV of land has increased, 66% and 99%, respectively (compare Figures 2 and 3).

Table 1 Spatial autocorrelations using Mantel's r with randomization Mantel test

Variable	Mantel's *r*	*P*
Change in population density (1990–2000)	0.098	**0.001**
Change in proportion of properties >809 ha (1992–1997)	0.032	0.131
Change in proportion of properties 202–809 ha (1992–1997)	0.064	**0.043**
Change in proportion of properties <202 ha (1992–1997)	0.133	**0.001**
Change in proportion of properties <40 ha (1992–1997)	0.116	**0.001**
Change in % area in properties >809 ha (1992–1997)	0.030	0.172
Change in % area in properties 202–809 ha (1992–1997)	0.107	**0.003**
Change in % area in properties <202 ha (1992–1997)	0.042	0.100
Change in % area in properties <40 ha (1992–1997)	0.113	**0.005**
Change in % area in properties >809 ha (1987–1997)	0.058	**0.041**
Change in % area in properties 202–809 ha (1987–1997)	0.095	**0.003**
Change in % area in properties <202 ha (1987–1997)	0.146	**0.001**
Change in % area in properties <40 ha (1987–1997)	0.055	0.054
Change in nonagricultural land value (1992–2000)	−0.006	0.450
Change in nonagricultural land value (1992–1997)	0.105	**0.004**

