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**Spatial and Temporal Effects of Urbanization and Forest
Management on Local Diversity of Forest Trees**

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Abstract

Diversity is one of the major factors which define stability and efficient functioning of forest ecosystems and maintaining biodiversity is important for sustainable forest management. One of the many factors affecting forest diversity is urbanization. This paper analyses the diversity of tree species in Alabama forests. Shannon's diversity index and a species richness index are used as indicators of diversity. The influence of biological, climatic and anthropogenic factors on tree diversity and species richness during the period 1972-2000 was studied using regression analysis. Explanatory variables in the regressions included forest type, stand origin, DBH, volume, site class, and ownership (obtained from FIA plot data) and a population gravity index for each plot determined using ArcGIS software. The diversity of tree species is negatively affected by urbanization as represented by the population gravity index.

Keywords: Species richness; Shannon index; disturbance; urbanization; random effects; Poisson; Alabama.

Introduction

Biological diversity is the richness and evenness (relative abundance) of species amongst and within living organisms and ecological complexes. Biological diversity is important for the stability of ecosystems. According to McNaughton (1977), a diverse ecosystem is more resistant to environmental disturbances, and is likely to contain species that would thrive through natural or imposed perturbations in the ecosystem and compensate for the loss of other members (Stapanian et al. 1997). Other important features of diverse ecosystems include their productivity, soil protection capabilities, and their aesthetic values. Diverse ecosystems provide a greater variety of goods and services to humans. Tree diversity is a key component of forest ecosystem biodiversity and this is one of the reasons why biodiversity has emerged as a major objective of forest management and conservation since the Earth Summit in Rio de Janeiro in 1992. Forest diversity is an important criterion of sustainable forest management.

Forest ecosystem diversity is determined by the combination of a variety of natural factors at differing scales, ranging from the global, (climate), to regional, (elevation, precipitation, physiography), and local, (slope, aspect, substrate, and succession). At the same time, anthropogenic disturbances also have significant impacts on ecosystems, including their diversity (Forester and Machlis, 1996; McKee et al., 2003). The most radical anthropogenic impact is the conversion of ecosystems to other land cover types/land uses, such as conversion of a forest into an urban or agricultural use. Since 1630, some 56 million acres of forest lands in the US South have been converted to other uses, predominantly to agriculture (Smith et al., 2004). By the 1920s the clearing of forests for agriculture was largely completed. Subsequently farms have been abandoned, forests were reestablished and renewed, and the shares of land use allocated between forestry and agriculture are relatively stable (Wear, 2002).

With the increasing demand for timber and non-timber products from forests and the shifting of timber industry from northern states to the South, more Southern forest ecosystems were subjected to changes in forest management. Depending on the goals of forest management and their related management intensities, the dynamics of natural processes shifted to various extents, causing soil deterioration, habitat destruction, alteration, and fragmentation, and shifts in age structure and species composition, thus impacting biological diversity (Rosson and Amundsen, 2004).

Urbanization refers to increased intensity of settlement and/or location of businesses and other activities (urban land uses) in an area over time and has multiple effects on forest ecosystems. An obvious consequence of increasing urbanization is the change from forest to developed land use which has been extensively researched (Wear et al., 1998; Munn et al., 2002; Zhang and Nagubadi, 2005) in the past and will not be analyzed in this study. Urban pressure brings changes to forestry and agriculture (abandonment of agriculture and decreased intensity of forest management) in the immediate vicinity of human settlements. For example, Munn et al. (2002) found that factors representing urbanization negatively affected the probability of timber harvest in forests of the south central US. Urban sprawl often leads to increases in road density and forest fragmentation (Sampson and DeCoster, 2000; Radeloff et al., 2005;), thereby increasing the amount of forest edge and changing species composition. Also, the goals of forest management may shift from timber production to recreational/aesthetic uses, and this may or may not increase the diversity of tree species. On the other hand, use of forests for recreation can also lead to land degradation. Finally, forests are affected by air pollution, which may influence the mix of species and their diversity when located near human settlements with higher populations.

In order to estimate the effects of economic development and population growth on the environment, as well as the effectiveness of conservation strategies and policies, statistically sound quantitative estimates of trends in diversity are needed on the regional and national levels (Stapanian et al., 1997). Most empirical studies on ecosystem diversity were done in small areas using varied objectives and designs. When data from such studies are aggregated to regional or national levels they fail to provide unbiased estimates of known significance, primarily due to the inconsistency of underlying data sources.

Few studies have analyzed the impact of anthropogenic disturbances or different management regimes on the diversity of tree species either on a large spatial (state/province) or a long temporal (several decades) scale. Stapanian et al. (1997) evaluated the diversity of tree species in 14 US states using data from Forest Health Monitoring (FHM) plots. They found large variations in species richness across regions and significant negative effects of anthropogenic disturbances on species richness (except in the mixed western hardwoods in California). Stapanian and Cassel (1999) analyzed the diversity of tree species in Georgia and Alabama using FHM plots by forest type. Tree species richness was found to be significantly greater on undisturbed plots than on disturbed plots for all three forest types they considered. Additionally, they found that disturbance modifies the frequency of occurrence of individual tree species on sample plots.

