DEVELOPING CONTAMINANT LOADS FOR GEORGIA STREAMS

by

JAMES K. BRADSHAW

(Under the direction of David E. Radcliffe)

ABSTRACT

Comparing contaminant loads from streams in single-land use watersheds to appropriate reference streams and modeling water quality with a watershed-scale GIS model may be an effective method for determining background contaminant loads. Water samples taken during storm and base flow were used to calculate sediment and phosphorus loads in four small single-land use watersheds and the Soil Water Assessment Tool (SWAT) was calibrated for sediment and phosphorus from agricultural watersheds. Sediment loads ranged from 241 to 1725 kg $ha^{-1}yr^{-1}$ and total P loads ranged from 0.145 to 1.87 kg $ha^{-1}yr^{-1}$. Error between predicted and observed sediment loads ranged from 7-367% and error between predicted and observed TP loads ranged from 47-70%. The results of this study indicate that better best management practices (BMP's) need to be implemented in agricultural watersheds to control phosphorus and that SWAT may not be an appropriate model to aid in TMDL development for small streams.

INDEX WORDS: Suspended sediment concentration, Dissolved reactive phosphorus, Total phosphorus, Single-land use, SWAT, GIS, Stream water quality, TMDL, Water quality modeling

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TABLE OF CONTENTS

Page

LIST OF TABLES

Page

LIST OF FIGURES

Page

CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Introduction

Total Maximum Daily Loads

The 1972 Clean Water Act initiated the Total Maximum Daily Load (TMDL) program to aid states in meeting water quality standards. A TMDL is the maximum amount of a particular pollutant that can be allowed into a stream in order for it to meet its assigned designated use. For TMDLs to be developed, the USEPA requires that sources of contaminants be identified. Streams that have been identified as not meeting their designated use are listed on the State's 303(d) list. Once a stream is listed, a plan must be developed to set a TMDL for that particular stream (USEPA, 2006).

According to the USEPA, 15% of our nations watersheds have relatively good water quality, 36% are moderately impaired, 22% are seriously impaired, and 27% do not have enough information to be classified (USEPA 2002). The USEPA Clean Water Action Plan of 1998 states that polluted runoff, specifically from agricultural non-point sources, is the most important source of water contamination preventing the attainment of water quality goals identified by the Clean Water Act (USEPA, 1998a). Sediment and nutrients are two main contributors of poor water quality.

Sediment

Suspended sediment concentration (SSC) is a measure of the amount of total suspended solids (TSS) collected in a stream sample. Sediment concentrations in surface water should be as low as possible. High SSC in a stream can adversely affect the biota of a stream by any one or a combination of the following: by reducing photosynthesis, blocking sunlight penetration, abrading fish gills, reducing feeding efficiency for filter feeders, or reducing visibility for sight feeders. Of all the waters listed on the 303(d) list nationally, 17% are associated with sediment (Sediment TMDL Technical Advisory Group, 2002). Because of the adverse effect of sediment on biota, an Index of Biological Integrity (IBI) can assess the biological integrity of an aquatic ecosystem. Impaired streams usually have a low IBI score and high amounts of chemicals, which are often associated with excess sediment. Sediment also causes adverse physical effects such as lost reservoir storage, degraded recreational water uses, and increased cost of water purification (Sediment TMDL Technical Advisory Group, 2002). Data show that sediment concentrations vary with differing land uses (Bradshaw et al., 2005).

Phosphorus

Nutrients in surface water represent a group of contaminants that affect both aquatic and human health. Nitrogen and phosphorus are two major nutrients that adversely affect water quality. Phosphorus in particular, is an essential component of nucleic acids and metabolites. Generally, N and P are limiting factors for growth in aquatic plants. However, excessive amounts of these two nutrients can cause accelerated growth in aquatic plants eventually leading to eutrophication. This process leads to oxygen depletion and can cause fish kills (Cooper, 1993). Since most phosphates bind to soils and sediments, P in surface waters comes from surface flows and not groundwater, except for extreme situations where soils are water-logged and anoxic (Correll, 1998). Concentrations of P in surface water are linked to land use, making it important to develop TMDL standards to regulate nutrient loading from all different land uses. The potential for non-point source pollution from watersheds has recently increased where Confined Animal Feeding Operations (CAFOs) are found due to tremendous growth in the number of CAFOs in the U.S. Manure produced from CAFOs, especially broiler litter, is applied to crops as a low cost fertilizer. Even though N and P contents of animal manure are about the same, manure application rates intended to supply N to crops may result in over application of P, because crops generally require more N than P (Jongbloed and Lenis, 1998).