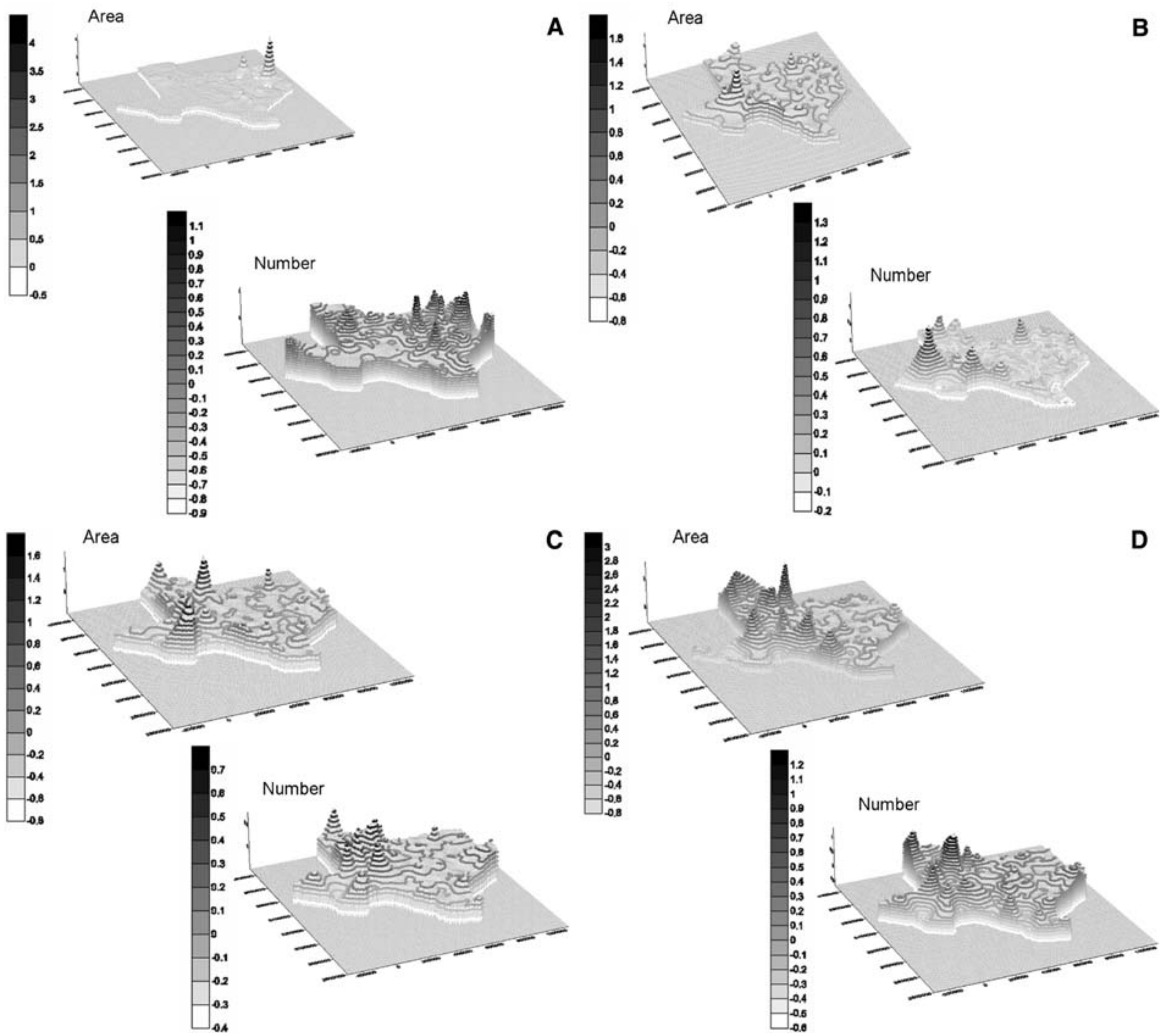

Fig. 2 Proportional change (1992 versus 1997) in Texas in land area and number of properties in each property size category: (**A**) >809 ha, (**B**) 202–809 ha, (**C**) <202 ha, and (**D**) <40 ha

When considering the relationship between changes in land area in each property size class and changes in the nonagricultural land value, both direct and spatial correlations were significant and positive for land area in properties <202 ha and <40 ha (Table 2). Of the 118 counties where NAV increased, land area in properties <202 ha and <40 ha increased in 72 (61%) and 88 (75%) of the counties, respectively, whereas the proportion of area in properties >809 ha and 202–809 ha decreased in 53% of them. The results support the contention that land subdivision generally increases in areas where NAV increases. However, proportions of land area in properties >809 ha and 202–809 ha were not significantly correlated with NAV, perhaps a result of lag times associated with economies of scale.

Discussion and Conclusions

We empirically examined spatial–temporal relationships among changes in population density, rural property size, and agricultural and nonagricultural (consumptive) land values in Texas. Changes in property size, specifically changes in relative number and area of properties in various size classes, were spatially correlated with changes in population density, although they had no direct correlation, with one exception for the proportion of properties 202–809 ha. Changes in property size were also significantly correlated with NAV, both directly and spatially for relative area of properties <202 ha and <40 ha, spatially for relative number of properties in all size classes, and directly for relative number of properties 202–809 ha and

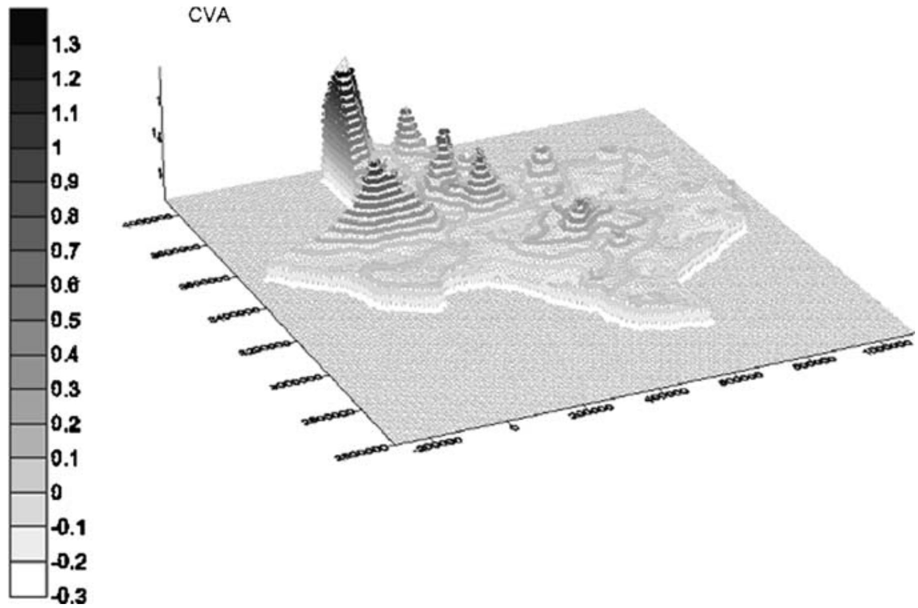

Fig. 3 Proportional change (1992 versus 1997) in nonagricultural land value per hectare in Texas. The nonagricultural value or Consumptive Value per Acre (CVA) is the difference between the market value and the agricultural productive value of land

Table 2 Direct (Pearson's r with Dutilleul's modified t-test) and spatial correlation between variables

