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# **Land Use Change and Ecosystem Valuation in North Georgia**

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## **Abstract**

This study seeks to forecast land use change in a North Georgia ecosystem, model related water quality changes and estimate the economic value of the same using Benefit Transfer. Upper Chattahoochee River Basin which is the North Georgia ecosystem in question is a source of water, recreational and ecological amenities. Rapid population growth in Georgia has led to increased encroachment on this ecosystem in recent years threatening the future ability of the basin to provide these environmental commodities. We use econometric, time series and structural time series models land use and benefit transfer to estimate willingness to pay. We find that population growth will impact negatively on forestry and farmlands. In addition, the people of the Upper Chattahoochee River Basin would be willing to pay a lower bound value between USD 15,785,740 and USD 16,141,230 per year to create and maintain quality standards for fishing and drinking water supply.

Keywords: Ecosystem, Economic value, North Georgia, land use, land use change, fish, water quality, structural time series, willingness to pay, benefit transfer, forecasting, vector autoregression, Upper Chattahoochee River.

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# **Land Use Change and Ecosystem Valuation in North Georgia**

## **Introduction**

Ecosystems play an important role in providing commodities (functions and services, beneficial to society. Ecosystem economic values reflect tradeoffs made to obtain more or prevent loss of such ecosystem resources. There is a general understanding that land use and land use change affects environmental quality and the level of ecosystem resources. This study combines land use modeling and ecosystem valuation to estimate the economic value of land use change in a north Georgia ecosystem.

Valuing environmental impacts has become increasingly important with the increase in public awareness of environmental issues, government requirements, and the rising scarcity of environmental commodities. The impact of land use change on ecosystem values could be negative or positive, depending on which land use takes what portion of the land. Normally though, land moves from farms and forests to urban and industrial areas, with negative consequences for runoff, and biological and chemical pollutants.

There has been a major movement towards ecosystem-based (including watershed-based) management of natural resources over the last decade (Lambert, 2003). This can be attributed to the realization that ecosystems functions and services are so intertwined that human activities on one service affect another. In line with this trend, economists have in the last decade begun to place emphasis on valuing entire ecosystems as opposed to individual services. This shift seems

to be prompted by the growing awareness that ecosystem and watershed services are seldom provided in isolation. Fragmenting ecosystem services might lead to overvaluation or under-valuation of the ecosystem.

Economic valuation of watersheds and other ecosystems is complicated by market failure and characterized by three main factors. The services are public goods, they are affected by externalities and have property rights that are poorly or so broadly defined that there is no incentive for conservation.

Few studies have been documented that combine land use and ecosystem valuation. Most studies major either on land use or some aspect of ecosystem valuation.

Generally, analysis of land use estimates the relationship between the share of land allocated to alternative uses and factors that determine such allocation. The researcher then uses the resulting model(s) to predict future land use by plugging in the forecasted determinants. Land use and land use change have mostly been modeled using econometric and time series methods and most recently, structural time series.

Recently, land allocation studies based on econometric modeling have been documented by a number of authors including Alig(1986), Parks, Murray and Maulding (1994), Miller and Plantiga (1999), Plantinga, Maulding and Miller (1999). Econometric and time series based studies include Duffy, Shaishali, and Kinnucan (1994), Houston et al. (1999), Wu and Segerson (1995), Plantiga (1996), Lichtenberg (1989), and Banerjee (2004). Structural Time Series Models (STSM) have

also been used in estimating farm acreage response models (Houston et al., 1999; Adhikari, 2004).

In the past, economists have used diverse approaches to valuing ecosystems. Many have used a combination of methods, including market prices, to value marketed components of ecosystem services and direct and indirect techniques of valuing public goods. Except for market prices, other methods of valuing ecosystem components or entire ecosystems are costly both in time and money.

Benefit (or benefits) Transfer is a set of techniques used for estimating the value of public goods whenever it is not practical to collect primary data on which to base economic valuation (Bergstrom and Civita, 1999). Benefits Transfer has been used to value ecosystems in numerous studies, including Constanza et al (1997), Verna (2000), Toras (2000) and Kramer et al (1997). We apply benefit transfer techniques to the valuation of a watershed ecosystem within the Upper Chattahoochee River Basin in North Georgia.

The Chattahoochee River rises in North Georgia and flows for 434 miles to the Florida. The river is important as the primary source of drinking water for the city of Atlanta and more than 4.1 million people in the states of Georgia, Florida and Alabama. In North Georgia, the Upper Chattahoochee River Basin (UCRB) ecosystem incorporates a national forest, and major recreational areas. The UCRB also contributes to provision of water for agricultural, industrial, recreational and sewage disposal purposes in a number of Georgia counties.

The population of North Georgia has continued to rise drastically over the last two decades with associated conversion of land from forests and farms to urban development. In 1974 only 3% of the non-government forest land in our area of study (policy site) was under residential/urban use. By 2005, 13% of the land was under urban use and the proportion of land converted to urban development has doubled every 10 years between 1984 and 2005. In the entire river basin, total farmland has decreased over the years, while confined animal operations and poultry production has increased, with increased demand for water and the risk of water contamination. In 1995, 49% of stream miles in the UCRB was contaminated with fecal coliform bacteria (FCB). Models of future land use and land use change could provide information on how the aforesaid changes would affect the value of ecosystem services and functions in this and other ecosystems.

The objective of this paper is to forecast land use change in Habersham and white counties of the Upper Chattahoochee river basin in North Georgia (the policy site), to model associated water quality changes and estimate the economic value of the ecosystem with emphasis on water quality.

The study on which this paper is based proceeded as follows: First, we sought to simulate three likely land use scenarios and resulting changes in ecosystem services and functions from a baseline, set at year 2005. Projections of land use and state of the ecosystem are then done to the year 2030. The simulated land use scenarios will be based on the following projections:

- a. The "High Growth" scenario, for instance assuming a high population growth rate for the policy site;
- b. The moderate growth scenario, with continuation of "past trends" in population and land use;
- c. The "Managed Growth" scenario, e.g. with limited population growth.

The second step was to estimate economic value of the changes in the ecosystem using benefits transfer techniques, with special emphasis on water quality (as an ecosystem service).

The remainder of the paper is arranged as follows: In the next section we discuss a review of literature on land use and ecosystem valuation (with emphasis on benefit transfer). We then discuss the theoretical frameworks for modeling land use and ecosystem valuation and outline our methodology for estimating and forecasting land allocation for the ecosystem. The next section deals with empirical analysis of land use and a discussion of results of the land use model followed by a forecast of water quality based on the said land use changes. Then we apply benefit transfer to value water quality in the policy site. The last section has the summary of our findings, conclusions and recommendations.

## **Review of Literature**

The majority of land use models make use of county level data. This is understandable as farm level data is difficult to come by or would be extremely expensive to compile.

Econometric analysis of optimal land allocation has been carried out by a number of authors including Miller and Plantiga (1999); Plantinga, Maulding and Miller (1999). The aforesaid studies have applied econometric models to estimate aggregate (such as farm and forest as opposed to crop/enterprise level) land allocation. Ahn, Plantinga and Alig (2000), document a comprehensive model of forestland allocation at the aggregate level. The aforesaid studies assume that land shares follow a logistic distribution and estimate econometric panel data models of land use shares, normalizing over one land use.

Land allocation, and the many factors affecting it, change over time. This makes land use (and land use change) a suitable candidate for time series and structural time series modeling. Farm acreage response/farm land allocation among (different) crop enterprises has been estimated using econometric and time series models (Duffy, Shaishali, and Kinnucan 1994; Houston et al., 1999; Wu and Segerson, 1995; Plantiga, 1996; Lichtenberg, 1989; Banerjee, 2004). Structural Time Series Models (STSM) pioneered by Harvey (1989) have seen recent use in estimating farm acreage response models (Houston et al., 1999; Adhikari, 2004). The STSM has the advantage of being able to capture structural and technological change, which are either overlooked or assumed to be deterministic in conventional econometric and time series modeling. Despite these benefits, the STSM has not been exploited much in aggregate acreage response modeling.