TMDLs and Models

Models represent a scientific understanding of certain stressors with respect to water quality (Committee to Assess the Scientific Basis of the Total Maximum Daily Load Approach to Water Pollution Reduction, 2001). One accepted watershed-scale model for identifying stressors of contaminants is the Soil Water Assessment Tool (SWAT). Models must produce reasonable results, especially when predicting TMDLs (Arnold et al., 1998). In order for a watershed scale model to produce reasonable results, it must first be calibrated using data collected from the site being modeled or a similar site. Data collected from research sites, along with any other applicable data sets that are available in the literature, can develop land use specific parameters (calibrate) for the SWAT model.

The literature implies that SWAT has been used worldwide to model watersheds of varying sizes, from 3.3 km^2 to 598,538 km^2 (Bosch et al., 2004). However, there is limited literature available on the use of SWAT in TMDL development. There is also limited literature on the use of SWAT to predict sediment and phosphorus in small, single land use watersheds. These criteria are especially true for the Piedmont region of Georgia.

SWAT parameters need to be evaluated for sediment and phosphorus predictions in small, Piedmont watersheds where there is a single land use defined. One way to do this is to collect as much data as possible from these watersheds and compare them to model predictions. If model predictions for sediment are reasonable for small watersheds, then the sediment TMDL development process will become much more efficient. The first objective of this study was to determine sediment and phosphorus loads from four watersheds in the Georgia Piedmont. The second objective of this study was to assess SWAT for TMDL applications, based on its performance when modeling sediment and phosphorus, in two small, agricultural, Georgia Piedmont watersheds.

Literature Review

Sediment

Soil erosion, whether by wind or water, results in the loss of soil productivity and an increase in surface water contamination by suspended sediment (Gardiner and Miller, 2004). Increased suspended sediment concentrations (SSC) adversely affect the biological health of streams. Increased stream sediment also causes other problems, such as loss of reservoir storage, which can result in flooding (Sediment TMDL Technical Advisory Group, 2002). Because of the biological effects of sediment, the state links excessive SSC with a low index of biological integrity (IBI) score. The IBI has been successfully applied to various regions of the world in an effort to develop a quantitative measurement of fish and invertebrate assemblages (An and Choi, 2003; King and Richardson, 2003). If SSC is determined to be excessive based on the IBI, the stream is listed on the USEPA 303(d) list of impaired waters, however, a measurement of sediment may or may not be taken (Sediment TMDL Technical Advisory Group, 2002). Modeling streams for suspended sediment concentration will ultimately reduce the number of streams inaccurately listed as impaired, as well as provide an assessment and correctional tool for those that are impaired.

Soil erosion and land use share a positive relationship because the potential for soil erosion is great in areas where agriculture, silviculture, and other land development occur. When best management practices are not implemented to control runoff, soil erosion rates are intensified. In a study conducted in the Georgia Piedmont, Byers et al. (2005) found that cattle-grazed pastures with unfenced streams contributed significantly to TSS loading in streams. Median storm flow TSS concentrations ranged from 218 to 507 mg L^{-1} and median base flow TSS concentrations ranged from 16.6 to 37.4 mg L^{-1} in two unfenced streams. Median TSS losses per storm event in this study ranged from 22 kg ha^{-1} to 121 kg ha^{-1}. A second study conducted by Hewlett (1979) in the Georgia Piedmont showed that poor access roads and undesirable streamside management during silvicultural operations contributed to 90% of sediment exports in a clear-cut forest. In this study, the maximum sediment yield reached 8000 kg ha^{-1} yr^{-1} after road construction and machine planting.

SWAT must be evaluated over an extended range of applications before it can be viewed as a reliable source for TMDL assessment and development. One of the most important applications is to determine ease with which the model can be calibrated for sediment. A study conducted by Jha et al. (2003) showed that SWAT calibrations for discharge and sediment yields could be reasonably obtained in the Raccoon River Watershed, located in West Central Iowa. In this study, SWAT was calibrated using measured stream flow and sediment loading data from 1981-1990. The results show a strong correlation (R^2 = 0.88) between predicted and observed flow data for the calibration period when a monthly time-step was used. This study also indicated that the SWAT sediment prediction was reasonable, but did not present the correlation coefficients.

The second most important application for which SWAT must be evaluated is how well it can be validated for sediment once it has been calibrated. Validation is the evaluation of model accuracy by making comparisons of simulated model output with experimental data collected at times, locations, or depths different from the experimental data used in the calibration phase (Mulla and Addiscott, 1999). A study conducted by Santhi et al. (2001a) showed that, during SWAT validation, observed and simulated values for sediment were satisfactory ($R^2 \geq 0.6$), except for when flow was under predicted. This study was conducted on the 4277-km^2 Bosque River Watershed, where water from the watershed flows into Lake Waco, TX, and is used for drinking water.