Variables	Direct correlation		Spatial correlation	
	Pearson's *r*	*p*	Mantel's *r*	*p*
Change in population density (1990–2000) vs. change in % area in properties >809 ha (1987–1997)	−0.093	0.210	−0.066	**0.031**
Change in population density (1990–2000) vs. change in % area in properties 202–809 ha (1987–1997)	−0.087	0.328	0.090	**0.041**
Change in population density (1990–2000) vs. change in % area in properties < 202 ha (1987–1997)	0.044	0.672	0.127	**0.004**
Change in population density (1990–2000) vs. change in % area in properties < 40 ha (1987–1997)	0.094	0.320	0.106	**0.034**
Change in population density (1990–2000) vs. change in % of properties >809 ha (1987–1997)	−0.062	0.486	−0.062	***0.051***
Change in population density (1990–2000) vs. change in % of properties 202–809 ha (1987–1997)	0.364	**0.036**	0.058	0.104
Change in population density (1990–2000) vs. change in % of properties < 202 ha (1987–1997)	0.403	0.070	0.090	**0.023**
Change in population density (1990–2000) vs. change in % of properties < 40 ha (1987–1997)	−0.370	0.089	0.076	**0.037**
Population density (2000) vs. Agricultural land value (2000)	0.117	0.112	0.018	0.279
Population density (2000) vs. nonagricultural land value (NAV) (2000)	0.449	**0.000**	0.490	**0.001**
Change in population density (1990–2000) vs. change in NAV (1992–2000)	0.061	0.322	0.114	**0.027**
Change in population density (1987–1997) vs. change in Average farm size (1990–2000)	−0.145	***0.050***	0.041	0.140
Change in NAV (1992–1997) vs. change in % area in properties >809 ha (1992–1997)	−0.031	0.655	−0.029	0.364
Change in NAV (1992–1997) vs. change in % area in properties 202–809 ha (1992–1997)	−0.008	0.915	0.012	0.300
Change in NAV (1992–1997) vs. change in % area in properties < 202 ha (1992–1997)	0.203	**0.003**	0.215	**0.028**
Change in NAV (1992–1997) vs. change in % area in properties < 40 ha (1992–1997)	0.388	**0.000**	0.424	**0.001**
Change in NAV (1992–1997) vs. change in % of properties >809 ha (1992–1997)	−0.041	0.504	−0.071	**0.014**
Change in NAV (1992–1997) vs. change in % of properties 202–809 ha (1992–1997)	0.203	**0.004**	0.205	**0.028**
Change in NAV (1992–1997) vs. change in % of properties < 202 ha (1992–1997)	0.075	0.289	0.103	***0.050***
Change in NAV (1992–1997) vs. change in % of properties < 40 ha (1992–1997)	0.297	**0.000**	0.362	**0.001**

< 40 ha. Furthermore, from 1987 to 1997 an increase in the proportion of properties and area in the large (>809 ha) and small (< 40 ha) property size classes, and a reduction in the medium property size class was observed. Also, the proportion of properties in the medium property size class decreased in 93% of Texas counties. All of the counties where the medium property size class decreased experienced an increase in the proportion of properties in the large property size class, whereas 81% of them also showed an increase in the proportion of properties in the small property size class.

Interestingly, the results of the 1992 to 1997 analysis indicate that the proportion of area in large and medium sized properties has decreased, whereas the proportion of area in small properties has increased in areas where nonagricultural land values have grown. Specifically, of the 118 counties where NAV increased, the proportion of area in properties >809 ha and 202–809 ha decreased in 53% of them, but the magnitudes of these decreases were not equal. Also, of the 53% of counties where the proportion of area in medium-sized properties decreased, 61% had an increase in the proportion of area in large properties.

However, of the counties where NAV decreased, the proportion of area in properties >809 ha only increased in 43% of the counties. Overall, the area in large properties actually increased because the gains in 43% of the counties were larger than the losses in the other 57%. From 1992 to 1997, there was a gain of more than 161,874 ha in ownership sizes <40 ha, a gain of more than 384,451 ha in ownership sizes <202 ha, a loss of more than 303,514 ha in ownership sizes between 202 and 809 ha (medium-sized farms and ranches), and a gain of more than 80,937 ha in ownership sizes >809 ha (Wilkins and others 2003). The trend of bimodal property size distribution can be observed via the statewide changes from both 1992 to 1997 and 1987 to 1997. With respect to Pope's (1985) hypothesis (objective 1) that a significant increase or decrease in nonagricultural (consumptive) land value leads to a bimodal distribution in rural property size, our results suggest that rural property size distribution significantly shifts more toward smaller properties as NAV increases. However, although property size distribution also shifts more toward larger properties as NAV decreases, this correlation is not statistically significant.

With respect to objective 2, the results suggest that changes in population density in Texas are not only spatially correlated with changes in property size but also are directly and spatially correlated with changes in consumptive values of rural land. This finding corroborates earlier empirical evidence that the NAV of rural land is positively correlated with population density in proximate urban areas (Shi and others 1997) and also demonstrates that this relationship exists in rural areas. Overall, these results suggest that patterns of changes in nonagricultural land values and population densities are effective, general indicators of trends in land fragmentation and consolidation in Texas.

