A SIMPLE BIOECONOMIC MODELING OF LAND USE CHANGE AND ITS IMPACT ON WATER QUALITY AND AGRICULTURAL RETURNS[1]

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ABSTRACT

The objective of this study is to assess the economic and water quality impact of land use change in a small watershed in the Wiregrass region of Alabama. The study compares changes in water quality and revenue from agricultural and forest productions due to changes in land use between years 1992 and 2001. The study is completed in two stages. In the first stage, a biophysical model is used to estimate the effect of land use change on nitrogen and phosphorus runoff and sediment deposition in the main channel; in the second stage, farm enterprise budgeting tools are used to estimate the economic returns for the changes in land use condition. Both biophysical and economic results are discussed and a case for complex optimization to develop a decision support system is presented.

INTRODUCTION

Point source pollution has been substantially reduced since the implementation of the Clean Water Act-1972. However, non-point source pollution (NPP) that threatens majority of the water bodies in the United States remains the major environmental concern. NPP is caused by the movement of water, over and through the ground, generally after a precipitation event (rainfall and/or snow). The runoff picks up and carries away natural and manmade pollutants, eventually depositing them in lakes, rivers and coastal waters. Thus the pollutants left on the surface from various sources accumulate in receiving water bodies. The U.S. Environmental Protection Agency (USEPA) found that over one-third of streams, lakes, rivers, and estuaries surveyed nationally in 1996 did not fully support their designated uses such as for drinking water or recreation (USEPA 1998), citing NPP as the major cause of water quality degradation.

The agricultural sector is alleged to be the largest contributor to NPP through runoff of nutrients, sediment, pesticides, and other contaminants (USEPA 1998). Crop cultivation requires more use of chemicals and nutrients than natural vegetative cover such as forests and

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grasslands. Tillage operations affect the soil structure and often make the nutrient rich topsoil fragile and cause it to lose chemicals and soil particles during rainfall. Further, a study by the National Assessment Synthesis Team (2000) suggests that future climate change will require higher use of nutrients, pesticides and other chemicals to maintain the current level of productivity for most crops. This will further accelerate the decline in water quality of receiving water bodies.

In addition to agricultural land, land in residential and developed uses, such as lawns and gardens, are managed more intensively, resulting in generation of even more pollutants. Urban areas also have higher percentage of impervious surface that results in lower percolation and higher runoff. During precipitation, runoff carries nutrients and sediment from agricultural and residential land, resulting in higher chemical levels and turbidity in receiving waters. Thus, increasing urbanization coupled with increasing use of nutrients and chemicals in agricultural lands creates significant challenges for water quality protection and enhancement.

Recent water quality studies have focused on developing and successfully applying various biophysical simulation methods to estimate levels of NPP and to identify critical locations from which these pollutants originate (Bhuyan et al. 2001; Marzen et al. 2000; Mankin et al. 1999). These models use various geospatial data and facilitate the spatial analysis of sources and effects of point and non-point pollutants with reference to their origin and geographical locations. Calibrated biophysical models have enabled researchers to simulate effects of different land use and best management practice (BMP) combinations on surface water quality. The findings of such models help environmental policy planners to understand both short-term and long-term effects of changes in land use and land management scenarios and ways to effectively reduce NPP through institutionalization of best management practices.

This study aims to quantify the effect of land use change on water quality by simulating levels of nitrogen, phosphorus and sediment under two land use scenarios. Specifically, this study demonstrates the use of geospatial technologies to gather and organize reliable and current data for inputs into the BASINS-SWAT model. In addition, the study also quantifies the economic impact of two land use scenarios through simple economic model.

