DIFFERENTIATING IMPACTS OF LAND USE CHANGES FROM PASTURE MANAGEMENT IN A CEAP WATERSHED USING THE SWAT MODEL

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ABSTRACT. *Due to intensive farm practices, nonpoint-source (NPS) pollution has become one of the most challenging environmental problems in agricultural and mixed land use watersheds. Usually, various conservation practices are implemented in the watershed to control the NPS pollution problem. However, land use changes can mask the water quality improvements from the conservation practices implemented in the watershed. The objectives of this study were to evaluate the linkage between nutrient input from various pasture management practices and water quality, and to quantify the impacts of land use changes and pasture management on water quality in a pasture-dominated watershed. Land use data from 1992, 1994, 1996, 1999, 2001, and 2004 were evaluated for the land use changes in the watershed, and the corresponding implemented management practices were also incorporated into the Soil and Water Assessment Tool (SWAT) model. The individual impacts of land use change and pasture management were quantified by comparing the SWAT simulation results for different land use change and pasture management scenarios. The results indicated that land use changes resulted in greater total sediment (499 kg ha^{-1}) and nitrogen losses (3.8 kg ha^{-1}) in the Moores Creek subwatershed, whereas pasture management resulted in greater total nitrogen losses (4.3 kg ha^{-1}) in the Beatty Branch subwatershed. Overall, the combined impacts of land use changes and pasture management resulted in greater total sediment (28 to 764 kg ha^{-1} of cumulative combined impacts between 1992 and 2007) and nitrogen losses (5.1 to 6.1 kg ha^{-1}) and less total phosphorus losses (1.5 to 2.1 kg ha^{-1}) in the Beatty Branch, Upper Moores Creek, and Moores Creek subwatersheds. By quantifying the individual impacts of land use changes and pasture management, we found that an increase in total nitrogen losses in the Beatty Branch subwatershed was mainly due to an increase in nutrient inputs in the pasture areas, and total sediment and nitrogen losses in the Moores Creek subwatershed were mainly due to an increase in urban lands. Therefore, the individual impacts of land use changes and conservation practices should be quantified to get a true picture of the success of CEAP programs in watersheds experiencing significant land use changes.*

Keyword. *Conservation Effects Assessment Project (CEAP), Land use changes, Pasture management, SWAT model.*

Intensive agricultural activities and fertilizer application rates in excess of plant nutrient requirements can lead to nonpoint-source (NPS) pollutant losses, especially sediment, nitrogen (N), phosphorus (P), and pesticides to receiving water bodies (Gillingham and Thorrold, 2000; Monaghan et al., 2005). Adverse impacts of agricultural production on surface and ground water can be minimized by implementing various conservation practices. Even though many studies have been published during last two decades evaluating the effectiveness of these practices in controlling losses of NPS pollutants at field and plot scales, their effectiveness at watershed scale is currently not clear. In 2003, the Conservation Effects Assessment Project (CEAP) was initiated by the USDA Natural Resources Conservation Service (NRCS), Agricultural Research Service (ARS), and Cooperative State Research, Education, and Extension Service (CSREES) to assess the economic and environmental benefits obtained from the conservation program (Mausbach and Dedrick, 2004). The overall goal of the CEAP efforts is to evaluate the performance of conservation practices in improving water quality in agricultural watersheds (Duriancik et al., 2008).

Assessment of conservation practices in reducing losses of pollutants from agricultural watersheds can be performed by (1) analyzing monitoring data for water quality as affected by timing and location of conservation practices in the watershed, (2) performing simulation modeling, and (3) a combination of measured data analysis and simulation modeling.

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Several studies have reported the water quality data from CEAP watersheds (e.g., Feyereisen et al., 2008; King et al., 2008; Kuhnle et al., 2008; Locke et al., 2008; Richardson et al., 2008; Smith et al., 2008; Steiner et al., 2008, Tomer et al., 2008; among others). The conservation practices attributed to improvement in water quality in these studies include controlled drainage and constructed wetlands (Tomer et al., 2008), conversion of cultivated lands into conservation reserve program (CRP) lands (Kuhnle et al., 2008), vegetated filter strips (Locke et al., 2008), land grading and vegetation establishment in gullies (Stenier et al., 2008), and nutrient management (Richardson et al., 2008). Many studies have indicated that performance of conservation practices could be affected by factors such as weather (Garbrecht et al., 2006; Garbrecht, 2008), topography and land management (Roth et al, 1996; Herlihy et al., 1998; Alexander et al., 2008), and land use changes concurrent with the installation of conservation practices in the watershed. Garbrecht et al. (2006) showed that when precipitation increased by 33%, corresponding runoff and sediment losses increased by 100% and 183%, respectively, in a CEAP watershed in Oklahoma. Chaubey et al. (2010) showed that weather could result in greater variability in pollutant losses compared to the reductions caused by the conservation practices implemented in a CEAP watershed. Therefore, assessment of the interactions among climate variables, land use, and land management is required to accurately evaluate the environmental benefits of the conservation practices (Steiner at al., 2008).

Differentiating the impacts of land use changes and conservation practices is not straightforward. If an agricultural watershed managed with conservation practices also experiences significant urbanization, then the negative impacts of land use changes must be separated from the positive impacts of conservation practices to evaluate the success of the conservation programs in the watershed. Land use changes may have either positive or negative impacts on water quality and quantity. Certain land changes, such as deforestation or urbanization, are known to result in negative water quality impacts. For example, increase in urban and agricultural land use could result in greater streamflow. Lenhart et al. (2003) found that the nitrate losses increase with increasing deforestation and land use intensity. Contrarily, Miller et al. (2002) reported that a shift from agricultural land to forest resulted in a 4% streamflow decrease. It is clear that without such a differentiation, it is possible that land use changes can mask the impact of conservation practices, leading to a wrong conclusion about the effectiveness of conservation programs in improving water quality.

Use of watershed models is one of the most efficient ways to quantify the impacts of conservation practices at various spatial and temporal scales. The most commonly used models in the CEAP assessment are the Soil and Water Assessment tool (SWAT; Arnold et al. 1998) and the Annualized Agricultural Nonpoint Source model (AnnAGNPS; Bingner et al., 2009). For example, Lerch et al. (2008) used the SWAT model to simulate atrazine concentration and load, and the impact of grass waterways on atrazine concentrations in the Mark Twain Lake/Salt River basin in Missouri. Similarly, Heathman et al. (2008) evaluated the performance of SWAT and AnnAGNPS in predicting streamflow and atrazine losses from various conservation practices in the Cedar Creek watershed, Indiana, and concluded that the SWAT model was preferable to the AnnAGNPS model. One of the limitations of these models is the assumption of a constant land cover condition during the simulation period. For example, land cover distribution present in the beginning of the simulation period is preserved throughout the simulation period in the SWAT model (version 2005) and is not allowed to change over time. This limitation may not pose a significant problem in watersheds that do not experience considerable shift from one land cover to another (e.g., agriculture to urban). However, quantification of effects of conservation practices using the SWAT model can be erroneous if land cover distribution also changes concurrent with the conservation practice implementation in the watershed.

In this study, we used the latest SWAT model (SWAT2009, released in January 2010) to quantify impacts of land use changes and conservation practices in dynamic watersheds that experience both land use and conservation practice changes concurrently over time. The specific objectives of this study were: (1) to evaluate the linkage between nutrient inputs from various pasture management practices and water quality, and (2) to quantify the individual impacts of land use change and pasture management on water quality in the subwatersheds in the Lincoln Lake watershed, a pasture-dominated CEAP watershed located in northwest Arkansas. Water quality impacts of various land use change and pasture management scenarios were evaluated using the SWAT model. The hypothesis was that in dynamic agricultural watersheds, land use changes can mask the impacts of pasture management on pollutant losses and water quality.

Material and Methods

Site Description

This study was conducted in the Lincoln Lake watershed, a 32 km^2 agricultural watershed within the Illinois River basin located in northwest Arkansas and eastern Oklahoma. Moores Creek and Beatty Branch are the two major tributaries in the watershed, representing 19 km^2 and 11 km^2 of the watershed area, respectively (fig. 1). This watershed is one of the 13 watersheds in the Conservation Effects Assessment Project (CEAP) funded by the USDA-CSREES to evaluate the effectiveness of agricultural BMPs in improving water quality. The Lincoln Lake watershed is a mixed land use watershed, with pasture, forest, urban residential, and urban commercial representing 36.5%, 48.6%, 11.5%, and 1.5% of the watershed area in 2004, respectively (fig. 1). Considerable land use changes, especially for pasture and urban areas, have occurred since monitoring started in the watershed in 1992 (fig. 2). The pasture area in the watershed has decreased from 48% to 37%, primarily due to increasing urbanization in the watershed where urban area increased from 3% in 1992 to 12% in 2004 (Gitau et al., 2010). Pasture fields in the watershed have numerous poultry, beef, and dairy cattle production facilities. Excessive fertilizer and manure usage for perennial forage crop production in the watershed have been shown to increase surface and ground water pollution due to increasing losses of sediment, nutrients, and pathogens (Edwards et al., 1996). The percentage of the watershed area that was managed with at least one conservation practice increased from 1% to 34% during 1992 to 1994, representing 53% of the total pasture lands in 1994 (fig. 3). These conservation practices included alum-treated poultry litter application, buffer strip, and continuous grazing management.