Rosson and Amundsen (2004) analyzed tree species richness in Mississippi for the period 1967–1994 using periodic USDA Forest Service Forest Inventory Analysis (FIA) plots. They found that average species richness in Mississippi increased between 1967 and 1977, and then decreased over the following periods, reaching a level in 1994 similar (not significantly lower) to the original level in 1967. At the same time, mean species richness of the plots, which were

not disturbed by harvesting during all four survey periods, did significantly increase between 1967 and 1994. However, selection of undisturbed plots in this way limits the sample to stands which develop simultaneously through similar stages of ecological succession and makes observations incomparable over time. They concluded that the use of FIA data collected all over the US for a long period can effectively help to analyze the dynamics of forest diversity.

Polyakov and Teeter (2005) used stand-wise forest inventory data collected in 1996 and a Tobit regression technique to analyze the influence of forest management regime (zoning and restrictions on final harvesting) on the diversity of woody vegetation in the Sumy administrative province of Ukraine. They found that tree species diversity, as indicated by Shannon's index, is higher in stands where final harvesting is not permitted, *ceteris paribus*. Additionally, their analysis shows that diversity is affected by the tenure arrangements.

In this study we investigate the effect of forest management activities and urbanization on diversity of tree species in Alabama forests over the period 1972 to 2000 using data from FIA sample plots.

Data

Historically, FIA data have been collected on approximately a 10-year cycle for sample plots located on roughly a 5 by 5 km grid pattern. We used data from the inventories conducted in Alabama in 1972, 1982, 1990, and 2000. In 1972, 1980, and 1990, trees between 2.5 and 12.7 cm diameter at breast height (dbh) were sampled on circular micro-subplots of radius 2.07 m, while trees with dbh equal or larger than 12.7 cm were sampled on variable radius subplots with a 37.5 ft²/ac (8.61 m²/ha) basal area factor prism. FIA sample plots in the 1972 and 1990 inventories consist of ten variable radius subplots and three fixed radius micro-subplots. FIA sample plots in the 1982 inventory consist of five variable radius subplots and three fixed radius

micro-subplots. Each 2000 FIA sample plot consists of four circular subplots of radius 7.32 m for trees greater than or equal to 12.7 cm dbh, and four circular micro-subplots of radius 2.07 m for trees between 2.5 and 12.7 cm dbh. In order to obtain consistent measures of tree diversity, which is a function of sample size, we selected a sub-sample of trees on each plot which would make the sample designs equivalent for all four inventory periods. For all FIA inventories, we included trees between 2.5 and 12.7 cm dbh from the first three micro-subplots. For the 1972, 1982, and 1990 FIA inventories, we included trees greater than or equal to 12.7 cm dbh from the first five variable radius subplots. For the 2000 FIA inventory, we randomly sampled trees greater than or equal to 12.7 cm dbh from all four fixed radius subplots with the sampling probability proportional to the squared diameter of the trees to obtain a sampling intensity equal to that of five variable radius plots. All expansion factors were adjusted accordingly. Furthermore, we excluded sample plots with no trees on any of the subplots.¹

To characterize diversity of tree species, we used two indices: species richness (R), which is the number of tree species found on a sample plot, and Shannon's index (Shannon and Weaver, 1963; Odum, 1971), calculated by the formula:

$$H' = - \left(\sum_{i=1}^n p_i \ln p_i \right),$$

where H' is Shannon's index and p_i are the proportions of the individuals of each of the species within the system. We used expansion factors for individual trees, indicating the number of trees

¹ We would like to thank Andrew J. Hartsell, FIA Research Forester, Knoxville, Tennessee, for reviewing our procedures to make sampling designs comparable across FIA inventories.

per unit area a particular tree represents, to calculate the proportion of individuals for each tree species. The distinctive feature of Shannon's index is that it allows us to characterize simultaneously the number of species and the variation of shares of individual species within an ecosystem. The value of the index increases with increasing numbers of species, achieving a maximum value when shares of individual species are distributed evenly.

To explore the effect of urbanization on the diversity of tree species, we use a population gravity index, which accounts for the combined influence of the size and proximity of populated places to the FIA plots. In this study we used a functional form of the gravity index initially developed in regional science as:

$$G_i = \sum_k \frac{P_k}{D_{ki}^2},$$

where G_i is the population gravity index for the point i , P_k is the population of populated place k , and D_{ik} is the distance between point i and populated place k . To calculate the population gravity index for a sample plot we use all census designated populated places within a 100 km radius. The data on population size of the US Census designated populated places was taken from the 1970, 1980, 1990, and 2000 decennial censuses conducted by the Census Bureau of the US Department of Commerce. Descriptive statistics of the quantitative explanatory variables used in the models are presented in Table 1.

