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, and estimate the economic value of the ecosystem using benefit transfer techniques. We forecast land use change based on a structural time series model and a simple growth rate model. The study suggests a lower bound willingness to pay value of about USD 16,000 per year to ensure compliance with fishing and drinking water quality standards with regard to fecal coliform bacteria and dissolved oxygen. Conservation efforts are likely to cost less than the cost of defensive behavior or ecosystem restoration.

Keywords: Ecosystem, Economic value, North Georgia, land use, water quality, structural time series, benefit transfer, forecasting.

Introduction

Ecosystems play an important role in providing commodities (functions and services), beneficial to society. To a great extent, the quality and quantity of these benefits is determined by the pattern of land use in the ecosystem. The economic value of an ecosystem reflects tradeoffs made to obtain more or prevent loss of such 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 depends on what land use takes what proportion (and portion) of the land. Normally though, as urbanization progresses, land moves from farms and forests to urban and industrial areas, increasing runoff, biological and chemical pollutants, and decreasing the level and quality of environmental benefits.

Generally, land use models estimate the relationship between the share of land allocated to alternative uses and factors that determine such allocation. Typically, analysis is based on either county-level data (aggregate models) or data on individual land units, say in at farm level (disaggregate models). The models are developed using either econometric or time series, or more recently, structural time series methods. The resulting model is then used to predict future land use by plugging in the forecasted determinants.

A structural time series model (STSM) has benefits over econometric and conventional time series models particularly when forecasting is required and time and budget are constraints. The basic form of the model is conservative in time and cost as the only data required is the dependent variable. Moreover, the STSM lends itself to simpler application in forecasting because the compounded error due to using pre-forecasted independent variables can be avoided. Finally, it has been demonstrated that the model could have better forecasting accuracy than econometric or conventional time series models (Ngugi, 2007).

Economists have in the last decade begun to place emphasis on valuing entire ecosystems as opposed to individual ecosystem benefits (Lambert). 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. Nonetheless, few studies have been documented that examine land use and the level of ecosystem benefits jointly, and, even fewer incorporate ecosystem valuation. Most studies major either on land use or some aspect of ecosystem valuation.

In the past, economists have used diverse approaches to valuing ecosystems. Most studies apply a combination of methods, including market prices (for marketed components) and direct and indirect techniques (for non-marketed components). Except for market prices, other methods of valuing ecosystem components or entire ecosystems are costly both in time and money.

In the recent past, "Benefit/Benefits transfer" (BT), is becoming increasingly useful as an approach to valuation of non-marketed public goods and service. Benefit 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). 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". Benefit Transfer has been used to value ecosystems in numerous studies, including Constanza et al (1997), Verna (2000), Toras (2000) and Kramer et al (1997).

The objectives of this paper are:

1. To forecast land use change and associated change in water quality for Habersham and White counties in the Upper Chattahoochee River Basin (UCRB) in North Georgia.
2. To estimate the economic value of the ecosystem with emphasis on water quality using benefit transfer techniques.

The Study Area

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. Observing land use data available to us suggests that between 1974 and 2005, the proportion of available land (excluding government forest, water, wetlands) in the area of study (policy site) that was under residential/urban use increased from 2.4% to 14%. This urban growth was mainly at the expense of forest that decreased from 79.4% in 1974 to 65.6% by 2005. Farmland increased marginally from 18.2% to 20.4% during the same period. 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 (Georgia Department of Natural Resources (GDNR), 1997). Models of future land use and land use change could provide information on how the aforesaid changes would affect the level of benefits from and value of this watershed with implications for similar ecosystems.

The remainder of the paper proceeds as follows: First, we review literature structural time series techniques and land use modeling. We also examine literature on ecosystem valuation emphasizing on benefit transfer. Then, we discuss the data and methodology, followed by empirical forecasting of land use and transferring of economic benefits. This is followed by a discussion of the results and our conclusion of the study and recommendations.

Literature Review

Land Use Modeling

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. These 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

Ecosystem valuation is typically approached from the perspective of total economic value (TEV). This can be viewed as the value of environmental resources, goods or benefits generated as determined by peoples' preferences. It is the sum of "Active use", and "passive-use" values. Active use values (AUV) are derived from use of a natural resource including option, direct and indirect uses. Passive-use values (PUV) on the other hand include existence, altruistic and bequest values.

Most environmental benefits 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. Changes affecting (the level of) such benefits result in changes in the consumer's bundle hence changes in consumer welfare.

Marshallian measures of consumer welfare are useful when constant marginal utility of income assumption holds. This assumption only holds in rare circumstances (homothetic preferences and very small changes in utility). Moreover Marshallian Surplus (MS) is not easy to measure. Hicksian (exact) welfare measures are often preferred because they hold utility constant so that the money metric will always accurately reflect ordinal utility changes. Equivalent Surplus (ES) and Compensating Surplus (CS) are two exact welfare measures for imposed or rationed quantity changes (Freeman, 1993). One could use either Willingness to pay (WTP) or Willingness to Accept Compensation (WTA) to measure welfare changes. The choice of welfare measure depends on the assumption we make about property rights of the individual. Eliciting realistic responses on WTA has been proven to be a difficult task so that often WTP approaches are used to measure WTA indirectly (Freeman, 1993).

Most ecosystem values, both active and passive use, render themselves to measurement by ES or CS. Most direct use values can be measured using market prices since they are marketable. For commodities whose preference is readily revealed through expenditure eg, camping, hunting, measurement is often done using Marshallian Surplus. With regard to producers, changes in welfare can be measured using producer surplus.