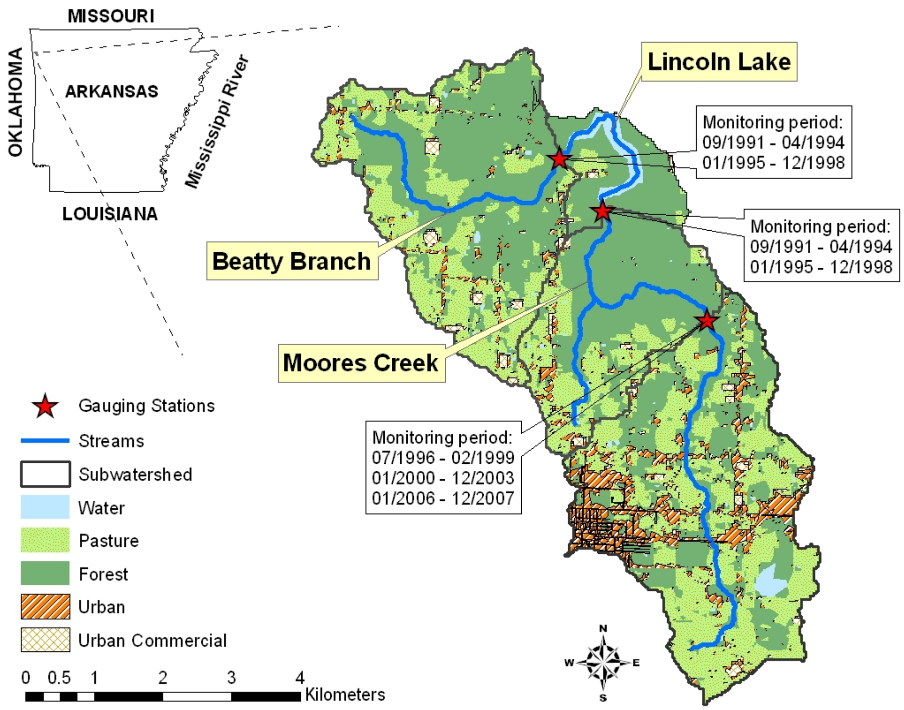


Figure 1. Location of Beatty Branch, Moores Creek, gauging stations with monitoring periods, and 2004 land use distribution in the Lincoln Lake watershed.

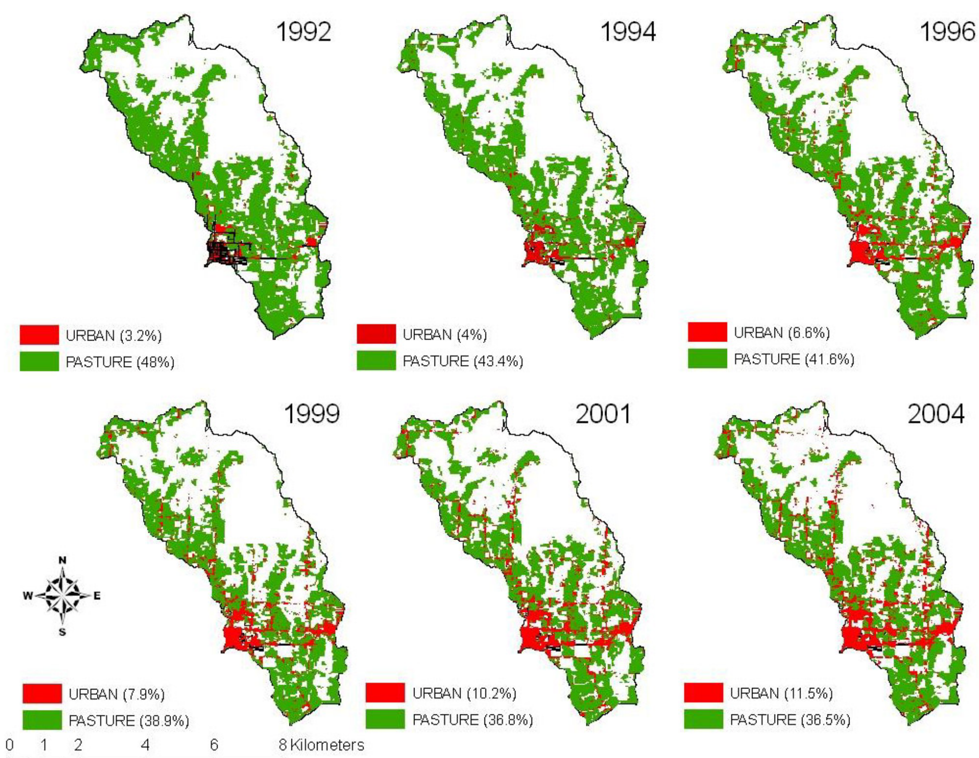


Figure 2. Major land use (urban and pasture lands) changes in the Lincoln Lake watershed from 1992 to 2004.

Recommendations for pasture management in the watershed have changed over time, ranging from poultry litter application based on meeting plant nitrogen demand in the early 1990s to phosphorous-based application in recent years. Currently, farmers are encouraged to apply alum-treated poultry litter based on the Arkansas Phosphorus Index to reduce soluble phosphorus concentration in poultry litter (DeLaune et al., 2004; DeLaune et al., 2006).

Since September 1991, stream flow and water quality data have been collected at one gauging station in Beatty Branch

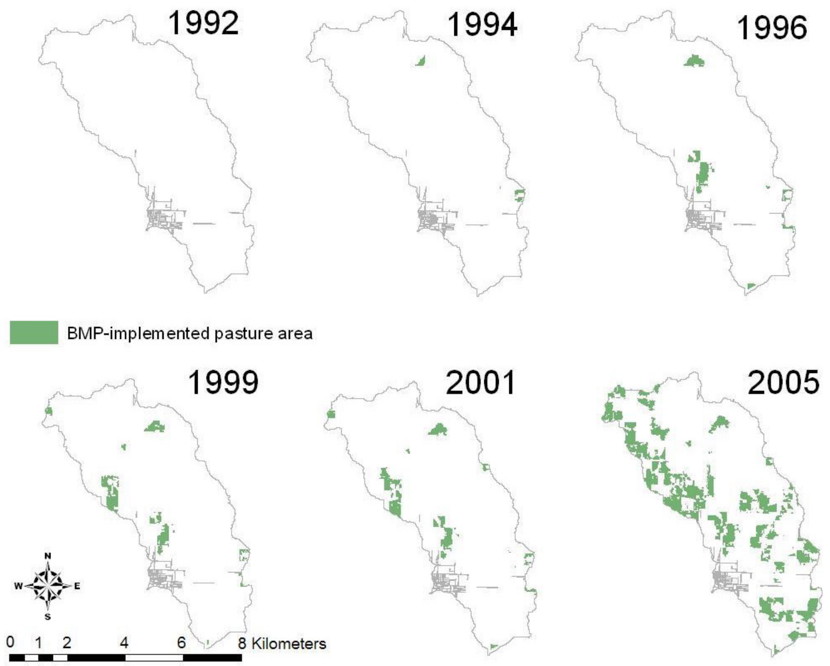


Figure 3. Location of pasture area that had BMP implemented from 1992 to 2005.

and two stations in Moores Creek (fig. 1). The data collection efforts coincided with various projects funded in the watershed. At all three sites, flow, sediment, nutrient (nitrate nitrogen, total nitrogen, phosphate phosphorus, and total phosphorus) data were collected. A pressure transducer was used to measure flow depth, and depth-discharge relationships were used to determine stream flow. Water quality data for sediment, nitrogen (N), and phosphorus (P) were collected separately during storm and baseflow conditions. Flow-weighted composite samples were collected during each storm event using an autosampler. Water quality during baseflow conditions was quantified by collecting grab samples every two weeks. All water samples were analyzed using standards methods of analysis (Greenberg et al., 1992). Details of flow and water quality monitoring were provided by Edwards et al. (1994, 1996, 1997), Vendrell et al. (1997, 2000), Nelson (2000), and Nelson et al. (2008).

Vendrell et al. (2000) concluded that the BMPs were able to retard nitrogen transport, as indicated by a decrease in mean concentrations of ammonia nitrogen (NH_4-N) and total Kjeldahl nitrogen (TKN) from January 1995 through December 1998 at the Lower Moores Creek and Beatty Branch sites, and from July 1996 through February 1999 at the Upper Moores Creek site, respectively. Similarly, Nelson et al. (2008) concluded that TP concentration were reduced by nearly 50% (from 0.19 mg L^{-1} in 2000 to 0.1 mg L^{-1} in 2007) and nitrate-nitrogen (NO_3-N) concentration declined by 66% (from 3.57 mg L^{-1} in 2000 to 1.21 mg L^{-1} in 2007) at the Upper Moores Creek site between 2000 and 2007.

Model Description and Input Data Preparation

The SWAT model was used to evaluate the impacts of land use changes and pasture management on water quality in the watershed. The model can predict long-term impacts of land use and land management on water, sediment, and agricultural chemical yields at different scales in a complex watershed (Arnold et al., 1998). More than 250 peer-reviewed journal articles have been published demonstrating SWAT applications on sensitivity analyses, model calibration, hydrologic analyses, pollutant load assessment, and climate change impacts on hydrology and pollutant losses (Gassman et al., 2007).

The key GIS input files to SWAT included a 30 m digital elevation model (DEM; USGS, 2004), 28.5 m land use/land cover (CAST, 2004), and Soil Survey Geographic (SSURGO) soil data at a scale of 1:24,000 (USDA-NRCS, 2002). The land use maps for the years 1992, 1994, 1996, 1999, 2001, and 2004 were developed using moderate spatial resolution (28.5 m × 28.5 m) Landsat Thematic Mapper (TM) satellite images. More details of land use data development for this watershed are presented by Gitau et al. (2010). The SWAT ArcView interface was used to delineate the watershed into several subbasins as a function of the DEM and specification of streams and inlets/outlets. Subsequently, the subwatersheds were partitioned into homogeneous units (hydrologic response units, HRUs) by setting threshold percentages of land use and soil type (Neitsch et al., 2005). In this study, a threshold for a land use and soil type covering an area of 0% and 0%, respectively, within any given subbasin was applied in order to capture all the land use changes that occurred during the study period (1992 to 2004). It should be noted that 0%/0% threshold is the most detailed representation of HRUs in the SWAT model as it does not lump any land use or soil type into another category. This resulted in a total number of 1,465 HRUs in the watershed. Weather data (daily precipitation, minimum and maximum temperature) were obtained from the Fayetteville Weather Station located approximately 25 km from the watershed. Other weather variables needed by the model (solar radiation, wind speed, and relative humidity) were estimated using the weather generator built into the SWAT model.