Ecosystem valuation has for a long time been done using traditional techniques of valuing non-marketed goods and services



including direct and indirect techniques. Direct (revealed preference) techniques rely on actual expenditure to *reveal* the preferences of individuals for environmental goods or services associated with the expenditure (e.g. the added value of a house near a forest, or the cost of traveling to a national park). These techniques include hedonic pricing (HPM) and travel cost method (TCM). These methods are limited in that they can only capture use values.

Indirect (stated preference) techniques rely on questionnaires to elicit participant's response to questions that simulate a market situation. Indirect techniques have the advantage of being able to capture non-use values. The major one of these techniques is Contingent Valuation Method (CVM).

The Contingent Valuation Method (CVM) seems to be the most commonly used techniques of measuring the value of improvements in ecosystem or resource quality.

In the recent past, "benefit transfer" (BT), sometimes called "benefits transfer", is becoming increasingly useful as an approach to valuation of non-marketed public goods and service. Brookshire and Neill (1992) suggest that, "A benefit transfer is the application of monetary values obtained from a particular nonmarket goods analysis to an alternative or secondary policy decision setting".

Benefits Transfer Estimation (BTE) is gaining importance because of its usefulness whenever it may not be practical for an organization to collect data on which to base economic value estimation at short notice (Bergstrom and Civita, 1999), and in cases where a high degree of precision is not critical (Du, 1998). This approach reduces costs

(Kask and Shogren, 1994) and is therefore important during times of public funding cuts. It enables estimation within a shorter time than traditional methods, reducing the time it takes for policy makers to make informed decisions (Bingham, 1992). It is no wonder BTE has become a method of choice for ecosystem-wide valuation.

The range of approaches to estimating ecosystem services is almost as wide as the studies. Benefits transfer seems to be a common threat that links studies that estimate the value of entire ecosystems.

A number of studies have attempted to place value on ecosystems. The most notable and ambitious attempt was by Constanza et al. (1997). The authors used 100 existing studies (BT) to estimate the value of the world's ecosystem services and natural capital (stock that provides these services).

The said study estimates that the world's ecosystem services are on average worth US\$ 33 trillion ( between US\$ 16-54 trillion) annually about 1.8 times the current global Gross National Product(GNP) at 1994 US prices. The authors advocate for giving the natural capital stock adequate weight in the decision making process to avoid the detriment of current and future human welfare.

Verma (2000) used benefits transfer to value forests of the Himalal Pradesh state of India. The study used forest valuation in India and other countries to come up with economic values for the state forests.

Toras (2000) estimated the economic value of the Amazonian deforestation using data from past studies. The original studies the

author adopted used a mixture of market prices, direct and indirect methods. For instance, market prices were used to value marketable commodities like timber and foodstuffs; replacement costs were used to value nutrient loss due to soil erosion; TCM was used to value recreational benefits, CVM for valuing existence benefits etc. The author came then discounted the TEV of the Amazonian forest and arrived at a Net Present Value (NPV) of \$1175/ha/yr at 1993 prices.

Kramer et al (1997) estimated the value of flood control services resulting from protection of upland forests in Madagascar. They used averted flood damage to crops to estimate the value of the service. They placed the flood protection value of the watershed at \$ 126700.00, the amount of losses the community avoided from the presence of the forest park.

Alp et al (2002) applied BT to the estimation of the value of flood control and ecological risk reduction services provided by the Root River watershed (as the policy site) in Wisconsin. The study sites included Oak Creek and Menomonee River watershed both located in Milwaukee County, Wisconsin, most of which neighbors the policy site to the North. They observe that the sites are very close (geographically), were almost identical and were affected by the same problem. The authors suggest that their study findings could be used for the purpose of screening related projects.

Bouma and Schuijt (2001) documented a case study conducted by the World Wide Fund for Nature (WWF) to estimate the economic values of the natural Rhine River basin functions. The authors used market prices to estimate losses in fish production; and shadow pricing

techniques to estimate losses in provision of clean drinking water, existence values and natural retention capacity. The total economic value of the four ecosystem functions was estimated at USD 1.8 billion per year.

Loomis et al (2000) estimated the total economic value of restoring ecosystem services in an impaired river basin using CVM. The services in question were dilution of waste water, natural purification of water, erosion control, fish and wildlife habitat, and recreation. Results from contingent valuation interviews suggested a willingness to pay for additional ecosystem services ranging from \$ 25.00 per month to \$252 per year.

### **Theoretical Measures of Economic Value**

Most environmental "commodities" (goods, services, functions) can be viewed as public goods with no real market transactions take place. This makes it difficult to measure changes in the quantities of such commodities. Such commodities are mostly available in fixed unalterable quantities. Policy changes affecting such commodities result in changes in the consumer's bundle hence the consumer's welfare.

Following Verderberg, Poe and Powell(2001) Willingness to Pay (WTP) for improvement of an ecosystem commodity (such as water quality) for the  $i$ th individual at the  $j$ th site can be specified as:

$$(1) \quad WTP_{ij}^* = \omega_j(Q_{ij}^0, Q_{ij}^1, I_{ij}, H_{ij})$$

Where  $\omega_j$  represents the average valuation function for the  $j$ th site;  $Q^0$  and  $Q^1$  are pre and post-improvement quality levels say resulting from a change in land use;  $I$  is income,  $H$  represents a vector of other socio-economic characteristics.

BTE involves using information from prior research (study) site(s), to provide information for the policy site (p). We may direct transfer a single (mean or median WTP) value from the study to the policy site. We may alternatively transfer the estimated "benefit function" ( $\hat{\omega}_s(.)$ ) from the study to the policy site by plugging in the policy site characteristics into the study site function.

Whichever approach is chosen one has to make adjustments for differences between study and policy site particularly in regard to time (date of reference versus policy study) and income.

Benefits function transfer enables accounting for differences in physical and demographic characteristics between study and policy sites and is considered superior to fixed value transfer (Loomis, 1992). Nevertheless this approach is often impossible particularly because data documentation is often insufficient, and few studies are conducted with benefit transfer in mind so the data they provide is not necessarily amenable to benefit transfer ((Rosenberger and Loomis). Value transfer is therefore a more common approach.

Several authors including Rosenberger and Loomis discuss a number of conditions necessary to ensure effective and efficient benefits transfer estimation. The policy context should be thoroughly defined; the study site data should meet certain conditions for critical

benefits transfer; there has to be correspondence between study and policy.

### **Models of Land Use**

Following Wu and Segerson (1995) and Miller and Plantinga (1999) we develop a model of land allocation at the aggregate watershed (two-county) level, assuming profit or net benefit maximization under risk neutrality. Consider a land manager/owner who maximizes total restricted returns to  $A$  acres of land, by allocating the land optimally among  $i$  alternative uses ( $i = 1, \dots, n$ ). We use discounted (present value) benefits approach to account for the fact that returns to forestry are realized over long periods of time. The land allocation process can then be expressed as:

$$(2) \quad \max_{A_i} \sum_{i=0}^n \Pi_i(X)$$

Subject to,

$$(3) \quad \sum_{i=0}^n A_i = A$$

Where  $X$  = Matrix of exogenous variables;  $A_i$  = Acreage of the  $i^{\text{th}}$  land use;  $A$  = Total available acreage;  $\Pi_i$  = expected returns from land use  $i$ .

Solving the constrained profit maximization problem above gives us the optimal allocation to land use  $i$ , denoted by

$$(4) \quad A_i^* = f_i(X) \text{ for all } i=1, \dots, n$$

We can rewrite equation 4 from the land share perspective as follows:

$$(5) \quad S_i^* = \frac{A_i^*}{A} = \frac{f_i(X)}{A} = S_i(X)$$

where  $S_i^*$  = optimal expected share of land use i.