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).

Traditional techniques of measuring the value of improvements in ecosystem or resource quality have one two major limitations; they are costly both in time and money. 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).

Although 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. Following Verderberg, Poe and Powell(2001) Willingness to Pay

(WTP) for improvement of an ecosystem commodity (such as water quality) for the ith individual at the jth site can be specified as:

(1) $WTP^{*}_{ij} = \omega_j(Q^0_{ij}, Q^1_{ij}, I_{ij,} H_{ij})$

Where ω_j represents the average valuation function for the jth 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. 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.

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.

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 form contingent valuation interviews suggested a willingness to pay for additional ecosystem services ranging from $ 25.00 per month to $252 per year. Other notable applications benefit transfer to ecosystem valuation include Verma (2000); Toras (2000) ; Bouma and Schuijt (2001).

Data and Methodology

County level land use (farm, forest and urban) data covered the period between 1974 and 2005. Data are from the Natural Resources Spatial Analysis Laboratory (NARSAL).

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:

(2) $Y_{it} = v_t + \delta_{it}^{'} X_{it} + \varepsilon_{it}$

Where α_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, v_t is the trend component, and ε_{it} is the white noise disturbance term.

The simple STSM without explanatory variables may be represented by,

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

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

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

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

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

Equations (4) and (5) represent the level and the slope of the trend, respectively; v_{t-1} is a random walk with a drift factor, β_t. The drift factor follows a first-order autoregressive process as provided in equation 5. The stochastic trend variable (v_t) captures the technological progress and structural change.

The form that the trend takes depends on whether the variances, σ_η^2 and σ_ξ^2 (hyper parameters) are zero or not. If either σ_η^2 or σ_ξ^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),

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

where, $v_t = v_{t-1} + \beta_t$, with β_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.

Empirical Estimation: Forecasting Land Use and Water Quality 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.

Once land use scenarios are forecasted inputting this data into a land use-water quality model is necessary to forecast the effect of land use on water quality. The Long Term Hydrologic Impact Assessment Model (LTHIA) 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.

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 OLS model. A few diagnostic tests are introduced: Rd^2, the Q statistic and the H statistic. The STS analysis software, STAMP, uses Rd^2 instead of R^2 as the coefficient of determination whenever the model incorporates trend of 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 Rd^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 and Discussion

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. We assume farm and forest acreage models follow the same processes and estimate a seemingly unrelated structural time series equations (SUTSE). The results of the equations are presented in Table 1 (DTNS) and Table 2 (STNS).

Table 1: 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^2	-0.5590	-0.4912

Table 2: 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^2	0.9440	0.9514
R^2	0.9991	0.9988
Forecast Chi2(6)	2.0926[0.9110]	1.8313[0.9345]

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 most suitable 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 2.

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 Rd^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.

Land Use Change Forecasting

Table 3 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%.

Table 3: Land Use Shares Under Different Scenarios

Scenario	Forest	Farm	Urban
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

From Land Use Change to Ecosystem Value

We applied the LTHIA to forecast water quality given the three land use scenarios. 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, Dissolved oxygen (DO), and Fecal Coliform (bacteria). Nitrates, 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.).

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 LTHIA making it hard for us to estimate the levels and changes in dissolved oxygen.

An alternative indicator of DO is Biochemical Oxygen Demand (BDO). 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.

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. Table 4 provides the LTHIA program output of average annual water quality parameters for the study area under the different scenarios for major NPS pollutants and water quality criteria.

Table 4: Runoff and NPS Pollutant Loadings in 2030

Pollutant	Criteria	Baseline	Scenario I	Scenario II	Scenario III
Runoff depth (in)	N/A	69555.99	111088.40	97399.67	76246.05
Nitrogen (mg/L)	10	0.99	1.57	1.43	1.10
BOD (mg/L)	15	7.25	21.19	17.75	9.90
Fecal Coliform (col/100ml)	200	483.04	1439.50	1203.11	664.57
TSS (mg/L)	50	12.91	37.47	31.40	17.57

Figures represent Maximum Contaminant Level (MCL).

Source: Drinking water criteria are from USEPA (2000) and GAEPD (2004) and are based on the 25^{th} percentile. TN stands for Total Nitrogen. Fecal coliform numbers are based on 30 day geometric mean. BOD criterion is from CRC (2002) and are assumed to be secondary. Fishing Water Criteria are from CRC (2002) for BOD and GAEPD (2004) for all other data.

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 is exceeded in scenario I and II. The FCB criterion 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: Application

We applied Benefit Transfer method to the valuation of water quality as an ecosystem service. 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. 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 difficult 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 5 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).

Table 5: FCB Comparison Between Study and Policy Sites

	Study Site	**Policy Site**
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

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/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 6 compares the study and policy sites for BOD violation.

Table 6: BOD/DO, Comparison Of Study Site and Policy Site

	Study Site	Policy Site
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

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 7. The water quality forecast results (Table 4) 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 7 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.

Table 7: 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.

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 forecast three likely land use scenarios, and the resulting changes in ecosystem services; 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. As past studies (Ngugi, 2007) suggest, increase in population density will all things equal result in increased encroachment of urban development on forests and farms.

Conclusions and Recommendations

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 scenarios pointed toward loss of forest land to urbanization. Loss of farmland is likely to 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.

There are not many ecosystem based aggregate land use models in the literature. Even fewer attempt to apply structural time series methods in estimating land use. 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 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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