It should be noted that we do not infer that the presence or absence of spatial or direct correlation indicates a causal association between variables included in the analysis; within the scope of this study, we were not searching for the causal relationships *per se*. We do not state that NAV causes changes in population density nor that population density causes farm size changes or vice versa. However, the strengths of the spatial correlations hold across the state of Texas and provide a consistent measure that can be incorporated into a model to forecast areas that may be in greatest jeopardy of future land fragmentation. Because demographers forecast population changes, and economists forecast economic conditions that are reflected in NAV and agricultural value changes, ecologists may be able to forecast land ownership and parcel size changes, and associated habitat changes in areas where endangered or threatened species depend upon a given habitat quality and size. Conceivably, using the same modeling approach, one could also forecast changes in watershed health based upon projected changes in native habitat or other land use conversions associated with land ownership and parcel size changes.

Viewed within the context of the State-and-Transition model (objective 3), the ongoing decline in midsized properties and the increase in smaller properties can hinder integrated ecosystem management. Integrated land management is likely to become more critical as nonrenewable resources become depleted and society becomes more dependent on the self-renewing capacity of ecosystems to produce biotic resources (Daly and Farley 2004). Moreover, an increase in smaller properties does not facilitate coordinating land management to maintain the supply of services produced by ecosystems, which extend across individual property boundaries. We briefly discuss two mechanisms for counteracting the ecologically deleterious effects of land subdivision: conservation easements and land management associations.

Fragmentation: Policy Options and the State and Transition Model

Assuming that the motivation for subdividing rural land is represented by a substantial portion of the NAV of land, incentives that retard the growth of consumptive value might provide an effective strategy to slow the rate of land ownership fragmentation. One option is the purchase of development rights associated with land ownership. Conservation easements have been proposed and adopted by numerous entities that are interested in maintaining the spatial integrity of land units threatened by increasing consumptive land values. Compared to alternative land use policies, conservation easements have a number of attractive features. For instance, the cost to the conservator is lower with easements than with land purchases, and easements tend to be politically more palatable than regulatory approaches because landowners enter into the agreements voluntarily (Plantinga and Miller 2001). As a result, some have touted transfer of development rights and programs aimed at purchasing such rights as the most effective tool for limiting rural land subdivision. However, landowners closest to urban areas who anticipate windfalls from development have little incentive to participate in such programs (Nelson 1992).

In order to find alternative sources of land-based income, many Texas ranchers have and continue to turn to fee-based hunting. To effectively manage wildlife at the landscape level and to mitigate the deleterious effects of land fragmentation on wildlife habitat, some landowners have formed wildlife management associations. These entities operate under voluntary Texas Parks and Wildlife Department plans for improving habitat and managing wildlife populations (mainly for white-tailed deer) (Texas Parks and

Wildlife 1998). Over 100 such groups representing nearly 4000 landowners and approximately 600,000 ha have organized in Texas. In addition to wildlife management, such associations have also been proposed for the sustainable management of other nonexclusive resources (Ostrom 1990), including groundwater in Texas (Wagner and Kreuter 2004). Such associations hold promise for managing many ecosystems and the natural resources and services they provide because membership represents stakeholders who benefit from collective success.

In the context of the State-and-Transition Model that we proposed as a framework for depicting multiple thresholds of land fragmentation with respect to integrated ecosystem management, both conservation easements and landowner associations represent mechanisms for avoiding or reversing shifts in property size conversions across thresholds (Figure 1). Specifically, conservation easements represent a mechanism for avoiding ''threshold 1,'' where large properties are reduced in size to midsized properties, by eliminating the option for exploiting the development value of land. This will enhance the survival of larger tracts of land that contain greater portions of individual ecosystems than smaller land parcels and thereby enhance ecosystem management at the individual property scale.

Landowner cooperatives, by contrast, could be conceived as a mechanism for counteracting the constraints to ecosystem management when land parcels transition from the midsized to the small landholding category, ''threshold 2.'' When properties decrease in size to a point where they can no longer support economically viable farming or ranching operations, neighboring landowners may have an increased incentive to share the costs and risks of managing their land and to jointly benefit from the resources, such as wildlife or groundwater, which their collective land area provides. Such cooperative landowner associations have the potential to facilitate ecosystem-level management by coordinating the land management decisions of participating members. For government agencies tasked with promoting ecosystem sustainability, such associations also facilitate the dissemination of information and the provision of support for integrated ecosystem management which benefits participating landowners and the public in terms of the services that these ecosystems provide.