Analytically, equation 5 can be estimated as a flexible functional form for the restricted benefits function or for the acreage function and the implied share equations can then be derived (Moore and Negri, 1992; Shumway, 1983; Wu and Segerson, 1995). Alternatively one can estimate such a functional form for the share equations themselves.

We choose the later approach as it best represents the way we view land allocation and estimate the share equations assuming a logistic distribution of the error terms. This assumption is fairly common in the literature including Lichtenberg (1989) and Plantinga (1996), and has the advantage of ensuring the shares lie in the zero-one range. In addition, the logistic distribution outperforms other functional forms such as the Almost Ideal Demand System (AIDS) and the translog model.

Given equation 5, the share of use i at time t, can be expressed as follows:

$$(6) \quad S_{it}^* = \frac{\exp[f_i(X_t)]}{\sum_{i=0}^n \exp[f_i(X_t)]}$$

where  $\exp[ ]$  is the exponential function.

We sum up over 3 land use types namely farming, forestry and "urban" (industrial, commercial and residential). We select the urban

category as the normalizing land use alternative and rewrite the share equations as:

$$(7) \quad \ln(S_{it}^*/S_{0t}^*) = f_i(X_t) - f_0(X_t)$$

Since  $i=0$  is the normalizing land use, equation 7 reduces to

$$(8) \quad \ln(S_{it}^*/S_{0t}^*) = f_i(X_t)$$

which can be simplified as:

$$(9) \quad Y_{it}^* = \alpha_0 + \mu_t + \beta_i' X_{it} + e_{it}$$

where  $\alpha_0$  is the intercept;  $Y_{it}$  is the land share of use  $i$ , (over share of use 0);  $\beta_i'$  is a vector of parameters to be estimated;  $X_{it}$  is a vector of independent variables for land use  $i$ ,  $\mu_t$  is the time trend variable which could be deterministic, stochastic or absent altogether, and  $e_{it}$  is the error term.

Equation 9 is a typical econometric model representation. If the vector of dependant variables is made up of lagged dependant variables, then we have a time series model. Structural time series analysis on the other hand views changes in the dependant variables as resulting from structural or technological change which can be modeled using trend variables.

### **Structural Time Series Modeling**

A Structural Time Series Model (STSM) of land use is advantageous as it incorporates existing structural or technological change.

Most authors incorporate trend dummy variables in their models to capture the impacts of technological progress (Chavas and Holt, 1990;



Shideed et al., 1987). However, one limitation of these studies is that they assume a deterministic trend component in acreage response and specify the model with a time trend.

Harvey (1989) first proposed the Structural Time Series (STS) Model. Unlike traditional ARIMA models, the STSM is developed directly in terms of components of interest, such as trend, seasonal, cyclical, and residual or irregular components. The model allows the unobservable components to change stochastically over time. In the absence of the unobserved components, the STSM reverts to the classical regression model.

Structural time series modeling can be carried out primarily as time series modeling, without including explanatory variables. Incorporating explanatory variables with the stochastic components results in a mixture of time series and econometric model (Koopman et al., 2000), which broadens the scope of the STSM.

Consider the following STS land allocation model:

$$(10) \quad Y_{it} = \nu_t + \delta'_i X_{it} + \varepsilon_{it}$$

Where  $\alpha_0$  is the intercept;  $Y_{it}$  is the land share of use  $i$ ;  $\delta'_i$  is a vector of parameters to be estimated;  $X_{it}$  is a vector of explanatory variables for land use  $i$ ,  $\nu_t$  is the trend component, and  $\varepsilon_{it}$  is the white noise disturbance term.

The simple STSM without explanatory variables may be represented by,

$$(11) \quad Y_{it} = \nu_t + \varepsilon_{it}$$

If the trend is stochastic, the trend component may be represented by,

$$(12) \quad v_t = v_{t-1} + \beta_{t-1} + \eta_t$$

$$(13) \quad \beta_t = \beta_{t-1} + \xi_t$$

where  $\eta_t \sim \text{NID}(0, \sigma_\eta^2)$  and  $\xi_t \sim \text{NID}(0, \sigma_\xi^2)$

Equations (12) and (13) represent the level and the slope of the trend, respectively;  $v_{t-1}$  is a random walk with a drift factor,  $\beta_t$ . The drift factor follows a first-order autoregressive process as provided in equation 13. The stochastic trend variable ( $v_t$ ) captures the technological progress and structural change.

The form that the trend takes depends on whether the variances,  $\sigma_\eta^2$  and  $\sigma_\xi^2$  (hyper parameters) are zero or not. If either  $\sigma_\eta^2$  or  $\sigma_\xi^2$  or both are non-zero, then the trend is said to be stochastic; STNS is the way to go. Otherwise, if both are zero, the trend is linear; the model reverts to a deterministic linear trend (DTNS),

$$(14) \quad Y_{it} = v_t + \varepsilon_{it}$$

where,  $v_t = v_{t-1} + \beta_t$ , with  $\beta_t$  being a fixed slope component, or, if the slope component is zero, then the expression reduces to,  $v_t = v_{t-1}$ .

If  $v_t$  is zero, there is no trend; the STS model reverts to a simple classical regression model without a trend term and the STS model may not be the way to go. Our third approach to estimating land use is time series analysis.

### **The Vector Autoregressive Model**

Vector Autoregressive (VAR) models are the multivariate estimation equivalent of Autoregressive Integrated Moving Average

(ARIMA) models in univariate estimation. Various criticisms of VAR models have been put forward, the major ones being that they are not based on economic theory (Gujarati, 2003; Kennedy, 1998).

But proponents of VAR approach argue that the models are useful for forecasting as they often outperform econometric models; they are also useful for describing various relationships in the data, and testing certain hypothesis and theories (Gujarati, 2003; Kennedy, 1998). Thus the VAR methodology has remained a line of choice for many economists particularly when the goal is forecasting as opposed to policy analysis. The basic VAR model can be represented as follows:

$$(15) \quad Y_t = \mu + \delta_1 Y_{t-1} + \dots + \delta_p Y_{t-p} + v_t$$

where,  $Y_t$  is a vector of endogenous variables;  $\delta$  is a vector of parameters to be estimated,  $p$  is the number of lags, and for all  $i$  and  $t$ ,  $v_t$  is a vector of uncorrelated error terms;  $v_t \sim (0, \Omega)$ , and  $\Omega$  is a diagonal matrix. The representation could also include a trend term.

Individually for two endogenous variables  $Y_1$  and  $Y_2$ , based on two lags ( $p=2$ ), the equations can be represented as:

$$(16) \quad \begin{aligned} Y_{1t} &= \mu_1 + \delta_{11} Y_{1t-1} + \delta_{12} Y_{1t-2} + \delta_{13} Y_{2t-1} + \delta_{14} Y_{2t-2} + v_{1t} \\ Y_{2t} &= \mu_2 + \delta_{21} Y_{1t-1} + \delta_{22} Y_{1t-2} + \delta_{23} Y_{2t-1} + \delta_{24} Y_{2t-2} + v_{2t} \end{aligned}$$

where the  $\delta$ s are parameter estimates;  $v_{1t} \sim (0, \sigma_1^2)$  and  $v_{2t} \sim (0, \sigma_2^2)$  for all  $t$ .

Where the goal is forecasting the endogenous variables, simply deriving the best forecasting fit for the data is appropriate.

## **Factors Influencing Land Allocation**

A number of studies on aggregate level land use change have documented factors thought to determine or influence land allocation. Time trend is one variable that would intuitively feature in land allocation models. Land allocation from one use to another changes over time, and as a variable time may capture the unknown causes of land use change, so structural time series analysis majors on trend as a measure of structural and technological change.

In addition to time trend, other factors have been thought to impact on land use including net returns, farm wealth/equity wage, interest rate, number of large animal(cattle and pig)units per acre, population density, per Capita income and other policy variables, including conservation and wetland reserve program and Government payments

It can be postulated that one major driver of conversion of land to urban use is population density. As population grows, people push into the forest and farmland simply to acquire room or quality settlement. Virtually all studies on land allocation use population density as an exogenous variable including Plantinga and Miller(1999), Ahn, Plantinga and Alig (2000), Wu and Sergerson(1995).