Table 1. Descriptive statistics of quantitative explanatory variables

Variable	Minimum	Maximum	Mean	Std dev
Shannon's index all trees	0.00	2.55	1.26	0.52
Shannon's index trees 2.5–12.7 cm dbh	0.00	2.46	0.89	0.58
Shannon's index trees \geq 12.7 cm dbh	0.00	2.21	0.93	0.57
Species richness all trees	1.00	16.00	5.78	2.60
Species richness trees 2.5–12.7 cm dbh	0.00	12.00	3.17	1.93
Species richness trees \geq 12.7 cm dbh	0.00	12.	3.74	2.14
Population gravity index	34.80	22899.26	381.69	216.88
Stand volume, m ³ ha ⁻¹	0.00	455.35	100.55	65.65
Quadratic mean dbh, cm	2.51	96.64	12.80	5.44
Elevation, m	1.00	557.00	141.44	91.14
Slope, %	0.00	99.00	11.00	11.15
Latitude, °	30.28	35.00	32.75	1.13

Method

We used regression analysis methods to study the relationship between diversity of woody vegetation and natural and anthropogenic factors, including urbanization. Shannon's index is modeled as a function of fixed year effects, a vector of explanatory variables, and an error term representing the unobserved factors influencing the explanatory variables and assumed to be random. Because Shannon's index is observed on the same sample plot at several points in time, there is a possibility that unobserved factors influencing the dependent variable are correlated over time for a particular sample plot. In order to account for temporal correlation between observations of the same sample plot in the panel data, we use a random effects model:

$$H'_{it} = \alpha_t + \boldsymbol{\beta}' \mathbf{x}_{it} + \nu_i + \varepsilon_{it},$$

where α_t are fixed year effects, \mathbf{x}_{it} is a vector of stand attributes varying with time, $\boldsymbol{\beta}$ is a vector of regression coefficients, $v_i \sim N(0, \sigma_v^2)$ are normally distributed random effects for individual plots, and ε_{it} is an error term.

Species richness is a discrete count variable restricted to positive values and is properly modeled using a Poisson-lognormal model (Vincent and Haworth 1983; Teira and Peco 2002; Royle 2004). We assume that, conditional on their respective means λ_{it} , the number of observed species on a sample plot i in year t follows a Poisson distribution: $R_{it} \sim \text{Poisson}(\lambda_{it})$. We model the mean counts λ_{it} as being dependent on attributes of the stand where the plot is located and changing with time. The explanatory variables are linked with the mean via a log link function. We employ a random effects Poisson model, which can account for temporal correlation between observations of the same sample plot in the panel data:

$$\log(\lambda_{it}) = \delta_t + \boldsymbol{\gamma}' \mathbf{x}_{it} + u_i,$$

where δ_t are fixed year effects, \mathbf{x}_{it} is a vector of stand attributes varying with time, $\boldsymbol{\gamma}$ is a vector of regression coefficients, and $u_i \sim N(0, \sigma_u^2)$ are normally distributed random effects for individual plots.

Natural factors determining or affecting diversity are elevation, slope, latitude (as a proxy for the gradient of physiographic regions), site class (1 to 6 with site class 1 indicating productivity greater than $15.7 \text{ m}^3\text{ha}^{-1}$ and site class 6 indicating productivity between 1.4 and $3.5 \text{ m}^3\text{ha}^{-1}$), forest type grouping (coniferous, oak-pine, bottomland deciduous, and upland deciduous), stand volume and quadratic mean dbh. Anthropogenic factors influencing diversity include the population gravity index (the measure of urbanization), ownership class (public or

private), stand origin (planted or natural), and a dummy indicating whether the stand underwent any harvesting activity since the previous inventory. We used a semi-log specification (quantitative explanatory variables in log form) because distributions of most of the quantitative explanatory variables are heavily skewed towards zero.

Results

We estimated random effects OLS models for Shannon's index (Table 2) and random effects Poisson models for species richness (Table 3) for trees of all diameters and also separate models for trees 2.5–12.7 cm dbh and trees ≥ 12.7 cm dbh. As a measure of goodness of fit we used standard coefficient of determination for the random effects OLS models and deviance R-squared (R_d^2) (Cameron and Windmeijer 1996) for the random effects Poisson model:

$$R_d^2 = 1 - \frac{\sum_i y_i \log(y_i / \hat{\lambda}_i)}{\sum_i y_i \log(y_i / \bar{y})}$$

where y_i is the dependent variable, \bar{y} is the mean of the dependent variable, and $\hat{\lambda}_i$ is predicted y_i . The estimated models are highly significant and provide reasonable goodness of fit as indicated by R^2 and R_d^2 . Models estimated separately for trees 2.5–12.7 cm dbh and trees ≥ 12.7 cm dbh show better goodness of fit. The reason for this is that some factors affect diversity of smaller and larger trees differently. In the rest of this section we will analyze the results of the regression models, starting from the effects of natural factors, and pointing out if there are differences between the effects on diversity of smaller and larger trees as represented by the species richness and species diversity index values.