Several studies evaluate SWAT's performance when modeling large watersheds for sediment (Bosch et al., 2004; Santhi et al., 2001a). Modeling large watersheds is much easier because there are more data available, especially where larger rivers are present. Although lack of available data means that small watersheds are less frequently modeled, some studies show that SWAT performs well when modeling sediment in a small watershed. Chu et al. (2004) conducted a study on a small (346 ha), Piedmont watershed in Maryland. While SWAT predictions for monthly sediment loading were poor ($R^2 = 0.05$), predictions for yearly sediment loading were good ($R^2 = 0.90$).

Phosphorus

In general, P contamination of ground and surface waters poses a serious threat to water quality and is related to land use. The Upper Oconee Watershed of Georgia represents an area that consists of numerous agricultural land uses, including, but not limited to crop production, beef and dairy cattle production, and poultry production. In a study conducted in the Upper Oconee Watershed, Fisher et al. (2000) found that stream and river P concentrations ranged from 0.029 to 0.104 mg L^{-1} between 1995 and 1996.

Agriculture was cited by the USEPA as the primary source of 60% of impaired rivers, 30% of impaired lakes, and 15% of impaired estuaries (USEPA, 1998). In 1986, the USEPA recommended that to control accelerated or cultural eutrophication, total phosphates should not exceed 100 μg L^{-1} in streams not discharging directly into a lake or reservoir, 50 μg L^{-1} in any stream at the point where it enters into a lake or reservoir, and 25 μg L^{-1} within a lake or reservoir (USEPA, 1986). In 2000, the USEPA began to develop ecoregional nutrient criteria intended to address eutrophication. These criteria were developed based on reference waters (surface waters that have been minimally disturbed by human activity). The new USEPA recommendation for total phosphorus was ≤8 μg L^{-1} for lakes and reservoirs, and ≤10 μg L^{-1} for streams and rivers (USEPA 2000a; USEPA 2000b). A study conducted on the Iowa Clear Lake Agricultural Watershed showed mean concentrations of TP ranged from 275 to 474 μg L^{-1} in over a 2-yr period. Mean TP loads from two gauged sub-basins within the watershed were 1504 and 1510 g P ha^{-1} in 1999 and 759 and 951 g P ha^{-1} in 2000 (Klatt et al., 2003).

Phosphorus is delivered to surface waters in several ways, including surface runoff, and through direct defecation of animals into the water. High concentrations of P in runoff are sometimes associated with areas where there are high densities of animals such as pastures, CAFOs, and in areas where manure is applied as fertilizer. Several factors that influence P losses from areas where manure is applied include application rate, method, time of application, soil test P (STP), and the amount of time between manure application and runoff-producing rainfall. Byers et al. (2005) concluded that cattle-grazed pastures contribute significant DRP and TP loads to surface waters during both storm flow and base flow. Several studies have shown a strong correlation between stream P concentrations and land use during storm events when animal wastes were applied (Hunter et al., 2000; McFarland and Hauck, 1999; Richards et al., 2004). The literature also suggests P concentrations in runoff were highest after the first rainfall event occurred since poultry litter was applied (Edwards and Daniel, 1994; Heathman et al., 1999).

SWAT has been integrated into other nutrient models for large watersheds. Li et al. (2004) integrated the SWAT along with the DAYCENT biogeochemistry model (a daily time-step version of the CENTURY model) to simulate N export from the Hubbard Brook Experimental Forest watershed (3160 ha) located in central New Hampshire. The DAYCENT-SWAT model showed that combining the two models was feasible and that satisfactory predictions of stream flow and mineral N were observed when simulated data were compared to long-term monitoring data.

A second application of SWAT demonstrated the utility of the model in the TMDL development process for estimating P concentrations in the Bosque River Watershed (4277 km^2), located in north central Texas (Santhi et al. 2001b). Dairy manure Best Management Practices (BMPs) were analyzed for their effectiveness using model simulations. Three dairy site BMPs were considered in this study: 1) haul-off of all solid dairy manure from the watershed, 2) crop P requirement application of manure, and 3) P diet reduction in animal feed. When dairy site BMPs were used, time-weighted soluble P was reduced 1-12% and flow weighted soluble P was reduced 7-60%. The researchers concluded that the modeling methods were adaptable to similar large watershed TMDL projects in other parts of the United States.

Soil Water Assessment Tool (SWAT)

SWAT is a physically based, continuous time model that was developed to predict the impact of land management practices on water, sediment, and agricultural chemical yields in large complex watersheds with a variety of soils and land use and management conditions (Neitsch et al., 2000). The following input information is needed in order for the model to function: digital elevation model (DEM), national land cover database (NLCD), and soils (STATSGO or SSURGO). The Soil and Water Assessment Tool User's Manual provides a detailed description of the SWAT and the governing equations for processes simulated by the model (Neitsch et al., 2000). The main equations used by the model for simulating flow, sediment, and nutrients in a watershed are described below.