Ahn, Plantinga and Alig (2000), document a comprehensive model of land use at the aggregate level. Independent variables used in their model include revenues (real and discounted), population density, and measures of land quality. Using net revenues for competing land uses makes intuitive sense. Since we assume profit (or utility or net benefit) maximization, land will move from lower net revenue uses to

higher net revenue uses. In regard to land quality, we expect that fertile land is most likely going to be allocated to agriculture over forestry since will still do better than the former even in land of poor quality and higher marginal returns to fertile land are likely to be achieved with farming than forestry (Plantinga, Maulding and Miller, 1999).

The later study estimates a SURE model (assuming logistic distribution of share allocations) for forestry and farms (and normalizing on "other" land use). The independent variables applied include land quality, population density, net farm and forest revenues. Higher quality land tended to favor/increased allocation to farms.

Miller and Plantinga (1999) estimated least square (assumed a logistic distribution of share allocations), and maximum entropy models, for the allocation of cropland between corn and soybean, in three Iowa counties. They used government payments, fertilizer prices (production costs) and payment in kind (PIK) dummy variable, farmer prices, cost of inputs, and wages as independent variables.

Direct government payments to farmers, PIK, Conservation Reserve Program (CRP), and Wetland Reserve Program (WRP) incentives all contribute to increasing farm net revenue and the attractiveness of agriculture over other land uses.

We envisage that higher interest rates would push allocation of land to uses with higher returns as cost of capital and acceptable returns rise.

Wages add to the total revenue available to the land owner. Higher Off-farm/forest wages as compared to net farm/forest revenues could be an incentive to "sell the land and take a job". High wages may also imply higher cost of production and lower net returns that would push allocation to urban development as landowners sold out the land. The influence of wage is therefore indeterminate.

The number of cattle and other large animals like hogs and horses can reduce conversion of farmland to urban development particularly if large animals are a profitable enterprise as large animals normally require.

Land quality indices can come in handy in a panel data estimation setting or in models using geographical information systems (GIS) whereby land quality can vary across counties. In our kind of scenario however, land quality is assumed to be constant across the watershed and is therefore not an important variable.

Farm equity has been applied as an exogenous variable in modeling allocation among crop enterprises (Banerjee, 2004; Adhikari, 2004). Wealthier farmers may have higher investments in the farm and choose to keep it longer hoping for better days, hence positive effect on farmland; But higher wealth may imply higher expectations on returns forcing conversion away from farming if incomes consistently fail to meet expectations.

Per capita income is another factor that may be important in modeling land allocation. High per capita incomes may create an incentive for migration of population to counties with higher incomes, increasing pressure on land. High incomes may also increase pressure

on suburbs to encroach on agricultural and forest land as richer citizens demand higher quality of life away from the city core. But like extremely high wages, extremely high incomes may cause citizens to keep land unspoiled for aesthetic purposes. The impact of this on farm/forest allocation is indeterminate.

### **Econometric Model Estimation**

We estimated land allocation using the three approaches, namely econometrics, time series analysis and structural time series analysis. Our goal in estimation was basically forecasting land use and land use change. Without a priori knowledge of the type of model that would perform better in forecasting land use, between econometric, STS and VAR models, we opted to run the three types and let the data decide. We would then select the model with the best forecasting accuracy and utilize the results to forecast the water quality. Thus, in addition to the econometric and STSM, we estimated a VAR model to forecast forest and farm acreage in UCRB.

Since land use data is not available at the individual land owner level, we therefore aggregate county level data to arrive at ecosystem figures.

The theoretical model in equation 9 could be further simplified to:

$$(17) \ln(S_{it}/S_{0t}) = \mu_t + \beta_i' X_{it} + e_{it}$$

Where,  $\beta_i'$  is a vector of parameters to be estimated;  $X_{it}$  is a vector of independent variables for land use;  $\mu_t$  is the intercept,  $e_{it}$  is random white noise disturbance term; the model is identified if we set  $\beta_0 = 0$ .

Empirically the econometric model could be represented as:

$$(18) \quad A_{it} = \alpha_i + \theta T_i + \sum \beta_i X_{it} + Z_{it} + \varepsilon_{it}$$

where,  $A_{it}$  = log of share of (land)acreage allocated to use  $i$  over acreage allocated to use 0, at time  $t$ , excluding public land;  $T$  = time variable,  $\beta_i$  = vector of socio economic characteristics including ( $X_{it}$ ), that is, present discounted value of a stream of real revenues per acre (net returns) for  $i^{\text{th}}$  land use per acre at time  $t$ , ( $i=1$  for farm and  $i=2$  for forest); real farm wealth measured as average state level farm equity; real average wage per job; interest rate (20 year constant Treasury bill); Number of large animal (cattle and pig) units per acre; Population density; real per Capita income.

$Z_{it}$  = matrix of policy variables; including CRP/WRP (CWRP), PIK, Government payments per acre (GOV),

$\varepsilon_{it}$  = Gaussian white noise error terms

Wealth, Wage, Interest rate, government payments, and all income and return variables were deflated/normalized using Consumer Price Index for the south (CPI), (1982=100).

## **Data Sources**

Data (county level) covered the period between 1974 and 2005. Land use data are from the Natural Resources Spatial Analysis Laboratory (NARSAL). Timber yields are from Birdsey (1992), Birdsey (2003) and Plantinga (2007). Stumpage prices are from Timber Mart South. Forest rotation rates are from Griffin (2007). Timber revenues were compiled as weighted averages of the major types of timber occurring in the policy site.



Population data are from the US Census Bureau, government payments, net farm revenue and per capita income data are from the Georgia statistical system. Livestock numbers are from the various the Georgia County Guide. Farm equity and interest rate data are from the St Luis reserve bank. Wage and the Consumer Price Index (CPI) data are from the Bureau of Labor Statistics.

Forest management costs were minimal and were compiled from Dubois, Eric and Straka (1982) based on information provided by Griffin (2007). Final forest revenue was therefore present discounted value of streams of real timber revenues per acre.

Forest revenues were basically timber revenues discounted at a rate of 5%, as is the practice among studies applying forest returns as a variable (Plantinga, Maulding and Miller, 1999; Ahn, Plantinga and Alig, 2000).

## Results of the Econometric Model

We started out with the comprehensive outlay and estimated the model by equation-by-equation OLS (SURE). The all-inclusive model satisfies most OLS requirements except that it shows signs of multicollinearity in independent variables as evidenced by the very high  $R^2$  (at least 0.98), and the fact that other than the constant, few variables are significant at the 5% level. In addition there are "wrong" signs on three variables namely per capita income (INC), PIK, and wage (WAGE).

But this is not uncommon among land use models (Ahn, Plantinga, and Alig, 2000). As proposed by Gujarati (2003) we dropped groups of variables with pair-wise correlation coefficients exceeding 0.8, leaving about two per "group", and applied stepwise regression as determined by the value of partial  $R^2$ , to settle on the list of variables ultimately included in the model. We put off other measures of model suitability till we come up with a model that meets the requirements of OLS. The results of the forest acreage model are provided in table 1 and 2.

Both the forest and farm acreage equations meet the requirements of OLS. The models do not fail the normality, homoscedasticity and no autocorrelation tests. The coefficients of determination for both models are high (0.97 and above).

The F-values are significant implying parameter estimates do not equal zero and negative and significant signs for farm wealth and population.

The negative sign for population is as expected; as population density increases there is likely to be increased encroachment of urban development on forests and farmland. The negative sign on farm wealth may not have a clear-cut explanation. It may be that higher investment in the farm serves as an incentive to move land from forestry to agriculture, hence the negative sign.