THE MODELING APPROACH

The modeling framework is adapted from USEPA’s Better Assessment Science Integrating Point and Non-point Sources (BASINS) model which comprises a suite of interrelated components for performing the various aspects of environmental analysis including data extraction, assessment, watershed delineation, classifying digital elevation models (DEM), land use, soils and water quality observations, and watershed characterization reports. The system includes four specific watershed level biophysical models for the estimation of in-stream and watershed loading and transportation. The four models are: an in-stream water quality model, QUAL2E; two watershed loading and transportation models HSPF and SWAT; and a simplified GIS-based model to estimate PLOAD, non-point loads of pollution on an annual average basis. The BASINS framework provides a centralized platform for data extraction and analysis and helps in set up of individual watershed-based models and analysis at a variety of scales using different tools. BASINS can support implementation of TMDLs by state agencies using watershed-based point and non-point source analysis for a variety of pollutants under alternative assumptions about land management practices (USEPA, 1998)

Soil and Water Assessment Tool (SWAT) is a river basin or watershed scale model developed by the U.S. Department of Agriculture-Agricultural Research Service’s (USDA_ARS) Grassland, Soil and Water Research Laboratory in Temple, TX. It is designed to assist resource managers in the long-term assessment of sediment and chemical yields in large watersheds and river basins. The model predicts the average impact of land use and management practices on water, sediment, and agricultural chemical yields in large, complex watersheds with varying soils, land uses, and management conditions over long periods of time (DiLuzio et al. 2002; Neitsch et al. 2002). In comparative studies using hydrologic and NPP models, SWAT has been shown to be among the most promising ones for simulating long-term NPP in agricultural watersheds (Borah and Bera 2003).

BASINS-SWAT uses an ArcView Geographic Information System interface to derive the model input parameters. Within the interface, hydrological modeling is completed using U.S. Geological Survey’s (USGS) National Elevation Dataset (NED). The watershed drains at the lowest elevation point of the catchment’s area and contains several sequential subwatersheds with directional flow (raindrop flow) to the main channel based on the topography of land. Subwatersheds are grouped based on climate, hydrologic response units (HRU), ponds, ground water, and main channels (Borah and Bera 2003). Each subwatershed can be virtually divided into several hydrological response units (HRUs) which are uniquely lumped areas within the subwatershed based on weighted land cover, soil type, and management combinations at a certain threshold level (Saleh et al. 2000). SWAT model simulation requires weather inputs (daily records of precipitation, wind, minimum and maximum temperatures) and management inputs (irrigation, tillage, chemical and fertilizer application). These input variables are converted to standard SWAT input files within the model. A given model run simulates runoff levels of nutrients, sediment and chemicals under a particular combination of land use and land management scenarios. Outputs from SWAT are crop yields, stream flows, and sedimentation and nutrient runoff levels, which can be traced across the watersheds both for short and long period of times.

LIMITATIONS OF THE STUDY

The land use data used in this study come from 1992 and 2001 National Land Cover Datasets (NLCD). Although vegetative covers are broken down to different types of forestlands, pasture and rangeland, cropland, and different intensities of developed land, the datasets do not contain details regarding which row crops and forage crops are grown in the study area. Hence, simulations are based on a dominant type of row crop and a dominant forage crop grown in the study area. Although forestland is included in the model as a major land cover, more effort is given to understanding the effects of cropland and pastureland management on water quality.

Land cover is dominated by agricultural and forestland with relatively small share of developed land. The model selects the dominant land use in each subbasin, thus possibly leaving out unique land use with small land coverage and ignoring the effects of specific land use in a localized area.

STUDY AREA

The study area is Little Double Bridge Creek (HUC #03140201230), lying within the Choctawhatchee basin in southern Alabama (HUC#03140201), which is also called Wiregrass region in the state (Figure 1). The Little Double Bridge Creek (colored blue in Figure 1) has 21.4 square miles of upstream drainage area and lies to the west of Enterprise city in Coffee and Geneva Counties of Alabama. The study area for simulation covers approximately one tenth of the Coffee County and lies in a major corn belt.