To simulate flow, individual subbasins are delineated within a watershed based on digital elevation model data so that water movement can be divided between land and channel routing phases of the model. To accurately predict the movement of contaminants in a watershed SWAT uses the water balance equation as the driving force to simulate processes within the watershed:

$$SW_t = SW_o + \sum_{i=1}^{t}(R_{day} - Q_{surf} - E_a - w_{seep} - Q_{gw}) \qquad \textbf{Eqn. 1.1}$$

where SW_t is the final soil water content (mm H_2O), SW_o is the initial soil water content on day i (mm H_2O), t is the time (days), R_{day} is the amount of precipitation on day i (mm H_2O), Q_{surf} is the amount of surface runoff on day i (mm H_2O), E_a is the amount of evapotranspiration on day i (mm H_2O), w_{seep} is the amount of water entering the vadose zone from the soil profile on day i (mm H_2O), and Q_{gw} is the amount of return flow on day i (mm H_2O). Canopy storage, infiltration rates, evapotranspiration, subsurface flow, surface runoff, length and slope of channels, and land cover are all taken into account as flow out of a watershed is predicted by the model.

The erosion and sediment yield components of the SWAT are estimated for each HRU by the Modified Universal Soil Loss Equation (MUSLE):

$$sed = 11.8 \cdot (Q_{surf} \cdot q_{peak} \cdot area_{hru})^{0.56} \cdot K_{USLE} \cdot C_{USLE} \cdot P_{USLE} \cdot LS_{USLE} \cdot CFRG \qquad \textbf{Eqn. 1.2}$$

where *sed* is the sediment yield on a given day (metric tons), Q_{surf} is the surface runoff volume (mm H_2O ha^{-1}), q_{peak} is the peak runoff rate (m^3 s^{-1}), $area_{hru}$ is the area of the HRU (ha), K_{USLE} is the USLE soil erodibility factor (metric ton*m^2*hr m^{-3}*metric ton^{-1}*cm^{-1}), C_{USLE} is the USLE cover and management factor, P_{USLE} is the USLE support practice factor, LS_{USLE} is the USLE topographic factor, and *CFRG* is the coarse fragment factor. For sediment routing purposes, the maximum amount of sediment that can be transported from a stream segment is a function of peak channel velocity, and is governed by the following equation:

$$conc_{sed,ch,mx} = c_{sp} \cdot v_{ch,pk}^{\ sp\exp} \qquad \textbf{Eqn. 1.3}$$

where $conc_{sed,ch,mx}$ is the maximum concentration of sediment that can be transported by the water (ton m^{-3} or kg L^{-1}), c_{sp} is a coefficient defined by the user, $v_{ch,pk}$ is the peak channel velocity (m s^{-1}), and *spexp* is an exponent defined by the user that varies between 1.0 and 2.0.

Six different pools of P are monitored by SWAT. Phosphorus has three inorganic pools and three organic pools. The SWAT user has the option to define P concentrations at the beginning of a simulation. If these concentrations are not defined, the SWAT will automatically define initial nutrient levels in the different pools. The P cycle is modeled by SWAT with an extensive network of equations. Detailed equations for the P cycle are located on pages 193-204 of the SWAT Users Manual.

Phosphorus movement from land to the steam network is modeled by SWAT with the algorithms outlined below. SWAT calculates soluble P transport in surface runoff with the following equation:

$$P_{surf} = \frac{P_{solution,surf} \cdot Q_{surf}}{\rho_b \cdot depth_{surf} \cdot k_{d,surf}} \qquad \textbf{Eqn. 1.4}$$

where P_{surf} is the amount of soluble P lost in surface runoff (kg P ha^{-1}), $P_{solution,surf}$ is the amount of P in solution in the top 10 mm (kg P ha^{-1}), Q_{surf} is the amount of surface runoff on a given day (mm H_2O), ρ_b is the bulk density of the top 10 mm (Mg m^{-3}), $depth_{surf}$ is the depth of the surface layer (mm), and $k_{d,surf}$ is the P soil partitioning coefficient (m^3 Mg^{-1}).

The amount of P transported to the steam with sediment is calculated with the following equation:

$$sedP_{surf} = 0.001 \cdot conc_{sedP} \cdot \frac{sed}{area_{hru}} \cdot \varepsilon_{P:sed} \qquad \textbf{Eqn. 1.5}$$

where $sedP_{surf}$ is the amount of P transported with sediment to the main channel in surface runoff (kg P ha^{-1}), $conc_{sedP}$ is the concentration of P attached to sediment in the top 10 mm (g P metric ton soil^{-1}), sed is the sediment yield on a given day (metric tons), $area_{hru}$ is the HRU area (ha), and $\varepsilon_{P:sed}$ is the P enrichment ratio.