The equation yields a positive and significant sign for forest returns implying, in line with expectations, that increased forest revenue is likely to be an incentive for land owners to increase forest acreage.

The results of the final farm acreage model are provided in tables 2. The equation yields significant (at 1% level), estimates for the intercept, population, and farm wealth. The equation yields negative but not significant (at 5%) estimates for farm government payments and wage, and positive and significant estimates for forest revenue.

As is the case with the forest acreage equation, increased population is likely to result in increased encroachment of urban development on farm land hence the negative sign on population. The negative sign on farm wealth implies increased farm wealth leads to increased conversion of farms to urban development. It may be that richer resource farmers have higher expectations of profits forcing conversion away from farming when returns fail to meet expectations.

The farm equation yields a positive and significant sign on forest returns, implying that increased returns from forestry are likely to be an incentive for land owners to increase farm acreage.

Whereas one would expect this to be the case in regard to forest acreage, assuming the two land uses are competing, the reasons for this positive effect on farm acreage are not clear. It may mean that the two are actually complimentary, as would be the case if, say, land converts to farming first then to forestry. But the effect may also be simply coincidental given the high (0.99) correlation between farm and forest acreage variables.

The literature examining cropland allocation across different crop enterprises is fairly common. But studies of land allocation at the level of forest versus farm and other uses, based on economic theory or even time series analysis, are scarce. In the overall, it is not uncommon for land use models to fail to provide substantial evidence as to factors that contribute to land use changes (Ahn, Plantinga and Alig, 2000).

The overall picture that we got from comparison of the results of our econometric model with those from other studies is that, aggregate land use models (forest-farms-other, etc) seem to provide results, consistent with expectation that cropland share models. One possible explanation for the differences between our results and past studies could very well be the kind of data available to us. In regard to land use, we had only six data points spread over 30 years, and we had to resort to interpolation to fill in the gaps. Estimating a more comprehensive model with county level data, in a panel data framework, would be the better way to go in the future. Even with six data points, not having to interpolate will most likely create a data set that would provide more agreeable results, *ceteris paribus*.

## **Structural Time Series Model Estimation**

We used the Structural Time Series Analyzer, Modeller, and Predictor (STAMP) version 6.0 program (Koopman, et. al., 2000) for STS analysis. The program carries out maximum likelihood estimation using numerical optimization procedure.

Model diagnostic tests are similar to those of the OLS (econometric) model. A few diagnostic tests are introduced;  $R_d^2$ , the Q statistic and the H statistic.

The STS analysis software, STAMP, uses  $R_d^2$  instead of  $R^2$  as the coefficient of determination whenever the model incorporates trend or seasonality components. The former is a better measure of goodness of fit where the series appears stationary with no trend or seasonality (Koopman et al, 2000; Harvey, 1989). The value of  $R_d^2$  may be negative indicating a worse fit than a simple random walk plus drift model.

The  $H(g)$  test is an  $F(g,g)$  non-parametric test of heteroskedasticity (Koopman, et al, 2000). A large F-value calls for rejection of the null hypothesis of homoscedasticity.

## **Results of the Structural Time Series Model**

For both the forest and farm equations, we estimated two versions of the simple STS model (without explanatory variables), that is, DTNS, and STNS. As with the econometric model, we assume farm and forest acreage models follow the same processes and estimate a seemingly unrelated structural time series equations (SUTSE), which is the same as equation-by-equation STSM. The results of the equations are presented in table 3 (DTNS) and table 4 (STNS).

In regard to the DTNS equation, the normality (N) values are below the 5% critical value of 5.99 so we fail to reject the null hypothesis of normality. Other tests of homoscedasticity, no autocorrelation are rejected at 5% and 1% levels.

The best STSM had a STNS structure and included interventions for change in the slope (structural breaks) of the dependant variables. The results of the final STSM are presented in table 4 below.

The diagnostic tests suggest the STNS model explains the data adequately. The DW statistics are around 1.5 which falls within the region of indecision but below the 5% (d) critical value of 1.553, so we fail to reject the hypothesis of no autocorrelation. The p-values for our Q statistic are 0.0283 for the forest equation and 0.0279 for the farm equation which suggests we fail to reject the null hypothesis of no serial correlation at 1%. Both the DW and the Q statistic support the no residual autocorrelation hypothesis generally, we conclude that this may not be a significant problem in the model.

The normality statistics are below 5.99 (and 9.22) the 5% and 10% critical values; we do not reject the null hypothesis of normality distribution of the model residuals. The heteroskedasticity H(g) test critical values with 8 degrees of freedom are 3.44 for 5% and 6.03 for 1% significance levels. The statistics exhibited by our models fall below these cut-offs, so fail to reject the null hypothesis of homoscedasticity.

The both coefficients of determination,  $R^2$  and the preferred  $R_d^2$  are high; at a minimum of 94% and 99% respectively meaning the model explains at least 94% of the variation in the dependant variables. For

both farm and forest equations, the forecast failure chi-square statistics are not significant at 5% so we do not reject the hypothesis of parameter constancy between the sample and pos-sample periods.

Both farm and forest equations yield substantially smaller RMSE and MAPE values, particularly when compared to the econometric model. This causes us to select the STSM as the preferred approach to forecasting land use in our policy site.

The RMSE statistic yielded by the STSM is 0.0322 for both farm and forest equation. This is about fifteen times smaller than the RMSE values from the econometric model which were between 0.553 (for farm equation) and 0.502 (for forest equation). The MAPE statistics for the two models are similarly divergent. The MAPE values of 23.0913, for the farm equation, and 6.4305, for the forest equation, are about 4 times smaller than corresponding values from the econometric model; 101.68 for the farm equation and 28.114 for the forest equation.

The STSM outperforms the econometric model in regard to forecasting accuracy. In the next chapter, we use the STSM model to forecast future land use, then model resulting, water quality changes and assign economic value to the later.

### **Vector Autoregressive Model Estimation**

The VAR model is similar to an econometric simultaneous equation model, except that all the variables are endogenous. The assumption is that lagged values of a variable should be able to explain the

variation in the variable itself. In estimating the VAR model, we started out with equation 3.15.

We used Akaike (AIC) and Schwarz (sic) information criteria to determine lag length with smaller values indicating a better fit for the data.

Given that our dependant variables are in logs, we expect to encounter the problem of heteroskedasticity. To counter this we used White (1980) estimation method and estimated heteroskedasticity consistent standard errors (HCSE).

To avoid contemporaneous correlation could be a problem in VAR model estimation. To circumvent this problem, we apply equation by equation OLS which is also the efficient estimation method in a SURE framework.

### **Results of the Vector Autoregressive Model**

Results of the VAR model are provided in table 5 and 6. We applied White's heteroskedasticity consistent standard errors to circumvent the problem of non-constant variance in the model - the Breusch Pagan heteroskedasticity test is therefore omitted from the results.

The VAR model meets CLRM requirements. For both farm and forest equations, the DW test yields a d-value of 1.2, which falls within the indecision region for the 5% level of significance (1.038 to 1.767) for  $N=25$  and  $k=4$ . This implies we can not conclusively reject the null hypothesis of no autocorrelation based on this statistic. Nevertheless, for both equations, the LM autocorrelation tests



statistics is not significant at 5% level. As indicated earlier this is a more appropriate test when lagged dependant variables are present in the model as is the case here, so we may not reject the null hypothesis and conclude that autocorrelation may not be a problem in the model.

The Jarque-Bera normality tests statistic  $N$  is not significant at 5% level. These results suggest we can not reject either the normality hypotheses. The chow parameter consistency/prediction failure test statistic is not significant at 5%, suggesting we can not reject the null hypothesis of no structural change in parameter values.

The model is also a reasonably good fit for the data with  $R^2$  values of 0.99 and above. The overall equation F-statistics are significant at 1% level, which suggests we may reject the null hypothesis that the coefficients are collectively equal to zero.