Based on our analysis of shifts in property size distributions in response to shifts in the NAV of rural land resulting from spatially heterogeneous population changes, and the observation that conservation easements and landowner associations may facilitate integrated ecosystem management, we suggest a detailed evaluation of the effectiveness of such mechanisms for ensuring the continued provision of ecosystem services. Specifically, we suggest that agencies tasked with the maintenance of such services focus more actively on supporting institutional arrangements that enhance voluntary landowner cooperation in making land management decisions.

In many areas, efforts to protect biodiversity and ecosystem services need to focus on the condition and management of private land (Wear and others 2004, Theobold 2003), especially in states like Texas where land is predominantly privately owned. Involvement of private landowners in natural resource management plans is particularly important in North America where people have tended to settle in the most species-rich areas (Ricketts and Imhoff 2003), thereby limiting the effectiveness of remote parks, where species richness tends to be lower, for maintaining biodiversity (Wear and others 2004, Margules and Pressey 2000). Rather, maintaining species richness calls for an ecosystem management approach that meshes human productivity and biodiversity (Ricketts and Imhoff 2003; Daily and others 2001).

The State-and-Transition model can be used to spatially represent shifts between categories of property sizes that represent generally different land uses patterns, and to integrate the concept of ''resilience'' within a given ecosystem. Resilience is the ability of ecosystems to respond to pressures in such a way as to remain in, or have the ability to return to, a given state. Although undisturbed natural systems tend to be continually in a state of flux that enhances biodiversity and, thus, system resilience (Holling 1973), human actions tend to homogenize land cover characteristics by planting monocultures and applying herbicides and pesticides (Holling and Meffe 1996). Such homogenization tends to reduce resilience in ecological systems, thereby lowering thresholds for shifts to less desirable states with reduced natural capital, human quality of life, and policy options (Conroy and others 2003). Shifts to new resilient but undesirable states are often difficult and prohibitively costly to reverse (Conroy and others 2003). Therefore, the State-and-Transition model not only accounts for ecological resilience, but can also account for ''economic resilience'' with respect to the feasibility of reamalgamating small properties with high market values into larger agriculturally cohesive units.

In terms of ecological and economic resilience, the states and their defining thresholds in an ecosystem can be gauged using transition probabilities. Brown and others (2000) argued that models of land use change can be linked to critical land cover outcomes through the use of Markov land cover transition probabilities, calculated as a function of land use conditions and land use change. The land use transitions should be viewed as ''possible development paths where the direction, size, and speed can be influenced through policy and specific circumstances'' (Lambin and others 2003; Martens and Rotmans 2002). It might be assumed that the spatial sequence of future urban development will follow the order determined by transition probabilities, i.e., the higher the transition probability, the sooner the development of the land cell is likely to occur (Allen and Lu 2003). Locating where ownership size and land use transitions are taking place and likely to continue to occur is important from an ecological and economic perspective in terms of conserving biodiversity and ecosystem services.

Future Research

Importantly, the statewide relationships established herein provide an empirical framework that can be incorporated into a spatially explicit land use model for Texas. Models of land use change may address two separate questions: (1) Where are land use changes likely to occur (location), and (2) At what rate (quantity of change) (Veldkamp and Lambin 2001)? The rate of change is driven by the demand for land-based commodities (Stephenne and Lambin 2001) and is often modeled using an economic framework (Fischer and Sun 2001; Veldkamp and Lambin 2001). Such coupled models can be used as decision-support tools that can forecast regions that may be most prone to fragmentation, and assist policymakers and individuals in making complex land use conservation decisions based on projected future scenarios based on alternative sets of conditions. Such scenarios can be used to highlight the implications of alternative assumptions about critical uncertainties concerning the behavior of human and ecological systems (MEASR 2005). In essence, it offers the possibility to test the sensitivity of land use patterns to selected variables (Farrow and Winograd 2001, Veldkamp and Lambin 2001). For instance, policies such as the purchase of development rights could be implemented and fragmentation rates would be projected into the future for given locations. Such results could identify the areas where farms, ranches, and wildlife habitats may be in the highest degree of jeopardy. Simulation results could provide not only estimates of the costs of purchasing development rights, but also the potential ecological and economic benefits for selected areas under analysis. Moreover, benefits and costs of initiating a conservation program now versus some time in the future could be compared.

Lambin and others (2003) stated that an integrative framework that could provide a unifying theory for insights into the pathways of land use change has thus far been lacking (Lambin and others 2003). It is our view that the State-and-Transition model—incorporating relationships proposed by Pope (1985) and Shi and others (1997), Markov transition probabilities as modified by Brown and others (2000), and Holling's (1973) idea that ecosystems lose resilience when they transition to more homogeneous states—provides a useful integrating framework that can be applied using a spatially explicit simulation model. This approach accommodates a management approach based on resilience in that it views events in a regional, rather than a local, context and emphasizes heterogeneity while keeping management options open. Furthermore, it does not require the capacity to precisely predict the future, but only a qualitative capacity to devise systems that can handle unexpected future events (Holling 1973).