The SWAT model has the ability to define specific types of manure and fertilizers by building fertilizer and manure components, such as fractions of mineral N (P), organic N

(P), and a ratio of ammonium nitrate to mineral N in the SWAT fertilizer database. Seven types of manure and fertilizers were applied in the Lincoln Lake watershed. The inorganic fertilizers included urea, anhydrous ammonia, and Triple 17 (17% of N, P_2O_5, and K_2O). Organic manure included beef-fresh manure, hen/pullet manure, broiler-fresh manure, and alum-treated broiler manure. Nutrient content for broiler-fresh manure and alum-treated broiler manure were derived from the study by Moore and Edwards (2005), where untreated litter contained 3.85% total nitrogen (TN) and 1.89% total phosphorus (TP), and alum-treated litter contained 3.45% TN and 2.24% TP. The ratio of organic to mineral N (3.91) for untreated litter was obtained from *ASABE Standards* (2005), while the ratio of organic to mineral N (1.85) for alum-treated litter and the ratio of organic to mineral P (5.67 for untreated litter and 18 for alum-treated litter) were obtained from a previous investigation in the Lincoln Lake watershed (Chaubey et al., 2005). It should be noted that the nutrient input information was only applied on pasture lands. Loadings of sediment and nutrients from other land use areas (e.g., urban and forest) were estimated using default model parameter values.

The pasture management information, including amount of litter and fertilizer application, timing of manure and fertilizer application, grazing intensity, and dates, was obtained from a detailed review of historical nutrient management plans and interviews with 63 out of 75 farmers in the watershed (Pennington et al., 2008). The timing and amount of litter and fertilizer application varied in this watershed during 1992 to 2004. The average litter application and approximate dates of application were 2500 kg ha^{-1} applied on 30 April and 31 August. The HRUs delineated in the SWAT model were overlaid with the farm parcel GIS layer, and the portion of each farm parcel covered by different pasture HRUs was identified. The amount of litter and fertilizer application for each HRU was calculated as an area-weighted value from different portions of farm parcels for any specific application timing. The grazing dates and number of grazing days for each HRU were identified as the longest grazing period that covered all the specific dates applied in different farm parcels. The number of cattle in the watershed was obtained from the 2002 agricultural census report (USDA-NASS, 2002). The body weight, dry matter intake (% of average body weight), and total solids in the manure (% of the wet manure) for different types of cows were obtained from the ASABE standard (*ASABE Standards*, 2005). The average daily dry mass intake (kg ha^{-1} d^{-1}) for cattle in the watershed was calculated as multiplying the number of cattle by the standard body weight and the dry matter intake, and then divided by the watershed area. The dry mass of the manure (kg ha^{-1} d^{-1}) was calculated by multiplying the measured wet manure by the percentage of solids. Therefore, the daily dry mass intake was 10.14 kg ha^{-1} d^{-1} and the dry mass of the manure ranged from 0.01 to 14.2 kg ha^{-1} d^{-1} for grazing days ranging from 11 to 365 days in the watershed during 1992 to 2004.

In the SWAT management files, several operation schedules were fixed on specific dates, based on recommendations from the Washington County Cooperative Extension Service, as follows: harvest operation and then kill operation to remove fescue on 30 April, planting/beginning of growing season for bermudagrass on 1 May, harvest operation with the harvest efficiency of 0.8 on 15 July, harvest operation and then kill operation to remove bermudagrass on 15 September, and planting/beginning of growing season for fescue on 16 September with initial biomass of 3700 kg ha^{-1} (R. Morrow, personal communications).

Table 1. Content of the LUD.DAT file in the SWAT model to simulate time variant land use/land cover.

Start Date	End Date	HRU Fraction File	Year of HRU Fraction
1 Jan. 1990	31 Dec. 1993	file1.dat	1992
1 Jan. 1994	31 Dec. 1995	file2.dat	1994
1 Jan. 1996	31 Dec. 1998	file3.dat	1996
1 Jan. 1999	31 Dec. 2000	file4.dat	1999
1 Jan. 2001	31 Dec. 2003	file5.dat	2001
1 Jan. 2004	31 Dec. 2007	file6.dat	2004

The SWAT2005 model can only simulate the watershed responses at a constant land use condition. In order to incorporate the dynamic land use conditions in the watershed, the latest SWAT2009 model, which was publicly released in January 2010, was used in this study. The SWAT2009 model requires two additional input files to simulate dynamic land use conditions: (1) HRU fraction files that contain information about how HRU fraction related to different land use categories within a subbasin changes over time, and (2) the LUD.DAT file in which a user can assign the date when HRU fractions start to change. In this study, the HRU map of 2004 was used as the base HRU distribution because a greater number of urban HRUs and total HRUs were present with this dataset. If the HRU map based on 1992 land use was used as the base HRU distribution, then some HRUs, especially urban lands that existed only after 1992 would not fit to the base HRU distribution. Therefore, it is important to have a detailed HRU delineation to account for land use changes and their impacts by the SWAT model. Thus, HRU maps for the years 1992, 1994, 1996, 1999, and 2001 were matched with the HRU map for 2004 by exact subbasin-soil-land use combinations to generate the HRU fraction files. In the LUD.DAT file, the SWAT model read the 1992 HRU fraction file from 1 January 1990 to 31 December 1993, then read the 1994 HRU fraction file from 1 January 1994 to 31 December 1995, and similarly for the following years (table 1).

Calibration/Validation Methodology

The SWAT outputs of interest in this study were total sediment (TS), total nitrogen (TN), and total phosphorus (TP) losses at HRUs, subbasins, and watershed outlets at monthly and annual time steps. Model-simulated and measured values of stream flow, TS, TN, and TP were compared to evaluate the ability of the SWAT model to accurately simulate catchment responses. A warm-up period for the SWAT model is recommended to initialize and stabilize reasonable starting values for the model parameters. The warm-up period could range from two or three years depending on whether the model is representative of the watershed condition (Tobin and Bennett, 2009; White and Chaubey, 2005). In this study, the simulated period was from 1990 to 2007, where the first two years were used as the model warm-up years. A model sensitivity analysis, calibration, and validation were performed using the methods outlined by White and Chaubey (2005), Moriasi et al. (2007), and Engel et al. (2007). Sensitivity analysis is usually performed to identify which parameters in a model most influence outputs of interests. Based on the sensitivity analysis results and the identified calibration parameters in several SWAT publications, 13 parameters were modified for calibration in this study (White and Chaubey, 2005; Bracmort et al., 2006). Model calibration and validation were performed for monthly stream flow, TS, TN, and TP using the measured flow and water quality data collected at Upper Moores Creek for the periods January 1996 to February 1999, January 2000 to December 2003, and January 2006 to December 2007. Measured stream flow data were available for eight years, while water quality data (TS, TN, and TP) were only available for seven years, which was the same monitoring period as flow except the year of 1996. In order to make the model comprehensively capture the watershed responses, we selected January 2001 to December 2003 and January 2006 to December 2007 as the model calibration period due to major land use changes during this period. Subsequently, we selected January 1996 to February 1999 and January 2000 to December 2000 as the model validation period for flow and January 1997 to February 1999 and January 2000 to December 2000 for TS, TN, and TP. The variability of monthly precipitation in both calibration and validation periods were similar.

Flow was calibrated first because it can influence other outputs (White and Chaubey, 2005). Flow calibration was followed subsequently by TS, TN, and TP calibrations. Two quantitative statistics were used for model evaluation, namely Nash-Sutcliffe efficiency (NSE) (Nash and Sutcliffe, 1970) and coefficient of determination (R^2). The NSE is a normalized statistic indicating how well the observed and predicted data fit the 1:1 line (Nash and Sutcliffe, 1970). The model with an NSE value greater than 0.5 is regarded as satisfactory (Moriasi et al., 2007). The model with an R^2 value greater than 0.5 is usually considered acceptable (Van Liew et al., 2003). The process of calibration was repeated by adjusting the parameters and computing the NSE and R^2 between observed and predicted data. To test if the parameters were appropriately selected for model calibration, model validation was performed to evaluate the accuracy of the model in predicting values compared with a different set of measured data from the calibration dataset (Wilson, 2002). Model validation was performed using the optimal calibrated parameter values, and the predicted data were evaluated by calculating the values of NSE and R^2.

Evaluation of Land Use Change and Pasture Management on Water Quality

The simulated scenarios consisted of three different combinations of land use change and pasture management:

Dynamic land use, eighteen-year rotation (D18): Land use changed between 1990 and 2007 concurrently with the conservation practices in the watershed. This scenario represented the actual watershed conditions. The dynamic conservation practices were represented by using an 18-year pasture management rotation (1990-2007) in the SWAT model.

Constant land use, one-year rotation (C1): Land use and pasture management information were fixed to represent 1992 land use and management in the watershed, assuming that neither land use nor pasture management practices changed between 1990 and 2007. The static pasture management conditions were represented by using a one-year pasture management rotation in the SWAT model. This one-year pasture management rotation had 1,256 to 12,554 kg ha^{-1} of litter applied at one time or twice on 30 April, 31 August, 1 September, or 15 October, and grazing manure ranged from 1.92 to 5.85 kg ha^{-1} for 90 to 365 days. The model predictions for this scenario represented the watershed response if neither land use nor pasture management had changed in the watershed between 1992 and 2007.

Dynamic land use, one-year rotation (D1): Land use changed between 1990 and 2007, but conservation practices remained the same as in 1992. The land use was changed using HRU fractions described above; however, in this scenario, it was assumed that pasture management did not change over time. The model predictions for this scenario represented the impact of land use change alone if no pasture management practices were implemented in the watershed.

The cumulative impacts of land use change and pasture management on water quality in the Beatty Branch, Upper Moores Creek, and Moores Creek subwatersheds during the period 1992 to 2007 were calculated using a pairwise comparison of pollutant losses at these subwatersheds under three land use change and pasture management scenarios. Two pairs of scenario comparisons were D18 and D1, and D1 and C1. The cumulative pollutant losses for each scenario were first produced by summing the annual pollutant loss increments between 1992 and 2007. The difference between the first pair (D18 minus D1) denotes the cumulative impacts of pasture management on water quality. Similarly, the difference between the second pair (D1 minus C1) denotes the cumulative impacts of land use changes on water quality. The cumulative combined impacts of land use changes and pasture management on water quality were calculated as D18 minus C1. The positive value of the difference indicates a degradation of water quality (a net increase in pollutant losses), while the negative value indicates an improvement of water quality (a net decrease in pollutant losses).