Table 2. Estimates of random effects linear regression of tree species diversity (Shannon index) in Alabama forests.

N=10457

	All trees		Trees 2.5–12.7 cm dbh		Trees \geq 12.5 cm dbh	
Intercept	-4.521‡	(0.748)	-1.993‡	(0.737)	-2.974‡	(0.746)
Log latitude	1.857‡	(0.224)	1.453‡	(0.221)	0.922‡	(0.224)
Log elevation	0.029‡	(0.011)	0.019*	(0.010)	0.039‡	(0.011)
Log slope	0.003‡	(0.001)	0.004‡	(0.001)	0.004‡	(0.001)
Type pine	-0.159‡	(0.012)	-0.041‡	(0.013)	-0.316‡	(0.012)
Type oak-pine	0.020*	(0.012)	0.038‡	(0.012)	-0.038‡	(0.012)
Type bottomland hardwoods	-0.047‡	(0.018)	-0.060‡	(0.018)	0.006	(0.018)
Site class 1	0.057	(0.073)	0.141*	(0.080)	-0.019	(0.074)
Site class 2	0.194‡	(0.026)	0.164‡	(0.028)	0.146‡	(0.027)
Site class 3	0.132‡	(0.022)	0.119‡	(0.023)	0.135‡	(0.022)
Site class 4	0.115‡	(0.020)	0.112‡	(0.022)	0.100‡	(0.020)
Site class 5	0.072‡	(0.020)	0.084‡	(0.021)	0.060‡	(0.020)
Log quadratic mean dbh	-0.323‡	(0.013)	-0.961‡	(0.014)	0.257‡	(0.013)
Log mean volume	0.052‡	(0.002)	0.060‡	(0.002)	0.045‡	(0.002)
Log mean volume \times Planted	-0.009‡	(0.002)	0.001	(0.002)	-0.043‡	(0.002)
Log population gravity index	-0.032‡	(0.006)	-0.029‡	(0.006)	-0.028‡	(0.006)
Harvested	-0.219‡	(0.010)	-0.228‡	(0.011)	-0.210‡	(0.010)
Planted	-0.259‡	(0.015)	-0.168‡	(0.016)	-0.312‡	(0.015)
Ownership public	0.039*	(0.021)	0.007	(0.021)	0.013	(0.021)
Ownership forest industry	0.010	(0.012)	0.020	(0.013)	-0.027†	(0.012)
Year 1972	-0.151‡	(0.010)	-0.098‡	(0.012)	-0.116‡	(0.011)
Year 1982	0.063‡	(0.010)	0.106‡	(0.011)	0.034‡	(0.010)
Year 1990	0.031‡	(0.010)	0.071‡	(0.011)	0.004	(0.010)
R ²	0.322		0.423		0.441	

Notes: standard errors in parentheses;

* significant at 10%; † significant at 5%; ‡ significant at 1%.

Table 3. Estimates of random effects Poisson regression of tree species richness in Alabama forests. N=10457

	All trees		Trees 2.5–12.7 cm dbh		Trees \geq 12.5 cm dbh	
Intercept	-2.909‡	(0.668)	-1.068	(0.825)	-1.951‡	(0.728)
Log latitude	1.517‡	(0.199)	1.476‡	(0.246)	0.416†	(0.216)
Log elevation	0.016*	(0.009)	0.022*	(0.012)	0.041‡	(0.010)
Log slope	0.006‡	(0.001)	0.007‡	(0.002)	0.006‡	(0.001)
Type pine	-0.280‡	(0.013)	-0.051‡	(0.016)	-0.487‡	(0.016)
Type oak-pine	-0.043‡	(0.014)	0.033*	(0.018)	-0.103‡	(0.018)
Type bottomland hardwoods	-0.018	(0.019)	-0.040	(0.026)	-0.041†	(0.021)
Site class 1	0.060	(0.117)	0.121	(0.208)	-0.122	(0.116)
Site class 2	0.234‡	(0.033)	0.222‡	(0.039)	0.032	(0.042)
Site class 3	0.185‡	(0.028)	0.164‡	(0.032)	0.031	(0.035)
Site class 4	0.147‡	(0.026)	0.143‡	(0.030)	0.033	(0.034)
Site class 5	0.090‡	(0.027)	0.091‡	(0.031)	0.023	(0.034)
Log quadratic mean dbh	-0.340‡	(0.016)	-1.347‡	(0.019)	0.039*	(0.022)
Log mean volume	0.085‡	(0.002)	0.087‡	(0.002)	0.416‡	(0.013)
Log mean volume \times Planted	-0.031‡	(0.004)	-0.005	(0.003)	-0.101‡	(0.036)
Log population gravity index	-0.030‡	(0.005)	-0.033‡	(0.007)	-0.019‡	(0.006)
Harvested	-0.227‡	(0.014)	-0.299‡	(0.019)	-0.090‡	(0.020)
Planted	-0.175‡	(0.021)	-0.134‡	(0.021)	0.059	(0.170)
Ownership public	0.036*	(0.021)	0.018	(0.026)	-0.046†	(0.021)
Ownership forest industry	0.017	(0.013)	0.033†	(0.015)	-0.033†	(0.015)
Year 1972	-0.148‡	(0.017)	-0.121‡	(0.021)	-0.115‡	(0.023)
Year 1982	0.058‡	(0.018)	0.157‡	(0.022)	0.068‡	(0.025)
Year 1990	0.032*	(0.017)	0.110‡	(0.022)	0.012	(0.024)
R_d^2	0.396		0.483		0.614	