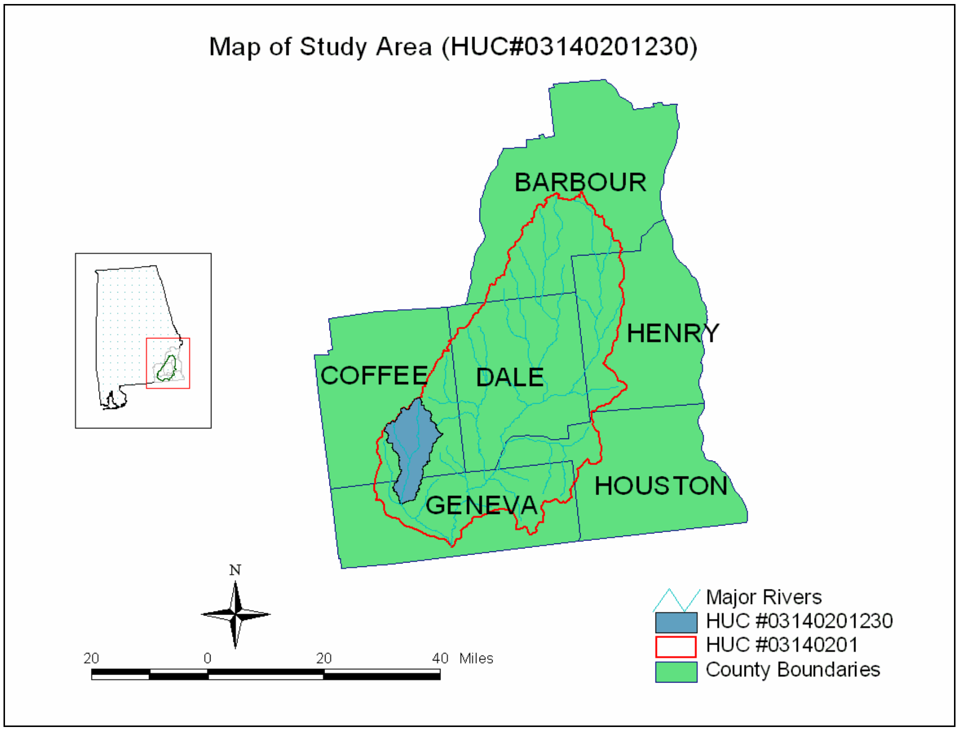

Figure 1 Map of the study area

The wiregrass region is characterized by agricultural and forestland, which are the dominant land use in the study area. Coffee County ranks 5th in broiler production, 7th in peanut production, 9th in corn production and 14th in cotton production within the state (Alabama Agricultural Statistics 2005). On one hand, production of corn and cotton crops require higher application of nitrogen fertilizer, while on the other hand, use of poultry litter in row crop production results in elevated phosphorus levels in the soil. The study area is selected based on the importance of agricultural pollution controls and availability of required input data to run the model described in the following sections.

A comparison of two national land cover maps suggests that there has been significant change in land use between 1992 and 2001 in this watershed (Table 1). Between the years

1992 and 2001 the share of developed land increased by more than 17 times, however, this share still remains less than 5.0% of the total land. Share of row crops decreased by 26.0% while that of pastureland increased by 13.9%. Total forestland area decreased by 10.1% while other wastelands such as range/brushland and wetlands increased by 73.4%. The changes in land use distribution are expected to bring changes in water quality including surface flow, nutrient runoff and sedimentation levels.

Table 1 Comparison of land use distribution in the study area

Land use category	NLCD 1992		NLCD 2001		Change*
	Area (ha)	Share (%)	Area (ha)	Share (%)	(%)
Cropland	1,886	34.3%	1,396	25.4%	-26.0%
Pastureland	740	13.5%	843	15.3%	13.9%
Forestland	2,341	42.6%	2,104	38.3%	-10.1%
Urbanland	14	0.3%	261	4.8%	1739.4%
Wastelands	514	9.4%	891	16.2%	73.4%
Total	5,495		5,495		

*Change weighted for area

DATA

The study uses data from various sources. Core set of data which include elevation, land use, state soil survey data and watershed boundaries are extracted from the BASINS website. Other sources of data are described below.

Elevation and land cover data: The 1:24,000 scale 30x30m resolution Digital Elevation Model (DEM) data as well as two sets of NLCD (1992 and 2001) for the entire area are downloaded from Seamless Data Distribution System of USGS Web Server. Overall thematic accuracy level of NLCD 1992 land use data at the Anderson Level I is 82% (Stehman et al. 2003). Although formal accuracy assessment reports are not available for NLCD 2001 land cover data for the region, a single-pixel accuracy assessment in three of the NLCD 2001 mapping zones elsewhere suggests that the accuracy range of 73 to 77 percent (Homer et al. 2004). The vertical positional accuracy for the elevation data is 2.70 RMSE

(NED Press Release June 2003). All core basins data as well as elevation and land use grids are projected in the same projected coorginate system.