In summary, effective conservation necessitates not only targeting areas where habitats are degraded, but also those where conditions may become degraded soon (Wear and others 2004). To identify broad-scale conservation priorities, there is a need to develop a quick but rigorous method for identifying areas where high biodiversity may be threatened by intensified future human activity (Ricketts and Imhoff 2003). Land use policies and projections of the future role of land use change in Earth System dynamics should capture socioeconomic and biophysical drivers of land use change and also account for the specific human-environment conditions under which drivers of change operate. This calls for advances that capture the generic qualities of both socioeconomic and biophysical drivers as well as the place-based, human-environment conditions that direct land use and land cover change (Lambin and others 2001).

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References

1. Alig R. J., R. G. Healy. 1987. Urban and built-up land area changes in the United States: an empirical investigation of determinants. Land Economics 63:215–226
2. Allen J., K. Lu. 2003. Modeling and prediction of future urban growth in the Charleston region of South Carolina: a GIS-based integrated approach. Conservation Ecology 8:2
3. Anselin L., A. D. Griffith. 1988. Do spatial effects really matter in regression analysis? Papers, Regional Science Association 65:11–34
4. Bentley J. W. 1987. Economic and ecological approaches to land fragmentation: in defense of a much-maligned phenomenon. Annual Review of Anthropology 16:31–67
5. Berry D. 1978. Effects of urbanization on agricultural activities. Growth and Change 9:2–8
6. Bradley G. A. (ed.). 1984. Land use and forest resources in a changing environment: the urban/forest interface. University of Washington Press, Seattle, Washington
7. Breslaw J. A. 1990. Density and urban sprawl: Comment. Land Economics 66:464–468
8. Brown D. G., B. C. Pijanowski, J. D. Duh. 2000. Modeling the relationships between land use and land cover on private lands in the Upper Midwest, USA. Journal of Environmental Management 59:247–263
9. Brueckner J. K., D. A. Fansler. 1983. The economics of urban sprawl: theory and evidence on the spatial sizes of cities. The Review of Economics and Statistics 65:479–482