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CHAPTER 2

LAND USE COMPARISON OF SEDIMENT AND PHOSPHORUS LOADS IN THE GEORGIA PIEDMONT, USA

[1]J.K. Bradshaw, D.E. Radcliffe, M.L. Cabrera, L.T. West, C.R. Jackson, D. Markewitz. To be submitted to *Journal of Environmental Quality*.

Abstract

In Georgia, over 600 stream segments are scheduled for TMDL development. The state does not have quantitative standards for regulating sediment and phosphorus (P) concentrations. The development of sediment and P loads using appropriate reference streams with minimum development may be a way to determine what the maximum daily contaminant loading should be for impaired streams. This study was initiated to compare suspended sediment, dissolved P, and total P in four streams with different predominant land uses. Two streams drain areas that were entirely forested represent reference streams for the purpose of this study. Two streams drain areas that are predominately agricultural (dairy land use and cattle and poultry land use). Suspended sediment concentration (SSC), dissolved reactive P (DRP), total P (TP) concentrations, and stream stage were measured and annual sediment and P loads were calculated for each stream from 2003 to 2005. Sediment loads ranged from 241 to 1725 kg $ha^{-1}yr^{-1}$ and were lowest in the control forest stream and highest in a second forested stream after clear-cutting occurred. Total P loads ranged from 0.15 to 1.87 kg $ha^{-1}yr^{-1}$ and were smallest in a forested watershed and greatest in the dairy watershed. Overall sediment loads were consistent between the two agricultural watersheds and the non-harvested forest watershed, while sediment loads in the harvested forest watershed were considerably greater, before and after timber harvest. Total P loads were consistent between the two forested sites and the cattle and poultry watershed for all years but considerably greater on the dairy watershed.

Introduction

The 1972 Clean Water Act initiated the Total Maximum Daily Load (TMDL) program to aid states in meeting water quality standards. A TMDL is the maximum amount of a particular pollutant that can be allowed into a stream for it to meet its designated use (USEPA, 2006). For TMDLs to be developed, the USEPA requires that sources of contaminants be identified. Streams that have been identified as not meeting their designated use are listed on the State's 303(d) list. Once a stream is listed, a plan must be developed to set a TMDL for that particular stream. According to the USEPA, 15% of our nation's watersheds have relatively good water quality, 36% are moderately impaired, 22% are seriously impaired, and 27% do not have enough information to be classified (USEPA, 2002). The USEPA Clean Water Action Plan of 1998 states that polluted runoff, specifically from agricultural non-point sources, is the most important source of water contamination preventing the attainment of water quality goals identified by the Clean Water Act (USEPA, 1998). Sediment and nutrients, especially from agricultural practices, are considered two key contributors of poor water quality.

Several studies have used rating curves (flow vs. contaminant) to develop a continuous concentration trace of sediment and phosphorus (Landers et al., 2002; Robertson and Roerish, 1999; Simon et al., 2002). Robertson and Roerish (1999) evaluated several different sampling strategies coupled with load calculations from rating curves and found that the most effective sampling strategy for estimating loads in studies longer that 2 yr in duration was a fixed-period semi-monthly sampling strategy along with storm-chasing (i.e. collecting samples throughout the duration of a storm). Median absolute errors were 25% for sediment and 28% for total P using this method. Yields in eight small watersheds (1400-11000 ha) in southern Wisconsin ranged from 156-1370 kg ha^{-1} yr^{-1} for sediment and from 0.61-3.15 kg ha^{-1} yr^{-1} for TP (Robertson and

Roerish, 1999). Using rating curves, Landers et al. (2002) calculated sediment loads from six watersheds (ranging from 21-420 km^2) in Gwinnett Co., Georgia. Sediment yields ranged from 567-4535 kg ha^{-1} yr^{-1} from 1998-2001. Overall this study concluded that water quality worsened during storm flow, especially nonpoint-source constituents, and seasonal and annual loads vary with precipitation and stream flow while base flow loads are relatively constant from year to year (Landers et al., 2002).

Simon et al. (2002) evaluated reference streams in eight Level III ecoregions to determine what sediment yields were in stable streams representative of the particular ecoregion. In this study, stable channel conditions were represented by Stage I and VI stream conditions and represented background transport rates. The median sediment yield at effective discharge was 3311 kg ha^{-1} yr^{-1} for the Southeastern Plains. The authors note that these findings are only preliminary and further study is required to determine reference conditions for sediment in each ecoregion.

Determining background or "reference" conditions for streams in Georgia is important because the state does not have numerical standards for sediment and P. It is also importing to evaluate how water quality conditions in a single land use, such as agriculture, compare to reference streams. The purpose of this study was to compare sediment and P loads between agricultural and forested (reference) streams and to determine what the maximum contaminant loads should be to improve water quality.