Climate and streamflow data: Precipitation, temperature, wind speed, solar radiation and relative humidity data are standard input to SWAT model. Observed daily precipitation and minimum/maximum temperature data are obtained from the National Climate Data Center (NCDC) database for four nearby climate stations between January, 1965 and December, 2005 (Source: SECC 2006). Daily streamflow data, an important requirement for model calibration and validation, is collected from USGS station (ID# 02362240) located at the end of Little Double Bridge Creek. Daily streamflow data are available for this station from September 1985 to October 2003.

Farm Management Practices: A single major row crop (corn) and a forage crop (bermudagrass) are selected and a table of operations for these crops is derived based on the recommended cultural practices published in Alabama Cooperative Extension System reports. Typical cultural practices such as fertilizer use, tillage operations, pesticide use, harvesting and killing operations are recorded for corn and bermudagrass.

METHODS

Watershed Delineation

The BASINS process starts with automatic delineation of subbasins from the digital elevation data. The NED is processed to remove any sinks in the data. Sinks are the grids erroneously recorded as being lower than surrounding areas. Automatic watershed delineation is processed aligned with the national hydrography stream network. Digital stream networks are created with a 40-hectare headwater threshold area, which defines the minimum area required to begin a stream flowing out of the area in any part of the watershed. The physical location of the USGS gage station is marked as the lowest outlet point of the stream network to define the watershed boundary. Subbasins are created along with their physical characteristics including area, length, width, slope, and elevation. Subbasins are physically bounded areas to which changes in management practices, yields and pollution levels can be traced during simulations.

The delineated area of Little Double Bridge Creek sub-watershed is 55.37 km^2. The mean elevation of the watershed is 97 meter above mean sea level (msl) with the range between 63 msl and 132 msl. The watershed is divided into 159 subbasins (ranging from 4.6 to 83.7 ha) with an average area of 34.8 ha. As a comparison, average farm size in Coffee County is 231 acres or 93.4 ha (Alabama Agricultural Statistics Bulletin 2005). Thus, an average farm covers 2-3 subbasins in terms of area coverage. The mean subbasin slopes range from 2.9 to 8.6% with the mean slope of 5.4%. The average slope of digitized streams across flow length is 1.8%, and ranges from 0.1% to 5.7%.

Table 2 shows relative number and area coverage of dominant HRUs under two alternative land use scenarios of NLCD 1992 and NLCD 2001. The number of HRUs with cropland decreased by 35%, a reduction from 81 to 53, in contrast, number of HRUs with pastureland increased by 72%, an increase from 18 to 31. Composition of forestland slightly changed from 50 HRUs to 51 HRUs, a 2% increase in the two periods. Other lands designated as range/brush and wetlands also changed by 130%, an increase in the number of HRUs from 10 to 23.

Table 2 Comparison of dominant hydrological response units

Major Land Use	Number of Sub-basins		
	NLCD92	NLCD01	Change
Croplands	81	53	-35%
Pastureland	18	31	72%
Forestland	50	51	2%
Urbanland	-	1	N/A
Wastelands	10	23	130%
Watershed	159	159	N/A

Model Calibration and Validation

Model calibration is done by comparing five years of average daily simulated versus observed streamflow between January 1991 and December 1995 using NLCD 1992 land cover map. Land cover condition affects the amount of surface flow, infiltration, evapotranspiration and underground recharge by exposing the rainwater to different surfaces. By choosing this period, model estimation is also kept close to the land cover conditions in the current period. Calibration for nutrients and sediment is not done because observed data are not available for these water quality parameters.

Adjustments are done in the curve number (CN2) of different land use by adjusting up to 10% of the recommended number for different hydrological group and land cover conditions (Santhi et al. 2001). Once the average daily total streamflows are reasonable, adjustment are done in HRU soil (SOL) and groundwater (GW) input parameters to adjust the surface flow and baseflow so that the Nash-Sutcliffe Coefficient of Efficiency (COE) and goodness of fit (R^2) are within the acceptable range of values. Simulated average daily streamflows[2] on a monthly time steps are compared with observed daily streamflows. The statistical results of calibration are given in Table 3 and the graphical results are given in Figure 3.