Notes: standard errors in parentheses;

* significant at 10%; † significant at 5%; ‡ significant at 1%.

Both Shannon's index and species richness increase from south to north and with higher elevation. Species diversity patterns in relation to latitude and elevation are well documented

(e.g., Pianka 1966). Slope is positively related to diversity in all models, because stands on steeper slopes have greater variation of microecotypes and are less accessible to anthropogenic disturbances.

Forest type is represented by three dummies (coniferous, oak-pine, and bottomland deciduous) with upland deciduous being the reference type. The regression coefficients of these dummies indicate differences in species diversity or species richness for a particular forest type in relation to species diversity or species richness of upland hardwoods. As expected, pine stands have the lowest diversity and species richness for both smaller and larger trees. Species diversity of smaller diameter trees is lower in bottomland hardwoods than in upland hardwoods, while species richness is not significantly different. Species diversity of oak-pine stands is higher for both smaller and larger diameter trees. Species richness of larger diameter trees is lower while species richness of smaller diameter trees is higher in oak-pine stands than in upland hardwood stands. Species richness of larger diameter trees is lower in bottomland hardwoods than in upland hardwoods.

Site class is represented by five class specific dummies (1 to 5) with 6 being the reference site class. Results suggest that both species diversity and species richness show an increasing trend from site class 6 to site class 2 except for species richness of larger trees where it is not significantly different among site classes. A similar relationship between species diversity and site quality was reported by Polyakov and Teeter (2005). Contrary to our expectations that highest species diversity and species richness values would occur on site class 1 plots (highest productivity class), we found that there is no significant difference in diversity index and species richness for plots on site class 1 lands relative to site class 6. This could be explained by the fact

that there are very few stands in the highest productivity class (site class 1) and they are mostly represented by the bottomland hardwood forest type.

Diversity increases with stand volume. When controlled for the mean dbh, greater volume indicates a larger number of trees and better productivity. Quadratic mean dbh has a positive relationship with both species diversity and species richness for the larger trees component (≥ 12.7 cm dbh) and a negative relationship with species diversity and species richness for the smaller trees component. Larger dbh indicates older stands where under-story may be suppressed.

As expected, artificially regenerated stands, as well as stands that underwent harvest, are less diverse, which is consistent with findings of Stapanian et al. (1997), Rosson and Amundsen (2004), and Polyakov and Teeter (2005). However, there is no significant difference in species richness of the larger trees component between natural and planted stands of the same type.

Publicly owned forests are more diverse than non-industrial private forests (NIPF, which is the reference ownership class) as indicated by species diversity and species richness calculated for trees of all sizes. This is consistent with the assumption that public forest management agencies are mandated to maintain biodiversity and put more emphasis on environmental, aesthetic, and recreational uses. However, the magnitudes of the coefficients are much smaller than those of the harvested and planted dummies, and the effect is not consistent across the larger and smaller tree components. Our a priori expectation was that forests owned by forest industry are less diverse due to higher intensity of forest management. Results of our models indicate that both species diversity and species richness of trees in forests owned by forest industry are not significantly different than those in non-industrial private forests, although diversity of the larger tree component is marginally lower. The regression coefficients for public and industry

ownership classes in our models indicate effects of respective ownership classes when all other effects are controlled for. However, ownership determines management intensity, in particular frequency of removals and proportion of pine plantations. If these variables are part of the model, the remaining effect of ownership class is negligible. Table 4 presents means and standard errors of means of some of the characteristics of forests by ownership class. It indicates that forest industry has the highest and public forests have the lowest proportion of plantations. Forest industry has the highest proportion of coniferous forests, while the proportion of public forests that underwent removals is only half that of forest industry and NIPF. Furthermore, the average population gravity index indicates that public forests are located in places with greater population pressure, while forests owned by forest industry are located in more remote places. This relationship is also illustrated by the curves of proportions of forest industry and public forests as functions of population gravity index shown on Figure 1. All these factors (except population gravity index) result in the highest species diversity for public forests and the lowest diversity of trees in the forests owned by forest industry. In particular, public forests have on average one more tree species than forests of forest industry.