Materials and Methods

Study Sites

Two forested watersheds located in Putnam County, Georgia were selected as reference watersheds for this study. Both watersheds consisted of a combination of 30-year-old loblolly pines (*Pinus taeda*) and mixed hardwoods. H-flumes were installed on the main outlet of each watershed during a forest water quality study conducted by Hewlett (1979). One watershed (42.5 ha) served as a Control Forest watershed and was not harvested during the study period. The second watershed (32.5 ha) served as a Treatment Forest and was harvested in January 2004. According to Hewlett (1979), the soils in these watersheds were severely eroded under cotton farming between 1880 and 1920. [1]Cecil, [1]Pacolet, and [2]Mecklenburg ([1]fine, kaolinitic, thermic Typic Kanhapludults; [2]fine, mixed, active, thermic Ultic Hapludalfs) soil series account for approximately 60% of the experimental area.

Two agricultural watersheds, located in different counties, were also evaluated in this study. The Springfield Creek watershed was located in Morgan County, Georgia. This watershed (295 ha) was used primarily for dairy cattle production and we will refer to it as the Dairy watershed. Land cover within the watershed consists of 55% pasture, 28% forest, and 6% row crops (NRSAL, 1998). [1]Davidson, [2]Lloyd, and [3]Tocooa ([1]fine, kaolinitic, thermic Rhodic Kandiudults; [2]fine, kaolinitic, thermic Kanhapludults; [3]coarse-loamy, mixed, active, nonacid, thermic Typic Udifluvents) soil series cover approximately 96% of the watershed (USDA, 2005a). The Cattle and Poultry watershed was located in Oconee County, Georgia. This watershed (855 ha) was used primarily for cattle and poultry production and we will refer to it as the Cattle and Poultry watershed. Land cover within the watershed consists of 67% pasture, 31% forest, and 2% row crops (NRSAL, 1998). [1]Appling, [1]Cecil, [2]Davidson, and [1]Pacolet ([1]fine, kaolinitic, thermic Typic Kanhapludults; [2]fine, kaolinitic, thermic Rhodic Kaniudults) soils series cover approximately 90% of the watershed (USDA, 2005b). Both watersheds receive anunual poultry litter applications of approximately 2000 kg ha^{-1} yr^{-1}. Measurements were taken at road culverts for both agricultural sites. Monitoring at the outlet of each watershed began in 2003 and continued through the end of 2005.

Sample Collection

Automated ISCO 6700 water samplers (ISCO Inc, Lincoln, NE) were installed at each monitoring site to collect storm samples. The ISCO samplers in the forested watersheds were powered by 12-volt, deep-cycle marine batteries, that were recharged biweekly. The ISCO samplers in the agricultural watersheds were powered by 12-volt, deep-cycle marine batteries connected to solar panels. Solar panels were not used in the forested watersheds because there was not enough sunlight penetration through the canopy for them to be effective.

The ISCO samplers were programmed to take multiple discrete samples during a storm event. This was done using a pressure transducer installed vertically in the stream through a PVC pipe with holes drilled in it. The PVC pipe was used to protect the pressure transducer from debris and sediment being transported by the stream. The pressure transducer was connected to the ISCO sampler's controller, which recorded the date, time, stream stage, and sample number (when applicable), every 5 min for the duration of the study. Sample collection was triggered by a predetermined stage height that was manually programmed into the ISCO sampler. Samples were picked up within 24 h of collection and stored in a refrigerator at 4°C until analyzed. Base flow grab samples were collected weekly in addition to the storm flow samples. Rainfall was measured in each watershed with a tipping-bucket rain gauge connected to a data logger that was programmed to record precipitation amounts every 5 min.

Standard equations for flumes were used to calculate flow based on stream stage at the forested sites. Rating curves were developed at each of the agricultural site culverts so that stream stage could be converted to flow. To construct the rating curves, culvert dimensions were measured at each site to determine the hydraulic radius of the stream. Stream velocity, hydraulic radius, slope, and an estimated roughness coefficient were used to estimate flow for a given stage height. Individual discharge rating curves were then created using FlowMaster (Haestad Methods, Waterbury CT). Finally, manual flow measurements were taken at each agricultural site and the roughness coefficient was adjusted to get a more accurate flow measurement.

Sample Analysis

Base and storm flow samples collected from each site were analyzed for sediment and P concentrations. Suspended Sediment Concentration (SSC) was determined by filtering a sample on pre-weighed 0.45µm filter, drying the filter at 106°C for 24 h, and reweighing the filter (Guy, 1969). Dissolved reactive P (DRP) for filtered storm and base flow samples was determined colorimetrically with an RFA-300 Series Alpkem Analytical autoanalyzer (Murphy and Riley, 1962). Total P (TP) for unfiltered storm and base flow samples was determined by persulfate digestion and measured colorimetrically with the autoanalyzer (Bremner, 1996).