In order to understand whether area-averaged pollutant losses from pasture lands were different from the pollutant losses from the entire subwatershed, a matched-pair comparison for years during 1992 to 2007 was performed for each land use change and pasture management scenario. The null hypothesis was that area-averaged pollutant losses from pasture lands and from the entire subwatershed were equal. In comparing the performances of the two groups, the 0.1 level of significance (α) was used.

Results and Discussion

Land Use Changes in the Watershed

Noticeable land use changes in the Lincoln Lake watershed have occurred since monitoring started in 1992 (fig. 2). However, the land use changes have not been consistent in the three subwatersheds. Land use distributions in each subwatershed (Beatty Branch, Upper Moores Creek, and Moores Creek) of the Lincoln Lake watershed from 1992 to 2004 are shown in table 2. Pasture lands were dominant in 1992 in all three subwatersheds. Pasture areas decreased from 49% to 35% in the Beatty Branch subwatershed, from 59% to 45% in the Upper Moores Creek subwatershed, and from 51% to 40% in the Moores Creek subwatershed during 1992 to 2004, respectively. Concurrently, urban lands increased during the same period. A relatively greater proportion of urban lands was concentrated in the Upper Moores Creek subwatershed, where it has expanded from 8% to 22% of the entire subwatershed area. Urban lands have increased

Table 2. Historical land use distribution in the Beatty Branch, Upper Moores Creek, and Moores Creek subwatersheds from 1992 to 2004. Land use area is in hectares, and numbers in the parentheses are the percentage of land use in the subwatershed.

Subwatershed	Land Use	1992	1994	1996	1999	2001	2004
Beatty Branch	Forest	525 (49.3)	576 (54.1)	574 (53.9)	557 (52.3)	585 (54.9)	589 (55.3)
	Pasture	525 (49.3)	461 (43.3)	443 (41.6)	445 (41.8)	401 (37.7)	374 (35.1)
	Urban	10 (0.9)	24 (2.2)	43 (4)	58 (5.4)	73 (6.8)	94 (8.8)
	Other	5 (0.5)	4 (0.4)	5 (0.5)	5 (0.5)	6 (0.6)	8 (0.8)
	Total	1065	1065	1065	1065	1065	1065
Upper Moores Creek	Forest	410 (31.7)	461 (35.7)	425 (32.9)	462 (35.8)	452 (35)	414 (32)
	Pasture	761 (58.9)	698 (54)	663 (51.3)	603 (46.7)	569 (44)	574 (44.5)
	Urban	107 (8.3)	124 (9.6)	187 (14.5)	215 (16.6)	262 (20.3)	286 (22.1)
	Other	14 (1.1)	9 (0.7)	17 (1.3)	12 (0.9)	9 (0.7)	18 (1.4)
	Total	1292	1292	1292	1292	1292	1292
Moores Creek	Forest	802 (41.8)	871 (45.4)	831 (43.3)	900 (46.9)	861 (44.9)	808 (42.1)
	Pasture	978 (51)	895 (46.7)	852 (44.4)	758 (39.5)	741 (38.6)	757 (39.5)
	Urban	122 (6.4)	142 (7.4)	215 (11.2)	245 (12.8)	304 (15.8)	331 (17.3)
	Other	16 (0.9)	10 (0.5)	20 (1.1)	15 (0.8)	12 (0.6)	22 (1.1)
	Total	1918	1918	1918	1918	1918	1918

by nine times in the Beatty Branch subwatershed, three times in the Upper Moores Creek subwatershed, and six times in the Moores Creek subwatershed. Forest lands remained relatively constant during 1992 to 2004, which were 49% to 55%, 32% to 36%, and 42% to 47% in the Beatty Branch, Upper Moores Creek, and Moores Creek subwatersheds, respectively. Gitau et al. (2010) concluded that there was a systematic conversion of pastures to urban areas in the watershed during the study period. About 90% of forest remained unchanged during the study period, which was attributed to the location of forest around the watershed outlet, whereas most of the land use changes happened in the headwater areas.

Pasture Management Changes in the Watershed

Land application of fertilizer/animal manure and grazing management are the only two pasture management practices in the watershed. The fertilizer and manure from confined animal systems were regarded as land application, while the manure directly excreted from cattle was regarded as grazing management in this study. The amounts of TN and TP inputs from land management decreased as pasture lands decreased during 1992 to 2004 (table 3). The decrease in pasture lands in these subwatersheds is noticeable, as pasture lands decreased from 525 ha to 374 ha, from 761 ha to 574 ha, and from 978 ha to 757 ha in the Beatty Branch, Upper Moores Creek, and Moores Creek subwatersheds, respectively. Total nitrogen (TN) and total phosphorus (TP) inputs from land application have declined by 40 kg ha^{-1} of TN, 52 kg ha^{-1} of TP in the Upper Moores Creek subwatershed from 1992 to 2004. Similarly, in the Moores Creek subwatershed, 41 kg ha^{-1} of TN and 51 kg ha^{-1} of TP decreased as a result of loss in pasture area and reduction of land application between 1992 and 2004. However, an increase in nutrient inputs from land application was found in the Beatty Branch subwatershed, where TN input increased by 185 kg ha^{-1} and TP input slightly increased by 42 kg ha^{-1} during 1992-2004. In 2004, the TN and TP inputs from land application in the Beatty Branch subwatershed (332 kg ha^{-1} for TN and 134 kg ha^{-1} for TP) were two times greater than those in the Moores Creek subwatershed (141 kg ha^{-1} for TN and 63 kg ha^{-1} for TP), indicating that a relatively greater amount of nutrients was applied in the Beatty Branch subwatershed. Broiler-fresh manure was the most common manure used in the watershed, and the application rate of broiler manure has decreased since 1992. Besides a decrease in usage of broiler-fresh manure, more alum-treated broiler manure was applied by farmers since 2000, when farmers were encouraged to develop a nutrient management plan based on the Arkansas P-Index (DeLaune et al., 2004). The amount of TN inputs from land application has varied spatially during 1992 to 2004 (fig. 4). TN inputs in the western portion of the Beatty Branch subwatershed increased since 1999, while TN inputs in the southern portion of the Moores Creek subwatershed decreased. Similarly, land application of TP changed in the southern part of the Moores Creek subwatershed, where TP inputs were greater than 109 kg ha^{-1} before 1999 but decreased substantially after 1999. These changes indicated that farmers reduced the use

Table 3. Annual nutrient inputs (kg ha^{-1}) from land application and grazing management on pasture lands at the Beatty Branch, Upper Moores Creek, and Moores Creek subwatersheds during the period 1992-2004.

Year	Pasture Area (ha)	Land Application TN (kg ha^{-1})	Land Application TP (kg ha^{-1})	Grazing Management TN (kg ha^{-1})	Grazing Management TP (kg ha^{-1})
Beatty Branch					
1992	524	146.9	92.4	23.5	6.5
1994	461	137.1	86.1	21.9	6.1
1996	443	146.0	91.9	23.5	6.5
1999	445	175.1	88.5	17.3	4.7
2001	401	240.1	106.0	21.2	5.7
2004	374	331.6	134.0	20.6	5.6
Upper Moores Creek					
1992	761	173.9	109.3	20.1	5.5
1994	698	160.3	100.6	18.6	5.2
1996	663	172.9	108.6	24.0	6.6
1999	603	132.7	64.7	30.3	8.3
2001	569	141.8	70.3	32.7	9.0
2004	574	133.6	57.7	33.6	9.2
Moores Creek					
1992	978	181.5	114.1	20.2	5.5
1994	895	168.8	106.0	18.8	5.1
1996	852	177.1	111.4	24.3	6.7
1999	758	138.9	69.0	31.7	8.7
2001	741	144.1	72.5	34.7	9.6
2004	757	140.6	63.1	36.3	10.0

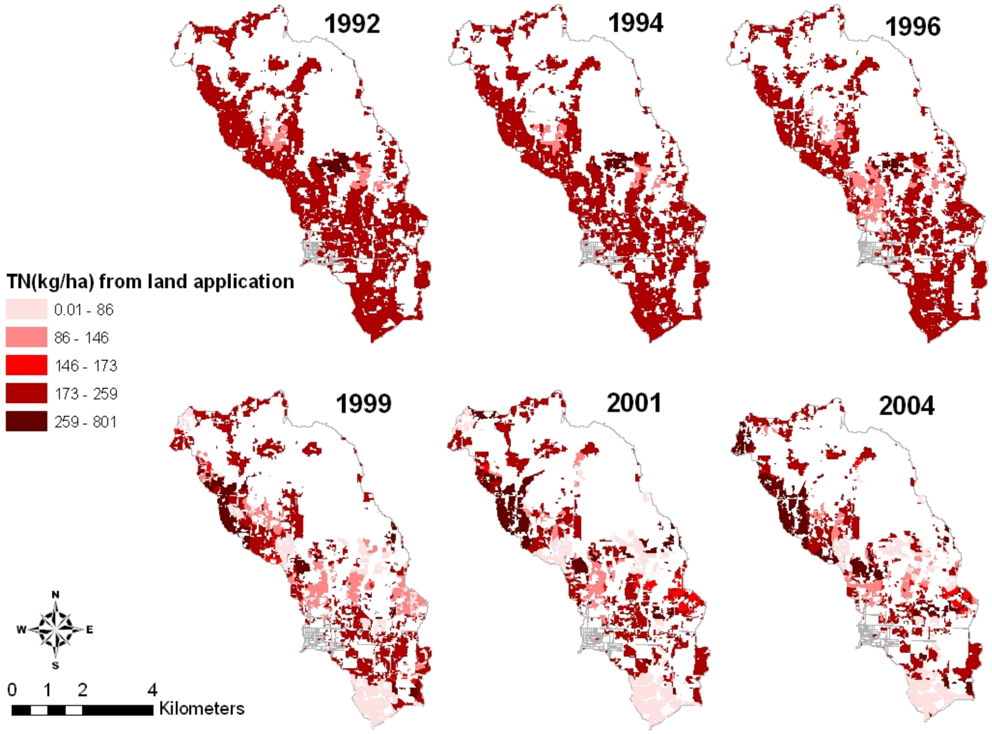


Figure 4. Annual total nitrogen (TN) inputs from land application on pasture lands during period 1992-2004.