10. Casgrain, P., and P. Legendre. 2001. The R Package for multivariate and spatial analysis, version 4.0 d5—user's manual. Département de sciences biologiques, Université de Montréal. http://www.fas.umontreal.ca/BIOL/legendre/ (Date Accessed: 9/12/2004)
11. Chicoine D. L. 1981. Farmland values at the urban fringe: an analysis of sale prices. Land Economics 57:353–362
12. Cliff A. D., J. K. Ord. 1981. Spatial processes: models and applications. Pion, London, pp 266
13. Clout H. D. 1984. A rural policy for the EEC? London: Methuen
14. Collinge S. K. 1996. Ecological consequences of habitat fragmentation: implications for landscape architecture and planning. Landscape and Urban Planning 36:59–77
15. Conklin H. E., W. G. Lesher. 1977. Farm value assessment as means for reducing premature and excessive agricultural disinvestment in urban fringes. American Journal of Agricultural Economics 59:755–759
16. Conroy M. J., C. R. Allen, J. T. Peterson, L. J. Pritchard, C. T. Moore. 2003. Landscape change in the southern Piedmont: challenges, solutions, and uncertainty across scales. Conservation Ecology 8:3
17. Costanza R. (ed.). 1997. An introduction to ecological economics. CRC Press, Boca Raton, Florida
18. Cowley J. S., S. R. Spillete. 2001. Exurban residential development in Texas. Real Estate Center, Texas A&M University, Technical report 1470, 22 pp
19. Cromartie J. B., J. M. Wardwell. 1999. Migrants settling far and wide in the rural West. Rural Development Perspectives 14:2–8
20. Daily G. C., P. R. Ehrlich, G. A. Sanchez-Azofeifa. 2001. Countryside biogeography: use of human-dominated habitats by the avifauna of southern Costa Rica. Ecological Applications 11:1–13
21. Daily G. C., K. Ellison. 2002. The new economy of nature. Island Press, Washington, D.C., 260 pp
22. Daly H. E., J. Farley. 2004. Ecological economics: principles and applications. Island Press, Washington, D.C., 454 pp
23. Dutilleul P. 1993. Modifying the t test for assessing the correlation between two spatial processes. Biometrics 49:305–314
24. Farrow A., M. Winograd. 2001. Land-use modeling at the regional scale: an input to rural sustainability indicators for Central America. Agriculture, Ecosystems and Environment 85:249–268
25. Fischer G., L. Sun. 2001. Model based analysis of future land use development in China. Agriculture, Ecosystems and Environment 85:163–176
26. Fortin M., J. Gurevitch. 1993. Mantel tests: spatial structure in field experiments. In: S. M. Scheiner, J. Gurevitch (eds), Design and analysis of ecological experiments, 2nd ed. Oxford University Press, New York, New York. pp 308–326
27. Fulbright T. E. 1997. Designing shrubland landscapes to optimize habitat for white-tailed deer. In: D. Rollins, D. N. Ueckert, G. Brown (eds), Brush sculptors: Innovations for tailoring brush rangelands to enhance wildlife habitat and recreational value. Texas Agricultural Extension Service, San Angelo, Texas. pp 61–67
28. Gilliland C. E. 1988 (February). Motives of Texas rural land buyers. Real Estate Center—Technical report #621. Real Estate Center, Texas A&M University, College Station, Texas, 16 pp
29. Greene R. P., J. Stager. 2001. Rangeland to cropland conversions as replacement land for prime farmland lost to urban development. Social Science Journal 38:543–555
30. Healy R. G. 1985. Population growth in the U.S. South: implications for agricultural and forestry land supply. Rural Development Perspectives 2:27–30
31. Holling C. S. 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics 4:1–23
32. Holling C. S., G. K. Meffe. 1996. Command and control and the pathology of natural resource management. Conservation Biology 10:328–337
33. King R., S. Burton. 1982. Land fragmentation: notes on a fundamental rural spatial problem. Progress in Human Geography 6:475–494
34. Krehbiel T. C., R. F. Gorman, O. H. Erekson, O. L. Loucks, P. C. Johnson. 1999. Advancing ecology and economics through a business-science synthesis. Ecological Economics 28:183–196
35. Lambin E. F., B. L. Turner II, H. J. Geist, S. B. Agbola, A. Angelsen, J. W. Bruce, O. T. Coomes, R. Dirzo, G. Fischer, C. Folke, P. S. George, K. Homewood, J. Imbernon, R. Leemans, X. Li, E. F. Moran, M. Mortimore, P. S. Ramakrishnan, J. F. Richards, H. Skanes, G. D. Stone, U. Svedin, T. A. Veldkamp, C. Vogel, J. Xu. 2001. The causes of land-use and land-cover change: moving beyond the myths. Global Environmental Change 11:261–269
36. Lambin E. F., H. J. Geist, E. Lepers. 2003. Dynamics of land-use and land-cover change in tropical regions. Annual Review of Environment and Resources 28:205–241
37. Legendre P., N. L. Oden, R. R. Sokal, A. Vaudor, J. Kim. 1990. Approximate analysis of variance of spatially autocorrelated regional data. Journal of Classification 7:53–75
38. Margules C. R., R. L. Pressey. 2000. Systematic conservation planning. Nature 405:243–253
39. Martens P., J. Rotmans (eds.). 2002. Transitions in a globalizing world. Swets & Zeitlinger, Lisse, Netherlands, 135 pp
40. McCann S. 1999. Keeping the estate tax. Rangelands 21:3–4
41. Millennium Ecosystem Assessment Synthesis Report (MEASR). 2005. A report of the millennium ecosystem assessment. Prepublication final draft, 219 pp. www.millenniumassessment.org (Date accessed: 6/06/05)
42. Munroe D. K., A. M. York. 2003. Jobs, houses, and trees: changing regional structure, local land use patterns, and forest cover in Southern Indiana. Growth and Change 34:299–320
43. Munton R. J. C., T. K. Marsden. 1991. Occupancy change and the farmed landscape: an analysis of farm-level trends, 1970-85. Environment and Planning A 23:499–510
44. Murdock, S. H., S. White, M. N. Hoque, B. Pecotte, X. You, and J. Balkan. 2002. The Texas challenge in the twenty-first century: implications of population change for the future of Texas. Department of Rural Sociology, Texas A&M University System Departmental technical report 2002-1 (December), 466 pp
45. Nelson A. C. 1992. Preserving prime farmland in the face of urbanization—lessons from Oregon. Journal of the American Planning Association 58:467–487
46. Ostrom E. 1990. Governing the commons: the evolution of institutions for collective action. Cambridge University Press, New York
47. Overmars K. P., G. H. J. de Koning, A. Veldkamp. 2003. Spatial autocorrelation in multi-scale land use models. Ecological Modelling 164:257–270
48. Peiser R. B. 1989. Density and urban sprawl. Land Economics 65:193–204
49. Peterson, R. 1997. Coping strategies of Utah grazing permittees under economic and social pressure. Masters Thesis, Utah State University, 132 pp
50. Plantinga A. J., D. J. Miller. 2001. Agricultural land values and the value of rights to future land development. Land Economics 77:56–67
51. Pope C. A., III. 1985. Agricultural productive and consumptive use components of rural land values in Texas. American Journal of Agricultural Economics 67:81–86
52. Pope C. A., III, H. L. Goodwin Jr. 1984. Socio-economic motivations for rural land purchases in Texas. Journal of the American Society of Farm Managers and Rural Appraisers 48:37–40