Rating Curves and Load Calculations

Rating curves were developed for SSC, DRP, and TP by plotting log of the concentration vs. log of the normalized flow (Q/Q_0) where: Q was stream discharge at the time a sample was collected, and Q_0 was the mean discharge for the stream for the duration of the study. Using the rating curves, loads were estimated at each 5 min interval that flow was above the average base flow by multiplying the measured flow by the rating curve concentration associated with that flow and by 5 min. Under base flow conditions, the average contaminant concentration from grab sample data was used to calculate a load for the each 5 min interval that flow was at or below the average base flow. Unit-area annual loads were calculated by summing the loads calculated at each 5 min interval and dividing by the watershed area. Unit-area annual loads were also calculated based on storm and grab samples that were collected so that we could compare them to loads calculated from the rating curves. Daily flow-weighted daily load for each time a sample was determined by summing the sample loads calculated for each day:

$$C_L = \sum_{i=1}^{n} C_t * Q_t * \Delta t \qquad \textbf{Eqn. 2.1}$$

where: C_L is the daily contaminant load, C_t is the contaminant concentration at the time the sample was taken, Q_t is the flow at the time the sample was taken, and Δt is the amount of time between samples for any given day. Annual loads were then calculated by summing the daily loads for each year and dividing by the watershed area.

Results and Discussion

Timber Harvest Effect on Sediment and Phosphorus in Forested Watersheds

Three rating curves (pre-harvest, harvest, and replant) were constructed for each forested watershed to compare the effects of timber harvesting between the two sites. For the purpose of comparing the two forested watersheds, “pre-harvest” represents all data collected from 1 Jan. 2003 to 18 Dec. 2003, “harvest” represents all data collected from 18 Dec. 2003 to 30 Jan. 2005, and “replant” represents all data collected from 31 Jan. 2005 to 31 Dec. 2005. Coefficients of determination (R^2) ranged from 0.55 to 0.74 for SSC rating curves, and from 0.07 to 0.55 for TP rating curves (Fig. 2.1-2.4). Rating curves were not used to calculate loads for DRP in the forested watersheds because we found no correlation between DRP and normalized flow ($R^2 \leq$

0.1). Instead, DRP loads were calculated from base and storm flow samples collected during each harvest period.

Regression slopes for SSC rating curves ranged from 0.470 in the Control Forest watershed during the harvest period to 1.09 in the Treatment Forest watershed in the replant period (Table 2.1). Regression slopes and intercepts were compared at both sites using PROCGLM (SAS Institute, 1999). We found that there was not a significant difference in slope at the Control Forest watershed between the three harvest periods but slopes were significantly different at the Treatment Forest watershed for all three periods (Table 2.1). At the Control Forest watershed the intercept for the "harvest" regression equation was significantly different, and in the Treatment Forest watershed the intercept for the "replant" period was significantly different (Table 2.1). Regression slopes and intercepts were also tested between sites for each harvest period. We found that slopes were significantly different between the Control Forest and Treatment Forest watersheds for all three harvest periods. Intercepts were significantly different between the two sites during the "harvest" and "replant" periods but not for the "pre-harvest" period.

Regression slopes for TP rating curves ranged from 0.146 in the Control Forest watershed to 0.574 in the Treatment Forest watershed. Slopes were not significantly different in the "pre-harvest" and "harvest" periods in both the Control Forest and Treatment Forest watersheds. However, the slope was significantly different for both watersheds in the "replant" period (Table 2.2). Rating curve intercepts were not significantly different for any of the three harvest periods in the Control Forest watershed, but were significantly different in the "harvest" and "replant" periods in the Treatment Forest watershed (Table 2.2). Figures 2.1-2.4 show relationships between log concentration (SSC and TP) and log *Q/Qo*.

Unit-area sediment loads ranged from 245 kg ha^{-1} yr^{-1} in the Control Forest watershed to 1543 kg ha^{-1} yr^{-1} in the Treatment Forest watershed (Table 2.3). Loads were approximately 3.5 times larger in the Treatment Forest watershed than in the Control Forest watershed in the pre-harvest period as well. Sediment loads increased slightly in both watersheds during the harvest period and decreased by approximately 50% during the replant period. Mean SSC's ranged from 140 to 544 mg L^{-1} from storm flow were smallest in the Control Forest watershed and greatest in the Treatment Forest watershed during the harvest period (Table 2.4). These results are lower than total suspended sediment (TSS) concentrations from Byers et. al (2005) who found median TSS of 0.42 and 0.64 mg L^{-1} in runoff from 2 pastures. Mean SSC increased significantly in the treatment watershed during the harvest period (Table 2.1).