Table 4. Characteristics of biodiversity and anthropogenic disturbance by ownership class

	NIPF		Industry		Public	
Shannon index	1.27	(0.01)	1.18	(0.01)	1.36	(0.02)
Species richness	5.84	(0.03)	5.33	(0.06)	6.33	(0.09)
Proportion of planted	0.10	(0.00)	0.31	(0.01)	0.05	(0.01)
Proportion of stands that underwent removals	0.34	(0.01)	0.35	(0.01)	0.19	(0.01)
Proportion of coniferous	0.31	(0.01)	0.49	(0.01)	0.35	(0.02)
Population gravity index	402	(10)	259	(10)	510	(39)

Note: standard errors in parentheses

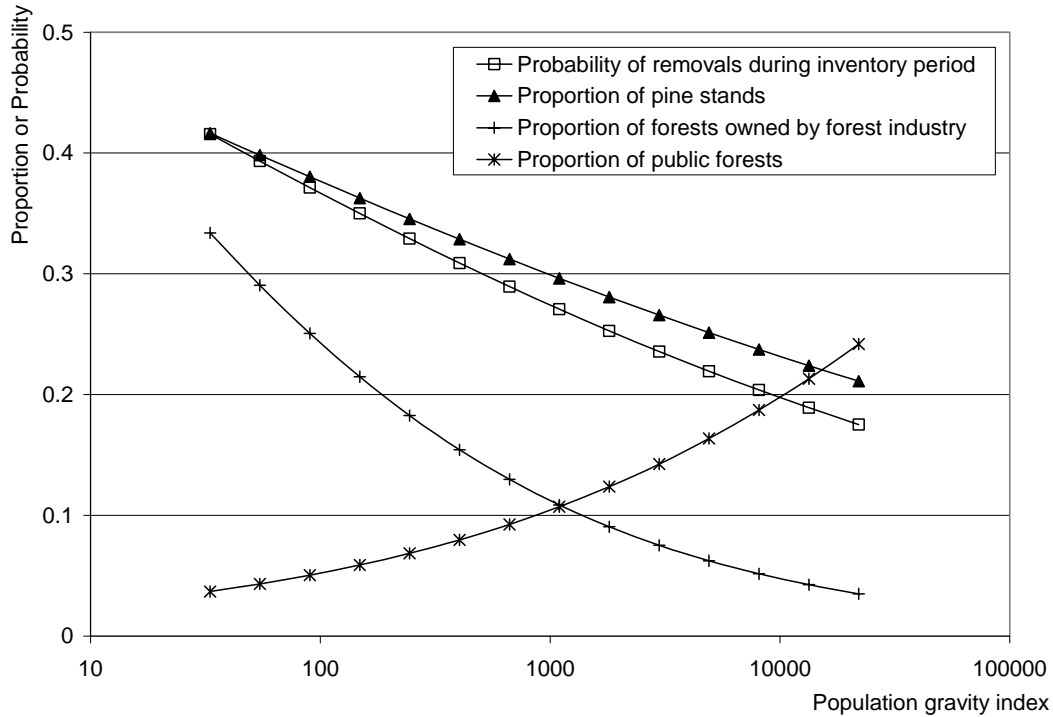


Figure 1. Changes in probability of removals, proportion of pine stands, proportion of forests owned by forest industry, and proportion of public forests as functions of population gravity index.

The “year” dummy variables show that, *ceteris paribus*, both species diversity and species richness were lower in 1972 and higher in 1982 and 1990 relative to 2000. This trend is consistent with the trend of species richness reported by Rosson and Amundsen (2004) for Mississippi during period 1967–1994. When we ran the models on a dataset with artificially regenerated stands and stands that underwent harvesting (disturbed stands) excluded, the results indicate that there is no statistically significant difference between species richness and species diversity in 1990 relative to 2000.

Finally, we find that urbanization pressures as indicated by population gravity index adversely impacts both diversity and richness of tree species across all models consistently. This is a partial effect of population pressure on measures of biodiversity, i.e., the effect when all other variables are controlled for. If all other factors are held equal, the difference in species

richness between a plot, where population gravity index is the highest (e.g., Birmingham, AL) and a plot where population gravity index is lowest (e.g., north-west of Clarke county, AL) would be approximately one species. The difference in Shannon index between plots in these locations would be 0.24, which is equivalent to the difference between a plot with three tree species evenly represented and a plot where one species is represented by 70% of individuals and the other two species are represented by 15% of individuals each. However, other factors are unlikely to be equal between plots at those locations since many of the explanatory variables, in particular, factors related to forest management activity, are influenced by population pressures.

For example, probability of stand undergoing removals and proportion of pine stands increase as population pressure indicated by population gravity index decreases (Figure 1). The proportion of planted stands increases over time following a similar relationship with PGI (Figure 2). This indicates that population pressure inversely affects forest management activity and corresponds with findings of Wear et al. (1999) and Munn et al. (2002). Because forest industry manages forest more intensively, forests owned by forest industry are concentrated in locations with lower population pressure (Figure 1). At the same time, proportion of publicly owned forests increases with an increase in population pressure, because recreation is one of the important management objectives of public forests, therefore they are located closer to populated places.