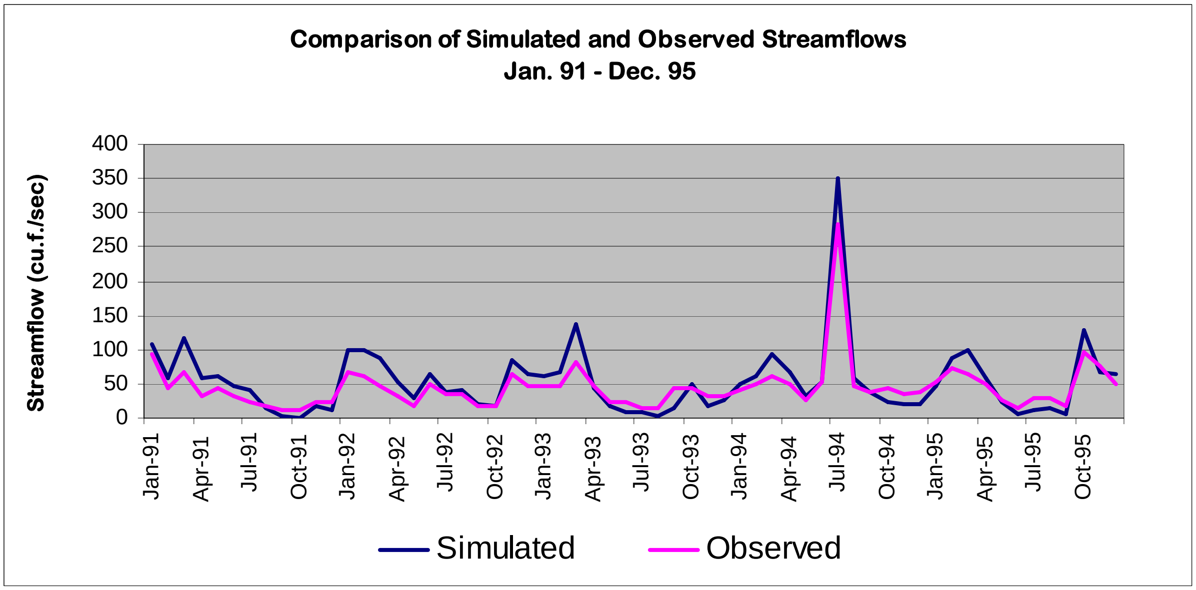

Figure 3 Observed Vs. simulated daily average streamflow, January 1991 - December 1995

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[2] Separate baseflow and surface flow for the monthly streamflow for the USGS monitoring data are extracted using the Web-Based Hydrograph Analysis Tool (WHAT) available from Purdue University system (http://pasture.ecn.purdue.edu/~what/). Simulated total streamflow are separated to baseflow and surface runoff by multiplying the total flow in drainage (SWAT output file: Rich) by the ratio of surface flow and groundwater flow to the total water yield obtained from the model (SWAT output file: Sub-basin). Several test statistics are derived for both separated flows and total streamflow.

Table 3 Model calibration results between January 1991 and December 1995 (use no more that one significant digits after decimal for flows)

Average Daily Streamflow (cu.ft./sec.)	Mean		Difference	R^2	COE
	Simulated	Observed			
Total Streamflow	46.14 (48.39)	45.00 (37.00)	3%	0.88	0.75
Surface Runoff	20.90 (21.92)	17.68 (23.37)	18%	0.77	0.74
Baseflow	25.28 (26.52)	27.32 (15.40)	-7%	0.86	0.21

Note: Figures in parentheses indicate standard deviation from mean.