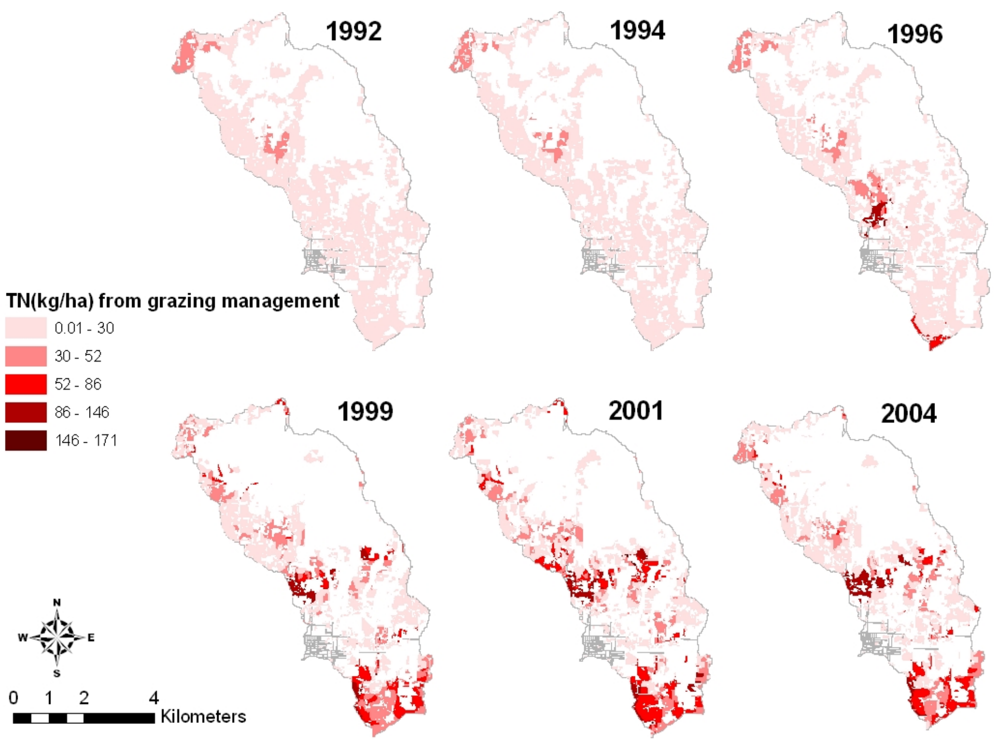


Figure 5. Annual total nitrogen (TN) inputs from grazing management on pasture lands during period 1992-2004.

of poultry manure and relied more on inorganic fertilizer to meet the N demand of forage production.

Unlike land application, nutrient inputs from grazing management increased in the Upper Moores Creek subwatershed (20 to 34 kg ha^{-1} for TN and 6 to 9 kg ha^{-1} for TP) and the Moores Creek subwatershed (20 to 36 kg ha^{-1} for TN and 6 to 10 kg ha^{-1} for TP), while the nutrient inputs from grazing management in the Beatty Branch subwatershed remained relatively constant at 17 to 24 kg ha^{-1} for TN and 5 to 6 kg ha^{-1} for TP (table 3). The TN inputs from grazing management was different among the subwatersheds during 1992 to 2007 (fig. 5) Compared to the nutrient inputs in the Moores Creek subwatershed, nutrient inputs in the Beatty Branch subwatershed were relatively low during the study period. Nutrient

Table 4. Ranges, default values, and calibrated values for calibration of SWAT parameters on flow, total sediment (TS), total nitrogen (TN), and total phosphorus (TP).

Output	SWAT Parameter[a] (unit)	Input File	Range	Default	Calibrated
Flow	CN	*.mgt	39 to 98 (-50% to 50%)	75.25	67.725 (-10%)
	ESCO	*.hru	0 to 1	0.95	0.26
	GWQMn (mm)	*.gw	0 to 5000	0	3000
TS	SLOPE (m m^{-1})	*.hru	0 to 0.6 (-50% to 50%)	0.072	0.036 (-50%)
	USLE_K	*.sol	0.01 to 0.65 (-50% to 50%)	0.345	0.1725 (-50%)
	ADJ_PKR	*.bsn		1	2
TN	NPERCO	*.bsn	0.001 to 1	0.2	1
	CMN	*.bsn	0.001 to 0.003	0.003	0.004
TP	PPERCO	*.bsn	10 to 17.5	10	17.5
	PHOSKD	*.bsn	40 to 300	175	100

[a] CN = curve number, ESCO = soil evaporation compensation factor, GWQMn = minimum threshold depth of water in the shallow aquifer for return flow to occur, SLOPE = average slope steepness, USLE_K = USLE soil erodibility factor, ADJ_PKR = peak rate adjustment factor for sediment routing in the main channel, NPERCO = nitrate percolation coefficient, CMN = sediment concentration in lateral flow, PPERCO = phosphorus percolation coefficient, and PHOSKD = phosphorus soil partitioning coefficient.

Table 5. Results of calibration and validation of the SWAT model for average monthly flow, total sediment (TS), total nitrogen (TN), and total phosphorus (TP).

Variable	Calibration NSE	Calibration R^2	Validation NSE	Validation R^2
Flow (m^3 s^{-1})	0.52	0.55	0.6	0.76
TS (kg ha^{-1})	0.58	0.73	0.25	0.67
TN (kg ha^{-1})	0.5	0.66	0.33	0.5
TP (kg ha^{-1})	0.6	0.72	0.73	0.89

inputs increased since 1996 at the northern and southern portions of the Moores Creek subwatershed, indicating that there would be higher stocking rates and intensive management systems (short-term rotations) on those pasture lands (Berry et al., 2002). The maximum daily manure inputs from grazing animals in the Lincoln Lake watershed were 8, 8, 10.1, 13.1, 13.3, and 14.2 kg ha^{-1} d^{-1} in 1992, 1994, 1996, 1999, 2001, and 2004, respectively. It should be noted that the

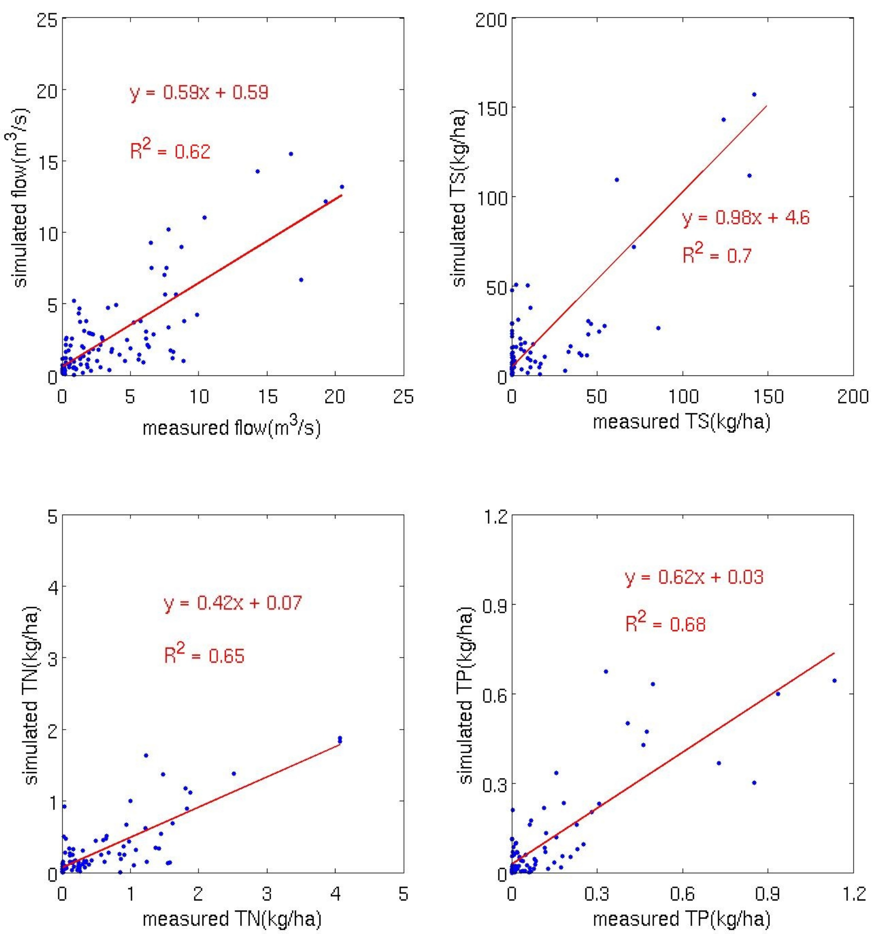


Figure 6. Comparison of measured and simulated flow, total sediment (TS), total nitrogen (TN) and total phosphorus (TP) losses in the Upper Moores Creek subwatershed for the D18 scenario.

nutrient inputs from grazing animals constituted a relatively minor portion of the total nutrient inputs in the watershed.

Performance of the SWAT Model

The parameter default values, their ranges, and calibrated values of the SWAT model are shown in table 4. The model performance during the calibration and validation periods is shown in table 5. For calibration, the NSE and R^2 values for flow, TS, TN, and TP were equal to or greater than 0.5, which is generally viewed as a satisfactory model performance. For validation, the performance of the model in simulating flow and TP was satisfactory in terms of NSE and R^2 values, which were both greater than 0.5. Aside from one indication of unsatisfactory model performances (NSE = 0.25 and 0.33 for TS and TN, respectively), the model simulation of TS and TN was satisfactory, as indicated by R^2 values greater than 0.5. We found that there were some extremely high measured TS and TN values that the model could not capture, which was the reason why the NSE values for TS and TN were lower than 0.5. Therefore, the NSE values increased to 0.64 and 0.53 for TS and TN after these outliers (a total of four data points) were removed. Concurrently, the R^2 values for TS and TN increased to 0.75 and 0.73, respectively. The SWAT-simulated flow, TS, TN, and TP losses at the Upper Moores Creek were similar to the measured data during 1996 to 2007 (fig. 6). Overall, this calibrated SWAT model incorporated with detailed dynamic land use and pasture management information (scenario D18) could be used to simulate land use change and pasture management scenarios (C1 and D1).