53. Riebsame W. E., H. Gosnell, D. M. Theobald. 1996. Land use and landscape change in the Colorado mountains I: theory, scale and pattern. Mountain Research and Development 16:395–405
54. Ricketts T., M. Imhoff. 2003. Biodiversity, urban areas, and agriculture: locating priority ecoregions for conservation. Conservation Ecology 8:1
55. Rollins D. 2000. Integrating wildlife concerns into brush management designed for watershed enhancement. In: J. Cearley D. Rollins (eds.), Proceedings of the Conference on Brush, Water, and Wildlife: a Compendium of our Knowledge. Texas Agricultural Resources and Extension Center, San Angelo, Texas. pp 38–46
56. Rosenberg M. S. 2001. PASSAGE. Pattern analysis, spatial statistics, and geographic exegesis. Version 1.1.3.4. Department of Biology, Arizona State University, Tempe, Arizona
57. Rowan R. C., L. D. White. 1994. Regional differences among Texas rangeland operators. Journal of Range Management 47:338–343
58. Sanders, J. C. 2005. Relationships among landowner and land ownership characteristics and participation in conservation programs in Central Texas. MS Thesis. Texas A&M University, 89 pp
59. Shi Y. J., T. T. Phipps, D. Colyer. 1997. Agricultural land values under urbanizing influences. Land Economics 73:90–100
60. Stephenne N., E. L. Lambin. 2001. A dynamic simulation model of land-use changes in the Sudano-Sahelian countries of Africa. Agriculture, Ecosystems and Environment 85:145–161
61. Texas Governor's Task Force on Conservation. 2000. Taking care of Texas—a report from the governor's task force on conservation. Executive Order GWB 00-1, Austin, Texas, 56 pp. http://www.tpwd.state.tx.us/publications/nonpwdpubs/media/taking_care_of_texas_report.pdf (Date accessed: 9/12/2004)
62. Texas Parks and Wildlife. 1998. A guide to wildlife management associations and co-ops. PWDBK W7000-336 (11/98). Austin, Texas
63. Theobald D. M. 2001. Land-use dynamics beyond the American urban-fringe. The Geographical Review 91:544–564
64. Theobald D. M. 2003. Targeting conservation action through assessment of protection and exurban threats. Conservation Biology 17(6):1624–1637
65. Veldkamp A., E. F. Lambin. 2001. Editorial: predicting land use change. Agriculture, Economics and Ecosystems 85:1–6
66. Vesterby M., R. E. Heimlich. 1991. Land use and demographic change: results from fast-growth counties. Land Economics 67:279–291
67. Wagner M. W., U. P. Kreuter. 2004. Groundwater supply in Texas: private land considerations in a rule-of-capture state. Society and Natural Resources 17:349–357
68. Wear D., J. Pye, K. Riitters. 2004. Defining conservation priorities using fragmentation forecasts. Ecology and Society 9:4
69. Weibe, K., A. Tegne, and B. Kuhn. 1996. Partial interests in land: policy tools for resource use and conservation. U.S. Department of Agriculture, Economic Research Service, agricultural economic report no. 744
70. Westoby M. B., B. Walker, I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management 42:266–274
71. Wilcox B. P. 2002. Shrub control and streamflow on rangelands: a process-based viewpoint. Journal of Range Management 55:318–326
72. Wilcox B. A., D. D. Murphy. 1985. Conservation strategy: the effects of fragmentation on extinction. American Naturalist 125:879–887
73. Wilkins N., R. D. Brown R. J. Conner J. Engle C. Gilliland A. Hays R. D. Slack D. W. Steinbach. 2000 (September). Fragmented lands: changing land ownership in Texas. The Agriculture Program, Texas A & M University, 8 pp
74. Wilkins, R. N., A. Hays, D. Kubenka, D. Steinbach, W. Grant, E. Gonzalez, M. Kjelland, and J. Shackelford. 2003. Texas rural lands: trends and conservation implications for the 21st century. Final summary report of the Texas A&M Rural Land Fragmentation Project, Texas Cooperative Extension Publication B6134, 26 pp
75. Wu X. B., W. J. Mitsch. 1998. Spatial and temporal patterns of algae in newly constructed freshwater wetlands. Wetlands 18:9–20