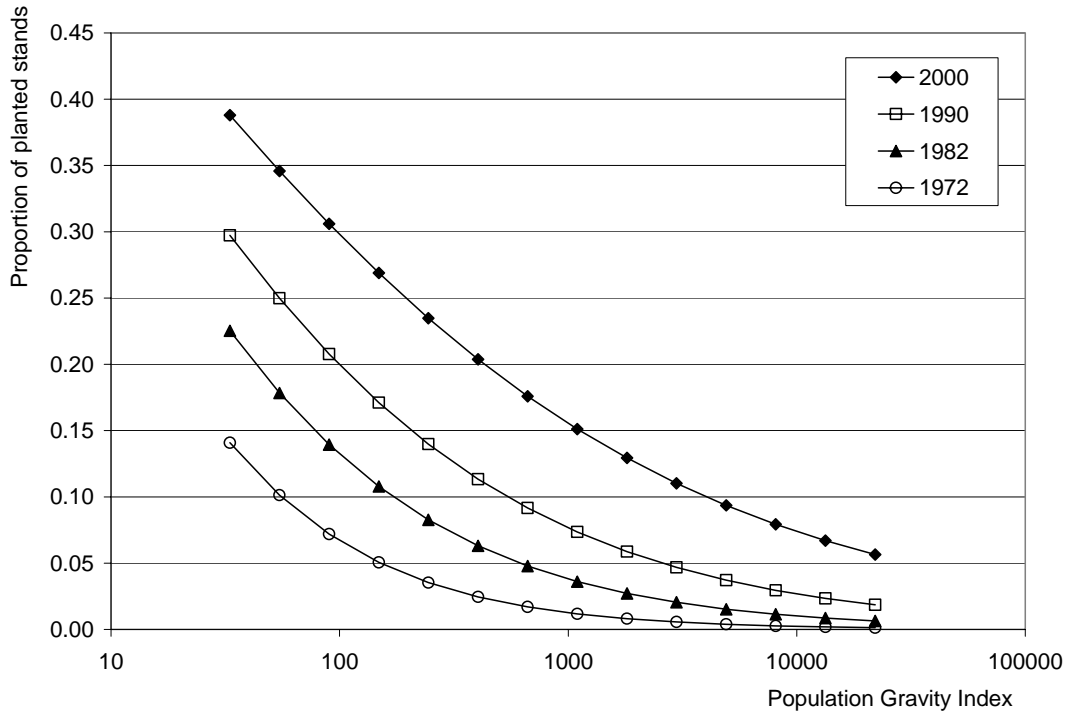


Figure 2. Change of the proportion of planted stands as a function of population gravity index by inventory period.

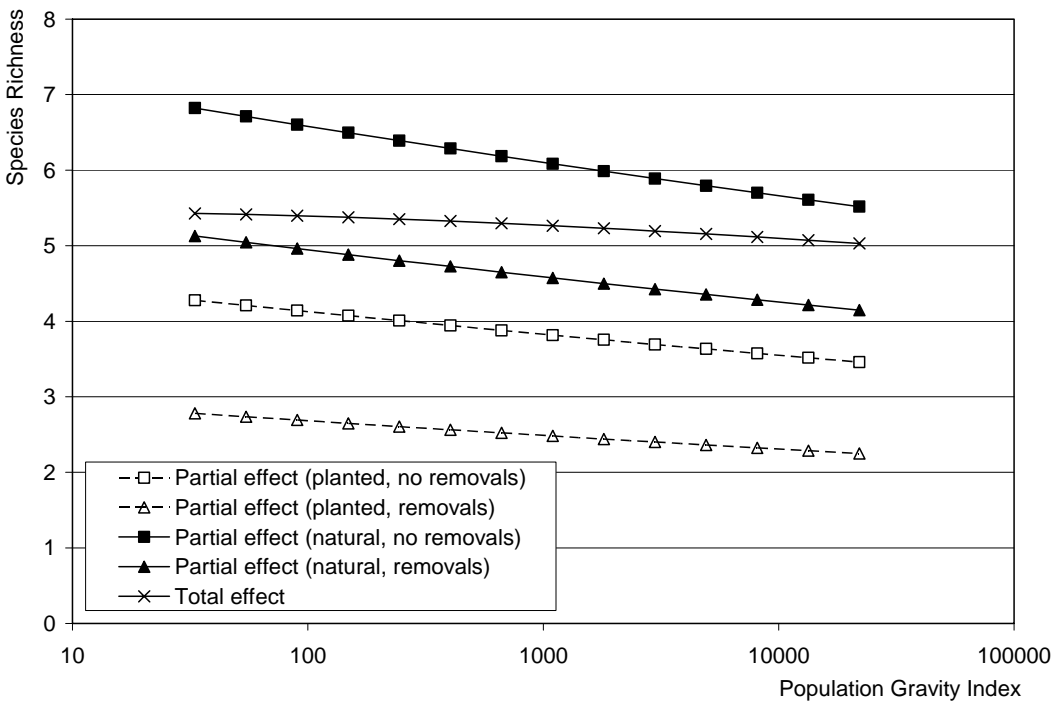


Figure 3. Partial and total effects of population gravity index on species richness index.