Sediment and Nutrient Losses for Different Scenarios

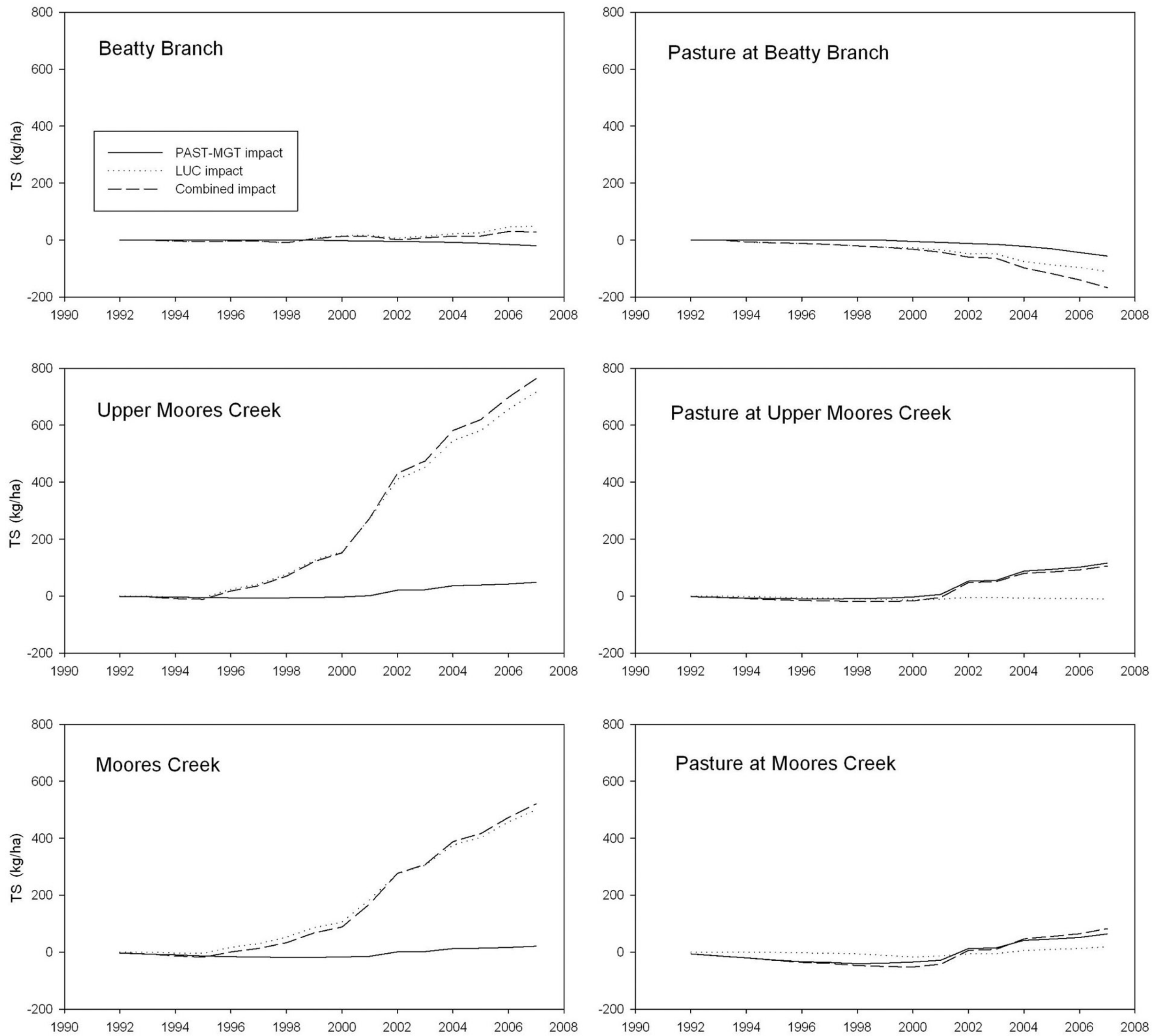
Losses of TS, TN, and TP from pasture areas compared to the entire watershed under different land use change and pasture management are shown in table 6. The annual TS losses (46 to 312.9 kg ha^{-1}) from the entire watershed were greater than TS losses (22.4 to 255.2 kg ha^{-1}) from the pasture lands. Similarly, TS losses (23.2 to 255.2 kg ha^{-1}) from the pasture lands for the 18-year rotation scenario (D18) were greater than TS losses (22.4 to 242.1 kg ha^{-1}) for the one-year rotation scenarios. TN losses (2.3 to 7.6 kg ha^{-1}) from the pasture lands for dynamic land use and pasture management (D18) scenario were greater than TN losses (1.5 to 5.8 kg ha^{-1}) from the entire watershed. TN losses from pasture lands of Beatty Branch subwatershed were significantly ($p < 0.001$) greater than those from the Upper Moores Creek and Moores Creek subwatersheds, which indicated that nutrient input amount had great impacts on nutrient losses from the subwatershed. Similar results were evident for TP, where losses from pasture lands ranged from 0.7 to 4.1 kg ha^{-1}, whereas TP losses for the entire watershed ranged from 0.3 to 2.1 kg ha^{-1}. For all land use change and pasture management scenarios, TN and TP losses from pasture lands were greater than the average nutrient losses from the entire watershed. The results showed that pasture lands were the main source of contributing nutrient losses from the watershed.

Table 6. Pollutant losses (TS, TN, and TP) from Lincoln Lake watershed and pasture lands for different land use change and pasture management scenarios (C1 = constant land use and one-year rotation, D1 = dynamic land use and one-year rotation, and D18 = dynamic land use and 18-year rotation.)

	From Entire Watershed (kg ha^{-1})			From Pasture Lands (kg ha^{-1})		
Year	C1	D1	D18	C1	D1	D18
Total sediment (TS)						
1992	156.9	156.9	155.1	124.5	124.5	120.6
1993	202.4	202.4	200.1	210.5	210.5	205.5
1994	124.9	120.2	118.5	110.9	105.9	101.8
1995	138.8	136.4	134.3	108.1	104.3	99.3
1996	137.0	150.4	148.9	98.1	96.6	92.8
1997	102.9	110.9	110.3	87.0	85.2	83.7
1998	139.8	151.2	150.0	162.1	159.0	155.9
1999	155.2	180.0	180.4	105.0	103.5	104.6
2000	118.2	133.2	133.6	89.0	86.6	87.5
2001	181.2	228.1	229.2	104.9	105.3	108.3
2002	252.2	302.1	310.8	228.9	229.9	255.2
2003	46.0	66.4	66.6	22.4	22.7	23.2
2004	264.3	307.8	312.9	242.1	236.0	250.9
2005	86.1	101.9	102.0	89.7	87.1	87.5
2006	106.9	146.8	146.7	78.2	76.2	75.9
2007	160.5	186.0	187.5	125.5	122.1	126.5
Total nitrogen (TN)						
1992	3.1	3.1	3.2	4.0	4.0	4.1
1993	4.1	4.1	4.2	5.2	5.2	5.4
1994	2.6	2.5	2.6	3.2	3.1	3.3
1995	2.7	2.7	2.7	2.9	2.9	3.0
1996	3.6	3.7	3.7	4.0	4.0	4.1
1997	3.3	3.4	3.5	4.5	4.5	4.7
1998	4.0	4.1	4.2	4.9	4.9	5.1
1999	3.4	3.6	3.7	3.9	4.0	4.3
2000	3.4	3.6	3.8	4.5	4.5	5.2
2001	3.5	3.8	3.9	4.4	4.4	4.6
2002	3.3	3.5	3.8	4.4	4.4	5.2
2003	1.2	1.4	1.5	1.9	1.9	2.3
2004	5.1	5.5	5.8	6.5	6.5	7.6
2005	2.9	3.2	3.4	3.7	3.7	4.6
2006	3.2	3.6	3.9	3.7	3.7	4.9
2007	4.6	4.9	5.4	5.4	5.4	7.2
Total phosphorus (TP)						
1992	0.8	0.8	0.8	1.6	1.6	1.6
1993	1.1	1.1	1.2	2.3	2.3	2.4
1994	0.6	0.6	0.6	1.2	1.2	1.3
1995	0.9	0.8	0.9	1.8	1.7	1.8
1996	1.1	1.0	1.0	2.1	2.1	2.2
1997	0.6	0.6	0.6	1.2	1.2	1.3
1998	1.3	1.1	1.2	2.4	2.4	2.5
1999	1.2	1.1	1.1	2.3	2.3	2.3
2000	1.0	0.9	0.9	2.0	2.0	2.0
2001	1.3	1.1	1.1	2.4	2.4	2.4
2002	1.4	1.2	1.3	2.7	2.7	2.9
2003	0.3	0.3	0.3	0.7	0.7	0.7
2004	2.1	1.8	1.8	4.1	4.0	4.1
2005	1.0	0.8	0.8	1.9	1.9	1.9
2006	1.2	1.1	1.1	2.3	2.3	2.3
2007	1.6	1.3	1.3	3.1	3.0	3.0

Impacts of Land Use Change and Pasture Management on Water Quality in the Subwatersheds

The cumulative impacts of land use change alone, pasture management alone, and the combined impacts of land use and pasture management changes on TS, TN, and TP losses from pasture lands of the Beatty Branch, Upper Moores Creek, and Moores Creek subwatersheds and entire subwatersheds are shown in figure 7. The cumulative impacts of land use change and pasture management on area-averaged