The Table 3 summarizes the statistical results of calibration. It is shown that the calibrations for total streamflow and surface runoff are very reliable. The mean of simulated average daily streamflow (mean = 46 cfs and std.dev.=48 cfs) lies within 3% of the observed streamflow (mean = 45 cfs and std.dev. = 37 cfs). However, the surface runoff is slightly over predicted (18% higher) and baseflow is slightly under predicted (7% lower). Both goodness of fit (R^2=0.88) and coefficient of efficiency (COE=0.75) are within the readily acceptable range of values for total streamflow. A quite similar result for surface runoff (R^2=0.77 and COE=0.74) is also obtained; for baseflow calibration a high goodness of fit (R^2=0.86) and very low coefficient of efficiency (COE=0.21) was obtained. These values are similar to the ones reported by Kirkch et al. (2002) in their study in Rock River Basin in Wisconsin (R^2=0.74 and COE=0.61) and by Fohrer et al. (2005) in their study (R^2=0.82 and COE=0.61) in the lower mountain range areas in Germany. Moon et al. (2004) report a R^2 value of 0.86 and COE value of 0.78 using monthly time step comparison in Trinity River Basin in Texas. Thus, the calibration results obtained here are within a reasonable range of values in previously published studies.

Once the calibration is complete, model is run for twenty years starting from January 1986 to December 2005 run in monthly time steps. It was important to keep 20 years of simulation because of a 10-year rotation in bermudagrass cultivation in the management file.

SWAT requires the number of years of simulation in the multiple of the number of years in the crop rotation in management input file. Two sets of simulations are done using the calibrated input values, one for NLCD 1992 land cover condition and the other for NLCD 2001 land cover condition.

RESULTS

Effects on Water Quality

The area received 1448.9 mm of precipitation annually across the surface of watershed. This precipitation level results in different streamflow levels using two land use conditions. There is a total water yield of 536.35 mm with 1992 NLCD land cover as compared to 519.78 mm with the 2001 NLCD land cover condition. Total water yield here is defined as the sum of surface, lateral, and groundwater flow minus transportation loss, which will eventually pass through the main channel. For NLCD 1992 land cover condition, the predicted surface runoff and baseflow contributions to the channel are 280.05 mm and 230.37 mm, respectively. In contrast, surface runoff and baseflow contributions to the channel with NLCD 2001 land cover conditions are 224.80 mm and 267.69 mm, respectively. Thus, the results show that total water yield decreases by 3.1%, whereas surface flow decreases by 19.7%, and baseflow contribution increases by 16.2%. This indicates that there is less surface runoff and higher infiltration and groundwater recharge with 2001 land cover as compared to 1992 land cover condition.

The model output calculates average nitrogen and phosphorus applied in the watershed. This is more or less proportionate to the amount of agricultural land and crop management practices. On average, 127.73 kg of nitrogen and 16.92 kg of phosphorus are applied to one hectare of land with the NLCD 1992 land cover condition. With the 2001 land use distribution, the amount of nitrogen and phosphorus applied in the watershed reduces to 118.00 kg/ha and 13.69 kg/ha, respectively. This average is based on the total watershed area.

Comparing land use conditions between NLCD 1992 and NLCD 2001 land cover shows that agricultural land decreases by 921 hectares (33%). The pastureland increases by 480 hectares (92%) followed by 362 hectares (91%) and 79 hectares (4%) increases in other land and forestland. Table 4 compares the aggregate annual nutrient runoff and sedimentation by land use type for two land cover scenarios. The aggregate results at watershed level show that total nitrogen runoff decreases by 23%, total phosphorus runoff by 22% and sedimentation by 40% when land use conditions change from those of 1992 to those of 2001.

Effects on Agricultural Production and Returns

A simple bioeconomic analysis is done to estimate effects of land use change on farm profits at the watershed level. The Table 5 compares the total agricultural and forest revenues and expenses with two land use scenarios. As described earlier in the limitation of the study and the methodology, farm income has been estimated only for a single row crop ‘corn’, single forage crop ‘bermudagrass’ and single ‘pine plantation’ for forestland. The average yield and cost of operation for corn are taken from Alabama Cooperative Extension System bulletin AEC BUD 1-1 (ACES 2006). The average yield and cost of operation under recommended practices for bermudagrass are taken from AEC BUD 1-2 (ACES 2005). Based on these publications, returns to corn production and bermudagrass cultivation are calculated in the absence of government payments. Similarly, returns to forest plantations ($63 per hectare) are taken from the online bulletin MTN 9C, a publication of Mississippi State University Extension Service (MSCARES).