### **Forecasting Ability: Comparing The Models**

Table 7 compares the three land use models, econometric, VAR and STSM in regard to forecasting ability. Overall, the VAR model RMSE and MAPE values were smaller than the corresponding values yielded by the econometric model but smaller than the STSM. Thus, in terms of model forecasting accuracy, the STSM would be most preferred followed by the VAR time series model, with the econometric model being the worst performer. Although this is as expected, many econometricians are of the opinion that wherever time series models outperform econometric models, the econometric model may be miss-specified (Green, 2000;

Kennedy, 2001). It may be important for future research to examine the validity of this notion in regard to land use modeling.

Whereas both econometric and the VAR models yielded negative mean error values, the STSM yielded a positive value. This suggests that the first two models would likely overestimate land acreages allocated to farms and forests, where the STSM would likely underestimate these allocations. We selected the STSM for forecasting on the basis of RMSE and MAPE statistics.

### **Forecasting Land Use and Land Use Change**

A key objective of our study was to forecast land use in North Georgia, in order to forecast changes in water quality and economic value of this environmental good. We forecasted land acreage for farm and forest uses for the years 2006 to 2030 under three scenarios, that is, Scenario I, the highest rate of conversion (to urban land use) as forecasted by the STSM; Scenario III, limited or managed conversion represented by average growth rate between 1974 and 2005; Scenario II, based on actual land use data; moderate conversion represented by the average growth rate between the two scenarios above. We would not expect conversion rates to fall below scenario III levels, as urbanization and deforestation have been rising steadily over the years.

### ***Land Use Change Forecasting***

Table 8 provides land use shares under different scenarios from baseline through scenario III. The tables depict the extent to which land allocation changes (for each category of use) between year 2005

(baseline) and 2030 under different scenarios. Summarily, Scenario I (STSM) represents highest conversion. Under this scenario, urban growth (commercial and residential areas) encroaches on farms and forests to increase from 14% to 68% as the later two reduce from 66% to 24% and 20% to 8% respectively. In scenario II, moderate growth, urban growth takes over from farms and forests to increase by a lower but significant magnitude to 50%. Land in farms drops by to 37% while forestry drops to 13%. Under scenario III with mitigating action/managed growth, urban growth increases to 21%, farm acreage increases marginally to 22%, and forestry drops to 56%.

#### **From Land Use Change to Ecosystem valuation**

The L-THIA model was developed by the Purdue Research Foundation as a tool for mapping out changes in run off, recharge and nonpoint source pollution (NPSP) resulting from land use changes (Purdue Research Foundation, 2004; Engel, et.al., 2003; Bhaduri, et. al.,1999). The model computes long term average annual estimates of the aforesaid hydrological parameters for specified land use scenarios, based on long term historical climatic data at county level.

The software requires selecting the hydrological soil group or groups and an input of the type and size of land use change. The software then computes expected runoff depths and volumes and nonpoint source pollution loadings to water bodies. For the purpose of this study and given our need for previous studies with benefit transfer

data, we zeroed down on major stressors including Nitrates, Phosphorous, Dissolved oxygen (DO), and Fecal Coliform (bacteria).

Nitrates, as nitrogen or otherwise, are about the most discussed contaminants of drinking water in the literature. Together with phosphorous, nitrates and nitrites are associated with agriculture (fertilizers and animal waste) and human residential waste disposal.

Pollution by nitrates is especially a problem with ground water as 22 per cent of domestic wells in agricultural areas, in the US, report nitrogen contamination (GAEPD, 1997). In humans, excess nitrogen (more than 10 mg/L) is associated with blue baby syndrome and nitrogen can also be transformed into carcinogenic compounds (Ward, et. al.).

Digestive problems can occur in animals and humans ingesting high levels of phosphorous (phosphates, etc) but toxic effects of phosphorous in humans are not common.

Excessive levels of phosphorous and/or nitrates in water bodies stimulates accelerated growth of planktons and other aquatic plants (eutrophication) resulting in choking of the waterways, diminished oxygen (hypoxia) and death of aquatic life (USEPA, 1986a).

The EPA has minimum in-stream water standard for the DO level of drinking, recreation, and fishing waters set at 4.0 mg/L. The standard for trout fishing waters is higher at 5.0 mg/L.

The amount of oxygen in water (dissolved oxygen) is important for the survival of aquatic life. Levels of in the water are dependant on temperature and the level of nutrients and solids in the water (GAEPD, 1997). Dissolved oxygen criteria are therefore meant to be lower

limits below which aquatic life is impaired. A low level of DO indicates high levels of nutrients and solids without specificity as to type. But DO criteria are not covered by L-THIA making it hard for us to estimate the levels and changes in dissolved oxygen.

An alternative indicator of DO is Biochemical Oxygen Demand (BOD). This measures the amount of oxygen that bacteria will require to decompose organic matter. If runoff or effluence entering a river is rich in organic matter, there will be intensive bacterial decomposition organic matter; BOD will be high resulting in competition, for oxygen, with aquatic life. This will decrease the amount of DO, at, and downstream of the point of discharge to the extent that in-stream life could die (CRC, 2000). High levels of BOD are accompanied by low levels of DO. Accordingly, BOD is a good indicator of the health of a stream, river or other water body. Recommendations for BOD are scarce, but the Australian government recommendation for BOD for protecting freshwater aquatic life is a maximum of 15 mg/L (CRC, 2000). We adopted this criterion for the purpose of this study.

Contamination of drinking water by bacteria, particularly the Fecal Coliforms group (including the infamous *Escherichia coli*), is a major water quality concern. Bacteria are mainly associated with human and animal waste that finds its way into ground or surface water. In low levels, fecal coliform bacteria (FCB) may cause no harm, but high levels they are considered an indicator of potential health risk to humans.

The GAEPD has water quality standards consisting of two groups of criteria; the general criteria that apply to all waters and the specific criteria that vary with the intended use of the water. Table 10 shows various (USEPA, GAEPD, other states) water quality criteria for some of the stressors, for drinking water (USEPA,2000; GAEPD,2004). For ease of presentation, only stressors covered by the L-THIA software are included in the table.

Although there are not many primary (enforceable) numeric criteria for pollutants like total nitrogen, in regard to rivers and streams with fishing as designated use, NPS pollutants do affect aquatic life in general and fish in particular. For instance, levels of TN in excess of 0.5 mg/L are toxic to rainbow trout (North East Georgia Regional development Center (NEGRDC), 2004). Available water quality criteria for fishing and recreation are presented in table 11.

For the purpose of this study the baseline year for land use change will be the latest year for which land use data exists, which is 2005. We plugged in the Control numbers to come up with runoff, and level of Non Point Source (NPS) pollutants in the waters within the ecosystem.

Table 12 provides the L-THIA program output of average annual water quality parameters for the study area under the different scenarios - the table covers only major NPS pollutants.

From the table it seems that although the level of TSS, BOD and FCB increase across all scenarios, only fecal coliform and BOD criteria are likely to be violated in the study area. The BOD criterion of 15 mg/L is exceeded in scenario I and II. The FCB

criterion of 200 colonies/100 ml is exceeded under all scenarios including current (2005) baseline land distribution. Current bacteria violations may be as a result of poor human waste disposal systems but more likely livestock waste is the culprit as chicken, hog and cattle farming are the main agricultural enterprises in the study area.

Future violations of biotic criterion may be related to increased urban development and accompanying problems with human waste disposal such as untreated/poorly treated waste and seepage from malfunctioning septic systems. The violations may also be related to loss of forest cover and increase in impervious (urban) surfaces, both of which may result in excess runoff and deposition of solids (TSS increase) and microbes in the water bodies.

Although farm acreage increases in scenario III, this happens at the expense of forests which reduce by 14%. The resulting reduction in land cover and animal waste may be responsible for increased levels of FCB and TSS. In the next section we estimate the value of water quality changes discussed above.