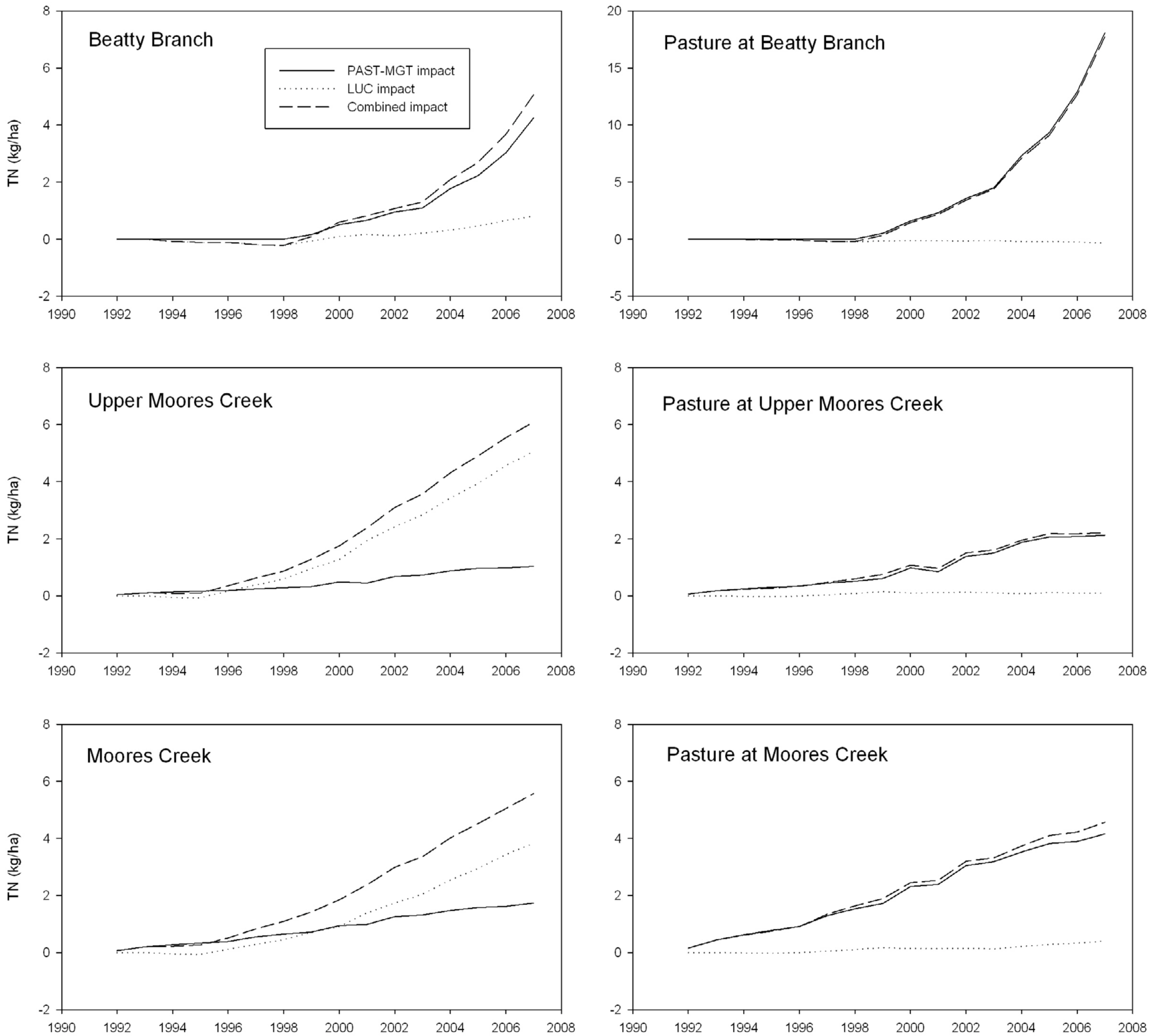


(a) Total sediment

TS losses in the Beatty Branch subwatershed were similar and relatively small (48 kg ha^{-1} and -20 kg ha^{-1} for the cumulative impacts of land use change and pasture management on TS losses in 2007, respectively), while the cumulative impact of land use change in the Upper Moores Creek (716 kg ha^{-1}) and Moores Creek (499 kg ha^{-1}) subwatersheds were much greater than the impact of pasture management (48 kg ha^{-1} and 21 kg ha^{-1} for the Upper Moores Creek and Moores Creek subwatersheds, respectively) (fig. 7a). Since pasture management practices applied in this study were mainly nutrient management and grazing management, they had relatively small impacts on TS losses in the subwatersheds. The similar small values of land use change and pasture management impacts in the Beatty Branch subwatershed could be explained by relatively smaller increase in the urban areas in the subwatershed during the study period 1992 to 2007, as urban area can be the main source of TS losses. The positive values of land use change impacts on TS losses in the Upper Moores Creek and Moores Creek subwatersheds indicated that an increase in TS losses was mainly due to land use changes in these subwatersheds.

The cumulative impact of pasture management on TN losses slightly increased through the period 1992 to 2007 (1 to 4.3 kg ha^{-1} in 2007), especially in the Beatty Branch subwatershed, indicating that an increase in TN inputs on the pasture lands resulted in increased TN losses (fig. 7b). Similar to TS losses in the Upper Moores Creek and Moores Creek subwatersheds, the cumulative impact of land use change on TN losses (3.8 to 5.1 kg ha^{-1}) was greater than the impact of pasture management (1 to 1.7 kg ha^{-1}), indicating that land use changes contributed greater TN losses than pasture management, while the impact of pasture management (4.3 kg ha^{-1}) was greater than land use change impacts (0.8 kg ha^{-1}) in the Beatty Branch subwatershed. Overall, when the impacts of land use change and pasture management were combined, they led to water quality degradation in terms of greater TS and TN losses in the subwatersheds (28 to 764 kg ha^{-1} and 5.1 to 6.1 kg ha^{-1} of combined impacts on TS and TN losses in 2007, respectively).

Unlike the impact of land use change on TS and TN losses, the negative values of the cumulative impact of land use change showed that TP losses for the constant land use scenario were greater than for the dynamic land use scenario in



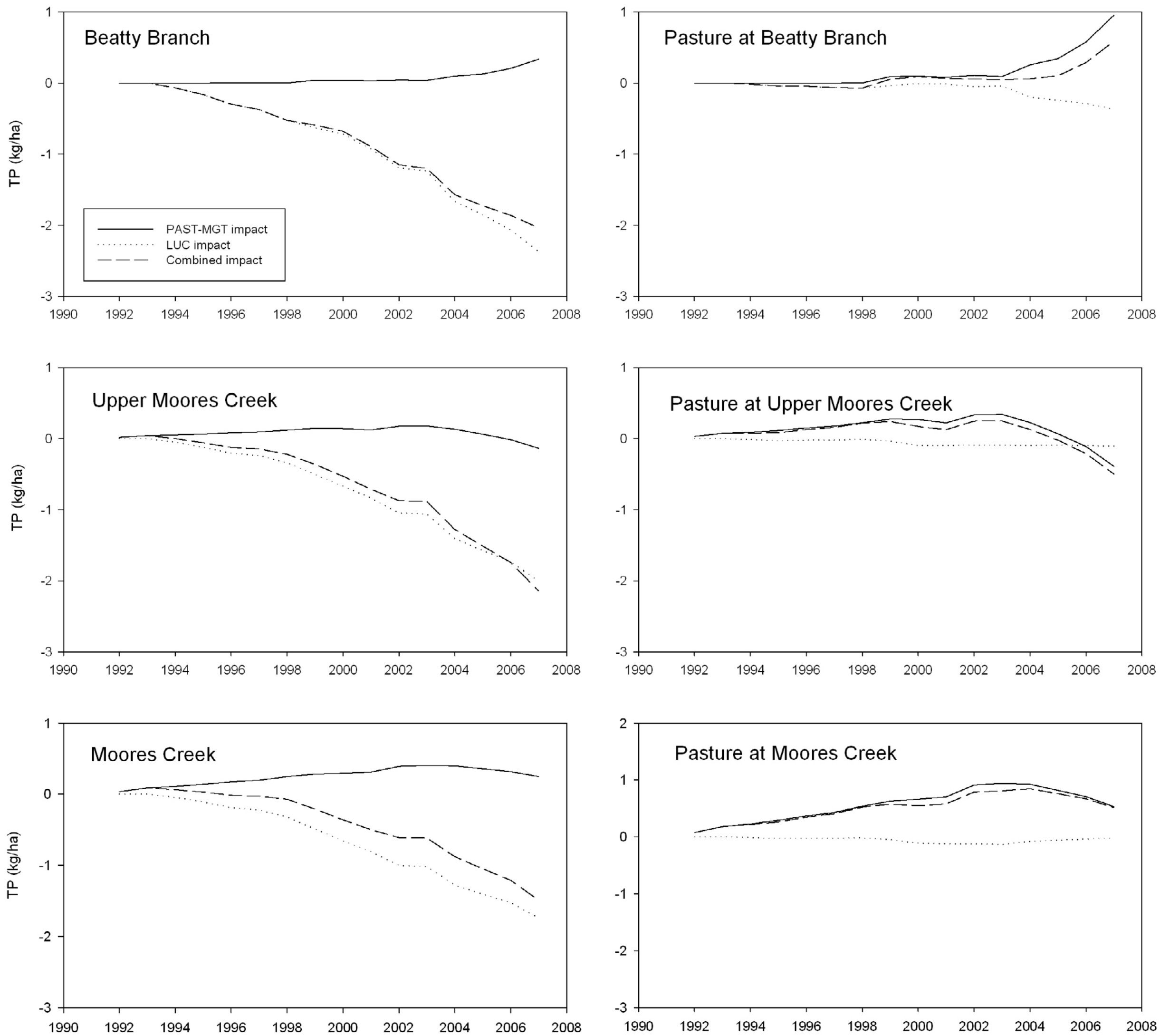
(b) Total Nitrogen

all subwatersheds (fig. 7c). This positive land use change impact on TP losses from the entire watershed could be due to gradually decreasing pasture areas in the watershed since 1992, with pasture lands being the main source of TP losses (table 6). It should be noticed that area-averaged TP losses from pasture lands were similarly high for both constant and dynamic land use scenarios when they were compared with TP losses from the entire watershed. Contrarily, pasture management had little impact on TP losses from pasture lands for each subwatershed. Pasture management had negative impact on TP losses from Beatty Branch subwatershed as a result of TP inputs increased during 1992-2004, while pasture management had reduced TP losses from Upper Moores Creek and Moores Creek due to decreased usage of land application (table 3). The impact of land use change was superior to pasture management on TP losses; therefore, the combined impact of land use changes and pasture management resulted in a decrease in TP losses (1.5 to 2.1 kg ha^{-1} in 2007) and improvement in water quality.