In the absence of government payments, farms are currently operating at a loss with both the corn and bermudagrass productions, losing an average of $251 and $171 per hectares, respectively. These figures are derived based on production costs using recommended inputs according to the above mentioned extension bulletin and setting the exiting output prices of corn ($2.50 per bushel) and bermudagrass ($70.00/ton). Five-year average crop yield in crop reporting district 60 is used for calculation of revenues. Based on these figures, changing land use from corn production to bermudagrass causes a large reduction in operating loss. For instance, the Table 5 shows a corresponding $231.1 thousand reduction in operating loss for 921 hectares decrease of corn acreage. If equal area is converted from corn to bermudagrass, net reduction in operating loss will be 73.6 thousands dollars only. No economic returns have been inputed for wetlands and wastelands like range/brush lands.

Table 4 Comparison of total annual nutrient runoff and sedimentation across the watershed under different land use types for land cover conditions of 1992 and 2001

Land use	Dominant Land Use (ha)			Total Nitrogen[a] (Kg)			Total Phosphorus[b] (kg)			Total Sediment (tons)		
	NLCD92	NLCD01	Change	NLCD92	NLCD01	Change	NLCD92	NLCD01	Change	NLCD92	NLCD01	Change
Cropland	2,814	1,893	-33%	1,506	1,152	-24%	567	438	-23%	2,389	1,372	-43%
Pastureland	520	1000	92%	136	98	-28%	19	12	-37%	45	23	-49%
Forestland	1,807	1,886	4%	156	35	-78%	16	3	-81%	88	3	-97%
Urban	0	29	n/a	0	137	n/a	0	22	n/a	0	137	n/a
Wastelands	396	730	84%	73	14	-81%	8	1	-88%	45	3	-93%
Watershed	5,537	5,538	0%	1,871	1,436	0%	610	476	0%	2,567	1,538	0%

[a] Total Nitrogen = Organic N + NO_3 in Surface Runoff + NO_3 in Lateral Flow + NO_3 in Groundwater

[b] Total Phosphorus = Organic P + P in Sediment + Soluble P + P in Groundwater

Table 5 Effects of land use change in farm returns under different land use distribution in study area

	NLCD 1992					NLCD 2001					Change	
	Area (ha)	Yield	Cost $/ha	Revenue $/ha	Total Profit	Area (ha)	Yield	Cost $/ha	Revenue $/ha	Total Profit	Area (ha)	Revenue ($)
Corn[a] (bushels)	2,814	208	770	519	-706,225	1,893	208	770	519	-475,101	-921	231,125
Forest[b]	1,808	n/a	n/a	n/a	113,004	1,886	n/a	n/a	n/a	117,912	79	4,908
Pasture[c] (tons)	520	15	1,209	1,038	-88,814	1,000	15	1,209	1,038	-170,929	481	-82,115
Other[d]	396	n/a	n/a	n/a	n/a	758	n/a	n/a	n/a	n/a	362	n/a
Total	5,537				-682,035	5,537				-528,117		

[a] Calculations are based on ACES Publication AEC BUD 1-1, January 2006

[b] Calculations are based on ACES Publication AEC BUD 1-2, May 2005

[c] Calculations are based on MSU CARES Publication MTN 9C

[d] Other lands include wetlands, wastelands, rangebrush and urbanlands.

The Table 6 presents the summary of bioeconomic impacts of land use change in the study area. It shows that a large decline in corn acreage with simultaneous increase in pastureland acreage and some forest acreage causes a net reduction in operating loss of 153.8 thousand dollars at the watershed level. At the same time, the impact on water quality is desirable for all kinds of land cover. Total nitrogen and phosphorus runoff reaching the channel decrease by 434 kg and 135 kg per year, respectively. Sedimentation decreases by 1030 metric tons per year across the watershed.