#### **Benefit Transfer: Empirical Application**

As discussed earlier, we apply Benefit Transfer method to the valuation of water quality as an ecosystem service. The lands of Habersham and White counties in the UCRB are the affected area. Land reallocation causes changes to water quality which is manifested by changes in biological oxygen demand (proxy for dissolved oxygen) and fecal coliform bacteria levels.

We start by assuming the individual has a right to the initial situation (higher drinking water quality). Given the difficulties of measuring WTA, we follow Freeman (1993) and measure WTA indirectly through WTP. We now assume the individual has a right to the subsequent lower quality and proceed to measure the welfare change



representing an income decrement corresponding to the individuals willingness to pay to prevent a water quality decrease.

Numerous studies on water quality valuation have been documented. Nevertheless most studies cover nitrate (nitrogen) contamination and studies on water quality as measured by BOD and FCB are not plentiful.

### ***Transferring Benefits, Valuing the Ecosystem***

In the USA, documented past studies on FCB contamination are few and those that exist offer limited use for BT. This is so because the in most relevant studies Fecal coliform is but one of the problems addressed so that it becomes impossible to extract values that would apply solely to the FCB problem.

Collins and Steinback (1993) apply the cost of averting behavior to study rural household willingness to pay for reduced water contamination by FCB, organic chemicals and minerals, in West Virginia. Estimates of WTP in this study are considered lower bounds as actual WTP is likely to be higher than defensive expenditures (Bartik, 1988) used to estimate WTP for this study. The study estimates WTP to eliminate FCB problem in drinking water to be USD 320 per household per year.

Table 13 compares the study and policy sites. Surface water is the predominant source of drinking water north of the Georgia fall line in the Piedmont province of the Chattahoochee River Basin (GAEPD, 1997). We can therefore make the assumption that 100% of the public in the policy site use surface water and have interest in local surface water quality. In addition, most agricultural water, used in the UCRB

(mainly for livestock and aquaculture) is surface water. Additionally, rivers and streams North of Lake Lanier are host to recreational cold-water trout fisheries (GAEPD, 1997). To transfer this value to the UCRB study site, we adjust for income and time as outlined earlier.

The WTP for programs that would clear the waterways of FCB is estimated at USD 631.70 per household or USD 248.00 per capita per year in constant 2005 dollars. This amounts to USD 15,785,740.00 per year for the entire population of the policy site. The 2005 constant prices WTP for the West Virginia study site was about USD 196.30 per capita per year. The two values compare reasonably considering the differences in per capita income between the two areas. We note that these are lower bound WTP values since they are derived from defensive expenditures.

In regard to BOD and DO water quality benefit valuation, Russell and Vaughan (1982) applied the Travel Cost Model of the number of one day fishing trips made by anglers in Indiana and neighboring fish and wildlife recreation regions in, to estimate WTP for water quality due to BOD/DO.

Their estimation yields annual economic values of between USD 2.05 USD 4.56 per capita. These values represent WTP for increasing BOD/DO to national standards through Best Available Technology (BAT). Table 14 compares the study and policy sites for BOD violation.

Transferring these values to the policy site with appropriate adjustments for income and time yields annual WTP of between USD 5.58 and USD 12.42 per capita, which translates to an aggregate WTP of USD

355490.10 and USD 790748.70, for the entire policy site. A summary of the benefits is provided in Table 15.

The water quality forecast results (Table 12) seem to suggest that we are likely to have existing FCB violations in the UCRB. We have therefore included a valuation for FCB control benefits in Table 15 for the baseline year.

Our results suggest that the lower bound WTP for creating and maintaining water quality standards for drinking water supply and fishing are about USD 15,785,740 under baseline and scenario III (managed growth) conditions and about USD 16,141,230 under scenarios I and II.

### **Summary Findings**

In this study, we sought to model land use change in the North Georgia and to provide economic valuation of subsequent changes in watershed ecosystem services and functions. Towards this end, we developed three models of land use change, a Structural Time Series Model, an Econometric Model and a Vector Autoregressive Model. We selected the Structural Time Series Model for forecasting land use, based on conventional criteria, namely, Mean Absolute Percentage Error and Root Mean Square Error. We then proceeded to forecast three likely land use and land use change scenarios based on the aforesaid results and the resulting changes in ecosystem services, basically water quality for drinking and fishing, for the year 2030. We applied Benefit Transfer Techniques to estimate the economic value of water quality in the North Georgia.

All future scenarios, except limited growth, showed excesses (worsening water quality) for BOD and FCB and worsening (increasing) runoff. In addition the baseline also showed violations for FCB.

A key result of the model is that both (farm and forest share) equations yielded negative and significant signs for population, showing as expected that increase population density will all things equal result in increased encroachment of urban development on forests and farms.

### **Conclusions and Recommendations**

The most important contribution of the econometric land allocation model was the negative and significant sign on population. This implies that, *ceteris paribus*, increased population density results in increased encroachment of urban development on forests and farms. The STSM outperformed both the VAR and the econometric model in terms of forecasting ability. The presence of a stochastic trend in the model suggests that models of land use that ignore the trend variable might be miss-specified and might lead to erroneous conclusions. All land use forecasts pointed toward loss of forest land to urbanization. Farmland may or may not be spared the encroachment; it would all depend on interventions that the community takes to control urban growth.

Water quality modeling revealed that land use change would result in increased runoff, and associated increase in FCB and BOD/DO violations. But the BOD/DO violations could be curtailed by managing urban growth as evidenced absence of BOD violations in the managed

growth scenario. Our study finds there may be problems of FCB under all postulated future land use scenarios. The findings also support existing literature that there are problems with FCB violation in the study area at the moment.

Finally, it seems that the people of UCRB would be willing to pay a lower bound value between USD 15,785,740 and USD 16,141,230 per year to create and maintain quality standards for fishing and drinking water supply.

### **Implications and Future Research Recommendations**

Thus far, few economic and statistical based ecosystem and aggregate land use models exist. The few that have been estimated are based on conventional econometrics and there have been no significant attempts to apply structural and/or time series methods in estimating land use. Additionally literature forecasting land allocation is noticeably scarce and scarcer still is literature exploring land use change implications for water quality particularly in the setting of an ecosystem.

A key contribution of this study is to estimate land use model using VAR and STS models as past studies have relied solely on traditional econometric models. STSM are also better placed for ex-ante forecasting as there is a reduction in the number of variables to be forecasted.

This study was particularly constrained by scarcity of land use data. The final data set consisted of six observations spread over the period between 1974 and 2005 compelling us to interpolate between the

observation to obtain sufficient data and degrees of freedom. Since land use data were available at county level, future research could surmount this problem by covering using a panel data approach; covering a larger portion of the watershed, hence having more data points from more counties.

Our study supports the literatures in finding problems of FCB in the North Georgia ecosystem. These and the problems of BOD/DO can be ameliorated by concerted efforts including introducing best management practices, reducing impervious surfaces, reducing urban sprawl so as to conserve the forest, and other activities that involve the community in watershed management. Such approaches are likely to cost less than the cost of defensive behavior or ecosystem restoration after the fact.