While artificially regenerated pine stand, stands that underwent removals, and stands owned by forest industry have lower diversity of trees, the proportion of stands in these categories increases with decreases in population pressure. As a result, the total effect of population pressure on forest diversity is lower. To demonstrate this, we plotted predicted species richness (Figure 3) and Shannon's index (Figure 4) as functions of population gravity index. To show partial effects, we held "Planted" and "Removals" variables equal to either 0 or 1 (planted or natural and harvested or not harvested) and all other variables equal to sample means. To show the total effects, we held the variables related to management intensity (harvested, planted, forest types, and ownership classes) equal their means conditional on population gravity index, and the rest of the variables equal their unconditional means. As can be seen from the Figures 3 and 4, the total effect (the slope of the curve) is lower than the partial effects. In particular, the difference in species richness between a plot located where population gravity index is the highest and a plot located where population gravity index is lowest would be approximately 0.4 species. Thus, while affecting tree species diversity in two opposite directions (negative direct effect and positive indirect effect), the net effect of population pressure on tree species diversity is negative.

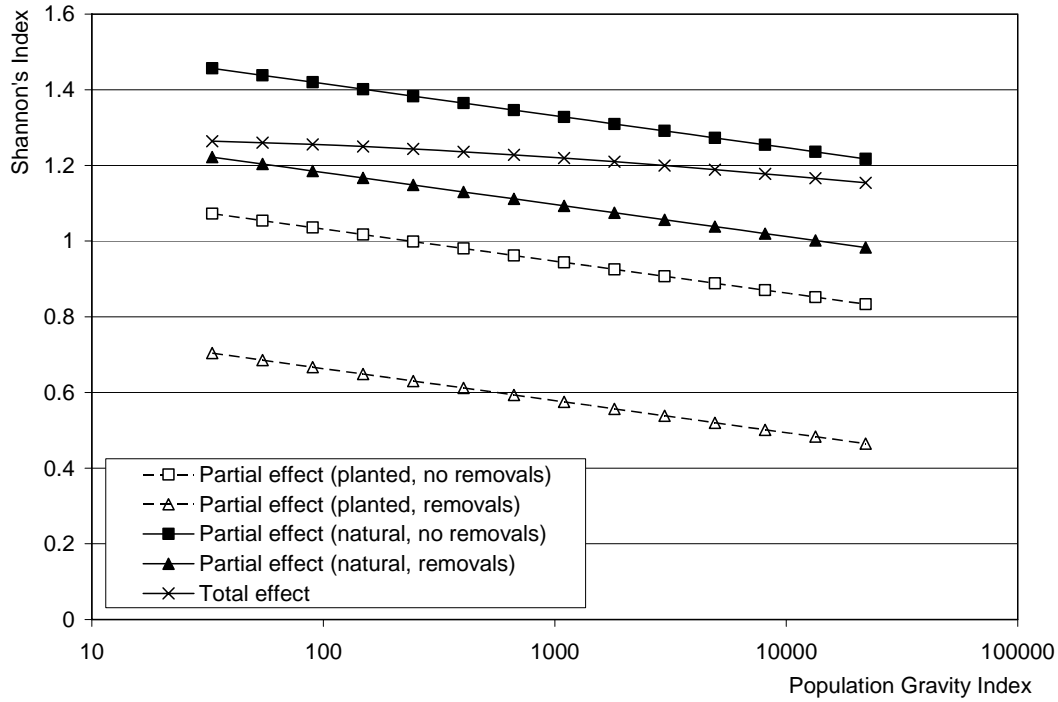


Figure 4. Partial and total effects of population gravity index on Shanon's index.

Conclusion

This study attempts to analyze the effect of urbanization on diversity of tree species in forests of Alabama. To quantify urbanization, the model utilizes a population gravity index, which reflects both population density and proximity to populated places. Our results indicate that urbanization affects tree species diversity in two ways. Directly, it negatively impacts tree species diversity in Alabama forests. At the same time, urbanization adversely affects intensity of forest management, which negatively impact species diversity. The combined effect of urbanization on species diversity is negative; however, its magnitude is lower than that of the direct effect.

However, when interpreting these results, several considerations should be made. First of all, urbanization brings several effects, and each of them could impact diversity of tree species in different ways. First, although forests provide important habitat for many species of plants and animals, the effect of urbanization on the diversity of different groups of species could differ from the effect of urbanization on the diversity of tree species. Second, formulation of the gravity index implies an assumption that the effect of population is inversely related to the square of the distance to a populated place (a routine assumption in regional science). However, the actual relationship could be non-monotonic, as was reported by Munn et al. (2002).

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