Table 6 Differences in the farm profit and water pollution when land cover changes from 1992 to 2001

Land use	Area (ha.)	Net Return Per Ha.	Profit ($)	N Runoff (kg)	P Runoff (kg)	Sediment (tons)
Row Crop	-921	-251	231,031	-354	-129	-1,018
Forest	79	63	4,908	-121	-14	-85
Pasture	481	-171	-82,082	-38	-7	-22
Other	362	n/a	n/a	78	15	95
Total			153,857	-434	-135	-1,030

[a] Loss minimization

CONCLUSION

The water quality in a watershed is affected directly by vegetative cover and agricultural and other land management practices. The pattern of land cover changed in the study area from 1992 to 2001. There is decline in both agricultural land (26%) and overall forestland (10%). However, the structure of forestland changed with a 45% increase in evergreen forest. Developed land increased by seventeen times, however, the weighted share of developed land still remains less than 5% of total land. The share of rangeland increased by 392% followed by a 14% increase in pastureland. Wetlands decreased by 72%.

Changes in agricultural crops such as switching from corn production to cotton production or other crop rotations remain unidentified at this level of study. However, comparison of sediment and nutrient runoff across subbasins that changed from one SWAT land cover class to another class shows a great variability in the results. Changing from forest to agricultural land or from wetlands to pasture land has great impacts on water quality, including the quantity of surface flow, nutrient runoff and sediment loadings at the main channel.

The study indicates that decrease in the cropland has resulted in lower overall nutrient application in the watershed. The surface runoff reduces by 19.7% with more surface vegetation as derived from 2001 land cover maps. While farm management practices are held constant over two land use scenarios, changes in the land use have caused the decrease in the application of total nitrogen and phosphorus across the watershed. The aggregate nitrogen runoff at the channel decreases by 23%, total phosphorus runoff by 22% and sedimentation by 40% when land use condition changes from 1992 to 2001 conditions.

In the absence of government payments, farms are currently operating under loss with both the corn and bermudagrass productions, losing an average of $251 and $171 per hectares respectively. Taking away land from corn production to bermudagrass causes a large reduction in operating loss. For example, about $231.1 thousands reduction in operating loss is experienced when corn acreage in the watershed is decreased by 33%. A net reduction of $73.6 thousands in operating loss is experienced when converting the same land to pastureland. Hence, a large decline in corn acreage with simultaneous increase in pastureland causes a net reduction in operating loss of $165.2 thousand dollars at the watershed level. At the same time, the impact on water quality is positive for all kinds of land cover. Total nitrogen and phosphorus runoff reaching the channel decrease by 434 kg and 135 kg per year, respectively. Sedimentation decreases by 1030 metric tons per year across the watershed.

The results presented in this paper are basic to the understanding of economic and water quality impacts on land use change. These results help regional planners and watershed management policy makers by providing estimates of changes in water quality when land use changes over time. The application of the model can be extended to include more detailed land use and soil distribution in the study area using recent satellite images to create maps depicting detailed cropping patterns which will help to understand the impact of alternative best management practices such as minimum or no-tillage practices and reduced use of fertilizers and pesticides. Using multiple hydrological response units to simulate best management practices will give more precise effects of those practices in water quality.

These biophysical models are extremely valuable in assessing the physical impacts of BMPs on quantity and quality of water bodies in a given watershed. They can help policy planners to assess water quality and plan for intervention through TMDLs. However, it is imperative to account for the effects BMPs may have on farmers' profitability. Specifically, the impact that changes in yield or changes in input costs have on profitability has not been examined in this study. Profitability can greatly impact the likelihood that farmers will voluntarily adopt BMPs, thereby improving water quality. Thus, watershed and farm level economic impacts must be evaluated to understand the magnitude of gains and losses to individual farmers through use of BMPs.

The complex bioeconomic model will extend beyond the biophysical simulations and will require a more complex mathematical programming framework. The modeling process involves the development of bioeconomic models in which the outputs of SWAT simulations are used as inputs to an economic optimization model under imposed environmental and land use constraints. Various hypothetical TMDL targets can be considered; for example, a 10% reduction in nitrogen runoff or a 20% reduction in sediment. The objective function of the model will be economic profit, based on revenues and costs resulting from alternative practices (e.g., Forster et al. 2000; Hite et al. 2002; Interpapong et al. 2002; Paudel et al. 2003). Profits can be modeled under a number of land use and management practice scenarios and compared to those of actual practices over the long run in the study area. A mathematical programming model finds the solution to optimal profits and allocation of land use. In addition, the resulting model can be used to predict the way water quality changes with changes in management practices.

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