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**Table 1: Forest Acreage Selected Model Estimates**

Parameter	Estimate	SE	t-value	p-value
Intercept	4.1770**	0.3657	11.4000	0.0000
POP	-0.0249**	0.0021	-12.1000	0.0000
WAGE	-0.0002	0.0031	-0.0796	0.9370
$\Pi_1$	4.8519**	0.4904	9.8900	0.0000
GOV	-0.0108	0.0070	-1.5500	0.1370
EQ	-0.0675**	0.0193	-3.5000	0.0020
Mean(LFORESr)	2.9104			
Variance(LFORESr)	0.2457			
RSS	0.1132			
$R^2$	0.9823			
F(5,20)	221.664[0.0000]**			
LM, F(2,18)	3.1679[0.0663]			
N, $\chi^2(2)$	0.5843[0.7467]			
Chow, F(6,20)	4.0977 [0.0077]**			
BP, F(10,9)	0.5849 [0.7924]			

Note: \*\* - implies significant at 1%; \* - significant at 5%



**Table 2: Farm Acreage Selected Model Estimates**

Parameter	Estimate	SE	t-value	p-value
Intercept	2.8282**	0.4071	6.9500	0.0000
POP	-0.0189**	0.0023	-8.2600	0.0000
WAGE	-0.0028	0.0034	-0.8370	0.4120
$\Pi_1$	4.7909**	0.5459	8.7800	0.0000
GOV	-0.0049	0.0078	-0.6380	0.5310
EQ	-0.0802**	0.0215	-3.7400	0.0010
Mean (LFARMr)	1.5941			
Variance (LFARMr)	0.1796			
RSS	0.1403			
$R^2$	0.97			
F(5,20)	129.19	[0.0000]**		
Chow, F(6,20)	3.5229	[0.0152]*		
LM, F(2,18)	3.4671	[0.0532]		
N, $\chi^2(2)$	0.6539	[0.7211]		
BP, F(10,9)	0.8048	[0.6321]		

Note: \*\* - implies significant at 1%; \* - significant at 5%

**Table 3: Diagnostic Summary of DTNS**

Statistic	forest	farm
Std. Error	0.0786	0.0827
Normality	1.1054	1.0214
H(8)	136.1200	140.8500
DW	0.3918	0.4061
Q(7,6)	35.4390	34.9900
Rd <sup>2</sup>	-0.5590	-0.4912

**Table 4: Diagnostic summary of the STS model**

Statistic	forest	farm
Std. Error	0.0149	0.0149
Normality	3.873	3.7856
H(8)	1.1001	1.1936
DW	1.5052	1.5151
Q(8,6)	14.119	14.164
Rd <sup>2</sup>	0.9440	0.9514
R <sup>2</sup>	0.9991	0.9988
Forecast Chi2(6)	2.0926[0.9110]	1.8313[0.9345]

**Table 5: Farm Acreage Selected VAR Model Estimation**

Parameter	Estimate	Std Error	t-value	p-value
LFOREST_1	12.2257	2.7130	4.5100**	0.0000
LFARM_1	-10.4162	2.4900	-4.1800**	0.0000
Trend	0.1725	0.0419	4.1100**	0.0000
Constant	-20.0108	4.5510	-4.4000**	0.0000
Mean(LFARM)		1.57682		
Variance(LFARM)		0.179105		
RSS		0.0608		
R^2		0.99		
F(3,21)		508.1[0.000]**		
DW		1.2		
Chow, F(6,21)		2.0852[0.0986]		
LM, F(2,19)		1.6139[0.2252]		
N, Chi^2(2)		5.4651[0.0651]		
AIC		-2.8604		
SIC		-2.6654		

Note: X<sub>i</sub> implies ith lag of variable X; \*\* - implies significant at

1%; \* - implies significant at 5%

**Table 6: Forest Acreage Selected VAR Model Estimation**

Parameter	Estimate	Std Error	t-value	p-value
LFOREST_1	12.2121	2.5090	4.8700**	0.0000
LFARM_1	-10.4746	2.3040	-4.5500**	0.0000
Trend	0.1580	0.0387	4.0800**	0.0010
Constant	-18.3559	4.2090	-4.3600**	0.0000
Mean(LFARM)		2.8939		
Variance(LFARM)		0.2415		
RSS		0.0534		
R^2		0.9911		
F(3,21)		784[0.000]**		
DW		1.2		
Chow, F(6,21)		1.9559[0.1184]		
LM, F(2,19)		1.6414[0.2200]		
N, Chi^2(2)		5.3485[0.0690]		
AIC		-2.9903		
SIC		-2.7953		

Note: X\_i implies ith lag of variable X; \*\* - implies significant at

1%; \* - implies significant at 5%

**Table 7: Forecasting Ability of the Land Use Models**

Model	Farm	<b>RMSE</b>		Farm	<b>MAPE</b>	
		Forest	Forest		Forest	Forest
Econometric		0.55	0.50		101.68	28.11
VAR		0.37	0.33		67.63	17.81
STSM <sup>*</sup>		0.03	0.03		23.09	6.43

<sup>\*</sup>: The STSM has the smallest RMSE and MAPE

**Table 8: Land Use Shares Under Different Scenarios**

<b>Scenario</b>	<b>Forest</b>	<b>Farm</b>	<b>Urban</b>
Baseline 2005	66	20	14
I. STSM highest growth	24	8	68
II. Moderate growth	37	13	50
III. Managed growth	56	22	21

Note: Values are percentages

**Table 10 Water Quality Criteria for Rivers and Streams and drinking water quality standards**

Intended Use	TN	TP	Total Suspended Solids	Fecal Coliform	Dissolved Oxygen (minimum)
Drinking water supply (not treated)	0.69 mg/L	0.57 mg/L	50 mg/L	200 colonies/L	>4.0 mg/L
Portable drinking water	0.10 mg/L	NA	NA	<=5% of samples per month	NA

Figures represent Maximum Contaminant Level (MCL). Figures are from USEPA (2000) and GAEPD (2004). TN and TP stand for Total Nitrogen and Total Phosphorous respectively; figures are based on the 25<sup>th</sup> percentile. Fecal coliform figures are based on 30 day geometric mean. BOD criterion is from the literature (CRC, 2002) and can be assumed to be secondary.



**Table 11 Water Quality Criteria for Rivers and Streams for non-human uses**

Intended Use	Fecal Coliform (30 day geometric mean)	BOD (maximum)	Dissolved Oxygen (minimum)
Fishing(all species)	200 colonies/L	> 15 mg/L	>4.0 mg/L
Fishing(trout)	200 colonies/L	> 15 mg/L	>5.0 mg/L
Recreation	200 colonies/L	NA	>4.0 mg/L

Source: CRC, (2002) for BOD; GAEPD (2004) for all other data.

**Table 12: Runoff and NPS Pollutant Loadings in 2030**

<b>Pollutant</b>	<b>Baseline</b>	<b>Scenario I</b>	<b>Scenario II</b>	<b>Scenario III</b>
Runoff depth (in)	69555.99	111088.40	97399.67	76246.05
Nitrogen (mg/L)	0.99	1.57	1.43	1.10
Phosphorous (mg/L)	0.15	0.44	0.37	0.20
BOD (mg/L)	7.25	21.19	17.75	9.90
Fecal Coliform (col/100ml)	483.04	1439.50	1203.11	664.57
TSS (mg/L)	12.91	37.47	31.40	17.57

**Table 13: FCB Comparison Between Study and Policy Sites**

	<b>Study Site</b>	<b>Policy Site</b>
Place	West Virginia	UCRB, Georgia
Authors	Collins and Steinback(1993)	Ngugi, D. G.
Problem	FCB in drinking water	FCB in drinking water
Per Capita Income(2005)	\$27215.00	\$24726.87
Water Source	98%	100%
Data Source	Survey, mail and personal	Benefit transfer
Rural/urban	Rural	Rural
WTP/Capita/Year	\$196.30	\$248.00

**Table 14: BOD/DO, Comparison Of Study Site and Policy Site**

	<b>Study Site</b>	<b>Policy Site</b>
Place	National, 48 states	UCRB, Georgia
Authors	Russel & Vaughan (1982)	Ngugi, D.G
Data Source	National Survey of Fishing (USFWS)	
Problem	Excess BOD/Low DO in fishing water	Excess BOD/Low DO in fishing water
Per Capita Income (2005)	\$34,586	\$24726.87
Rural/urban	Both	Rural
WTP/Capita/Year	\$2.05-\$4.56	\$5.58-\$12.42

**Table 15: Economic Value of Water Quality in the UCRB**

Pollutant	Baseline 2005	Scenario I	Scenario II	Scenario III
BOD/DO	NA	355.49	355.49	NA
Fecal Coliform	15785.74	15785.74	15785.74	15785.74
Total	15785.74	16141.23	16141.23	15785.74

Note: Values are in thousands of US dollars per year at constant 2005 prices.