Impacts of Land Use Change and Pasture Management on Water Quality at the Subbasins

The cumulative impacts of land use change, pasture management, and combined impacts on TS, TN, and TP losses of the SWAT simulation results in 2007 are spatially different at the subbasin level (fig. 8). Darker colors indicate a relatively greater increase in pollutant losses and imply a higher level of impacts. For TS losses, land use changes resulted in less TS losses in the Beatty Branch and Lower Moores Creek subwatersheds and greater TS losses in the western part and northeastern part of the Upper Moores Creek subwatershed, where most of urban land use changes occurred. Pasture management resulted in a decrease in TS losses for half of the entire Lincoln Lake watershed. The combined impact in the western part and northeastern part of the Upper Moores Creek subwatershed indicated that land use changes resulted in greater TS losses and masked the improvement due to pasture management, which resulted in less TS losses.

TN losses increased in the western part of the Upper Moores Creek subwatershed, where urban lands were primarily located. The western part of the Beatty Branch subwatershed had an increase in TN losses as the result of an increase in nutrient inputs, while the southern part of the Up-



(c) Total Phosphorus

Figure 7. Cumulated impacts of land use change (LUC) and pasture management (PAST-MGT), and the combined impacts on annual (a) total sediment, (b) total nitrogen, and (c) total phosphorus losses from the Beatty Branch, Upper Moores Creek, and Moores Creek subwatersheds during 1992-2007. Positive values indicate an increase and negative values indicate a decrease in pollutant losses.

per Moores Creek subwatershed had a decrease in TN losses as the result of reduced nutrient inputs (table 3). When the impacts of land use changes and pasture management were combined, TN losses increased in most of the western part of the Lincoln Lake watershed. The TN losses increased due to land use changes in the Upper Moores Creek. However, pasture management was the dominant factor resulting in TN loss increase in the Lower Moores Creek and the Beatty Branch subwatersheds.

Losses of TP decreased during the study period in all three subwatersheds (fig. 7c). Different impacts of pasture management in the Beatty Branch and Upper Moores Creek subwatershed were mainly due to different trends of land application in the subwatersheds (table 3). TP losses decreased as a result of the reduction in nutrient inputs in the Upper Moore Creek subwatershed, while TP losses increased in the western part of the Beatty Branch subwatershed were mainly due to increasing land application during 1992-2004. The impact of land use changes on TP losses was greater than the impact of pasture management when both impacts are added together. For example, a decrease in pasture lands resulted in reduction of TP losses (i.e., land use change impact) in the western part of the Beatty Branch subwatershed, and an increase in land application resulted in increasing TP losses (i.e., pasture management change impact). However, TP losses decreased when the two impacts were combined. Similar results were observed in the western part of the Upper Moores Creek subwatershed, where TP losses increased as a result of combined impacts and the negative impact of land use was superior to the positive impact of pasture management. Overall, TP losses from the entire watershed decreased as a result of the combined impacts of land use and pasture management changes. These results show that the impacts of land use changes must be evaluated in conjunction with the conservation practices impacts in agricultural watersheds, as the land use changes alone can mask the impacts of conservation practices.

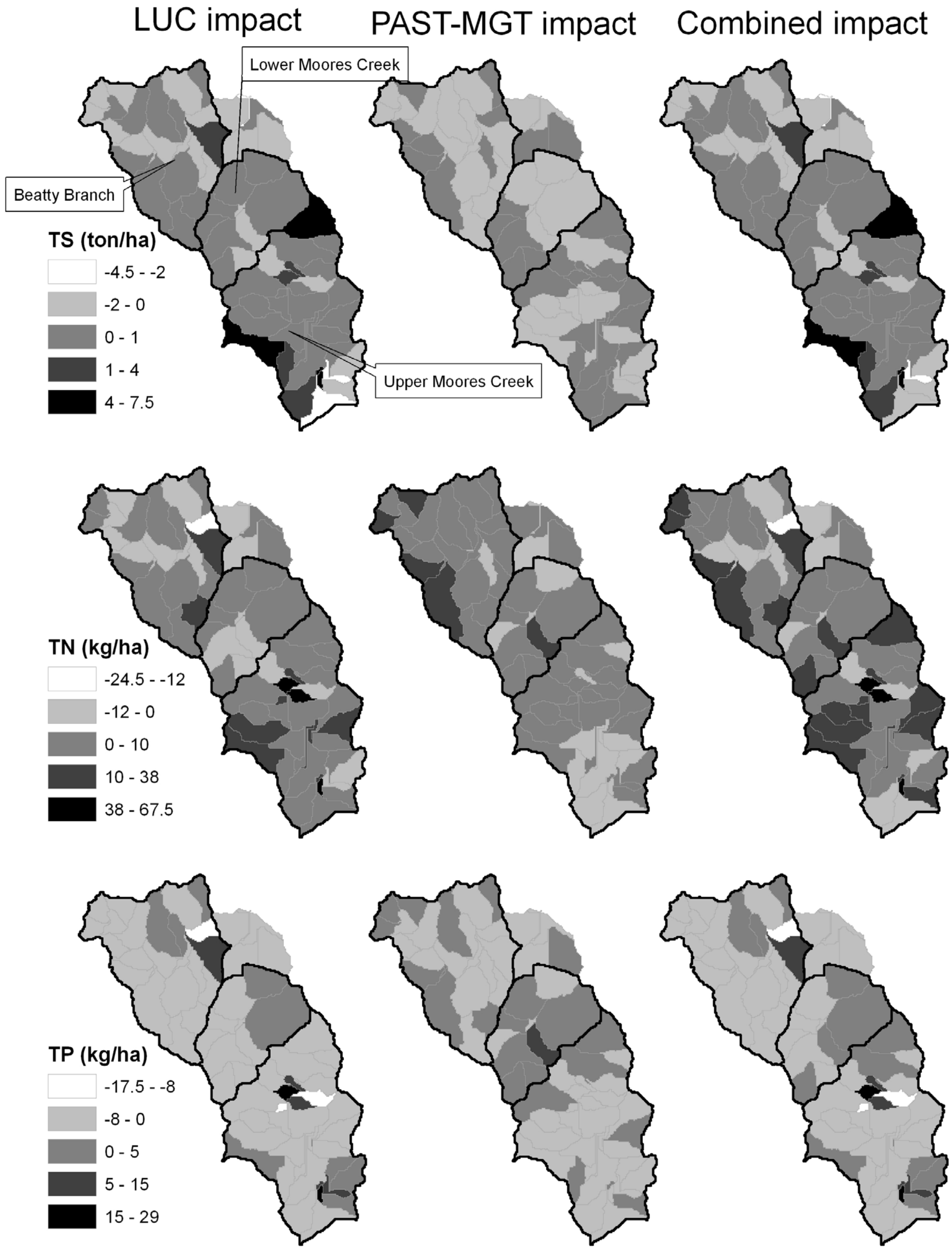


Figure 8. Distribution of cumulative land use change, pasture management, and combined impacts on pollutant losses at the subbasin level of the Lincoln Lake watershed in 2007.

Summary and Conclusions

The watershed responses at the Beatty Branch, Upper Moores Creek, and Moores Creek subwatersheds for different land use change and pasture management scenarios were simulated using the SWAT model, and the pollutant losses were compared at different spatial scales. The first objective of this study was to evaluate the linkage between nutrient inputs from various pasture management practices and water quality. Land application and grazing management were the main pasture management practices applied in the watershed. Historical pasture management showed that nutrient inputs from land application increased in the Beatty Branch subwatershed, while nutrient inputs decreased in the Moores Creek subwatershed due to a reduction in land application rates during 1992-2007. However, the nutrient inputs from grazing management increased slightly in the Moores Creek subwatershed due to more intensive grazing. An increased amount of nutrient application in pasture areas resulted in increased losses of nutrients in the Beatty Branch subwatershed. The greater TS, TN, and TP losses from pasture lands indicated that pasture lands were the main source of NPS pollution in the Beatty Branch subwatershed. However, urban areas were the main source of TS and pasture areas were the main source of TN and TP losses in the Moores Creek subwatershed.

The second objective of this study was to differentiate the impacts of land use changes and pasture management on water quality. The results showed that land use changes resulted in greater TS losses in the Upper Moores Creek and Moores Creek subwatersheds, indicating that the increase in TS losses was mainly due to the large portion of urbanizing area during the study period. Pasture management had a dominant impact on TN losses in the Beatty Branch subwatershed, where increased TN losses were observed due to an increase in nutrient inputs on the pasture lands. However, land use changes resulted in reduced TP losses in all subwatersheds. At a smaller spatial scale (the subbasin level used in the SWAT simulation), the individual impacts of land use changes and pasture management at each subbasin were identified. The subbasins where major urban lands were located in the western part of the Upper Moores Creek subwatershed had greater losses of TS and TN due to land use changes, indicating that an increase in urban lands could result in water quality degradation. Similarly, some subbasins in the western part of the Beatty Branch subwatershed had greater TN and TP losses from the impacts of pasture management due to an increase in nutrient inputs in the subwatershed. Overall, TS and TN losses increased and TP losses decreased as the result of combined impacts of land use and pasture management changes. The results of this study indicate a need for differentiating the impacts of land use changes from the impacts of conservation practices in order for a true picture of the conservation effectiveness to be realized. Without such a clear differentiation, it is possible that the negative impacts of land use changes can mask the positive impacts of conservation practices in the watershed.

Acknowledgements

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