Total P loads ranged from 0.142 kg ha^{-1} yr^{-1} in the control watershed to 0.498 kg ha^{-1} yr^{-1} in the Treatment Forest watershed (Table 2.3). Total P loads were approximately 2 times greater in the Treatment Forest watershed than in the Control Forest watershed at the beginning of the study. In the Control Forest watershed, TP loads were consistent during the pre-harvest and harvest periods and decreased during the replant period. In the Treatment Forest watershed, TP loads increased during the harvest period. Total P loads decreased by approximately 50% in both watersheds during the replant period. Mean TP concentrations ranged from 0.063 mg L^{-1} in the Control Forest watershed to 0.122 mg L^{-1} in the Treatment Forest watershed (Table 2.4). The lowest mean TP concentrations occurred during the replant periods in both watersheds and the highest TP concentration was observed in the Treatment Forest watershed during the harvest period. DRP loads ranged from 0.050 to 0.087 kg ha^{-1} yr^{-1} and did not show a trend specific to harvest periods in either watershed (Table 2.3). Phosphorus loads tended to fluctuate in the same manner as sediment loads, indicating that the disturbance of sediment in both watershed contributed more phosphorus to the stream.

Land Use Comparison of Sediment Loads

Regression slopes for sediment rating curves ranged from 0.470 in the Control Forest watershed to 1.25 in the Dairy watershed (Table 2.1). R^2 values ranged from 0.48 in the Cattle and Poultry watershed to 0.74 in the Treatment Forest watershed. In general, the best R^2 was obtained in the forested watersheds where stage measurements were taken from flumes. Figure 2.5 shows the sediment rating curves for the agricultural watersheds.

Sediment loads calculated with rating curves ranged from 241 to 1725 kg ha^{-1} yr^{-1} in the forested watersheds and from 269 to 1617 kg ha^{-1} yr^{-1} in the agricultural watersheds (Table 2.5). Annual mean sediment loads for the forested watersheds ranged from 737 to 1092 kg ha^{-1} yr^{-1} and mean sediment loads for the agricultural watersheds ranged from 362 to 978 kg ha^{-1} yr^{-1} (Table 2.6). Mean sediment loads from rating curves were tested, but there was not a significant difference between the two land uses at $\alpha = 0.05$. The overall range of mean sediment loads calculated in our study watersheds were smaller than loads calculated in the study by Landers et al. (2002). The maximum sediment loads calculated from our study watersheds (1725 kg ha^{-1} yr^{-1}) was also smaller than the median sediment yield calculated by Simon et al. (2002) as the reference sediment condition (3311 kg ha^{-1} yr^{-1}) for the Southeastern Plains.

Sediment loads calculated with samples taken during storm and base flows ranged from 179 to 1263 kg ha^{-1} yr^{-1} in the forested watersheds and from 46 to 1175 kg ha^{-1} yr^{-1} in the agricultural watersheds (Table 2.5). It must be noted that auto-sampler malfunctions occurred at Cattle and Poultry in 2004 and Dairy in 2005, preventing us from collecting an adequate number of samples to calculate loads. Using this method, calculated sediment loads were greater at three out of four watersheds when more storm flow days were sampled during the year.

Land Use Comparison of Phosphorus Loads

Figures 2.6 and 2.7 show phosphorus rating curves for the agricultural watersheds. Rating curves slopes for DRP in the agricultural watersheds were 0.522 and 0.563 for Dairy and Cattle and Poultry, respectively (Table 2.7). R^2 values were low in the agricultural watersheds (0.24-0.35).

Total P unit-area loads calculated from rating curves ranged from 0.145 to 0.542 kg ha^{-1} yr^{-1} in the forested watersheds and from 0.233 to 1.87 kg ha^{-1} yr^{-1} in the agricultural watersheds (Table 2.8). The smallest TP loads were observed in the Control Forest watershed and Cattle and Poultry watersheds and the largest TP loads were observed in the Dairy watershed. Averaged across land use, annual TP loads ranged from 0.252 kg ha^{-1} yr^{-1} in the forested watersheds to 0.902 kg ha^{-1} yr^{-1} in the agricultural watersheds (Table 2.6). TP loads between the forested and agricultural land uses were not significantly different at $\alpha = 0.05$. Total P loads calculated in our watersheds were slightly smaller, although comparable to those calculated in the study conducted by Landers et al. (2002), whose TP loads ranged from 0.61-3.15 kg ha^{-1} yr^{-1}. Phosphorus fractions from samples collected during base and storm flow were calculated. The percentage of organic and particulate P was determined by subtracting DRP from TP to get Organic/Particulate P, dividing Organic/Particulate P by total P, and then multiplying by 100%. DRP fractions were calculated by dividing DRP by TP and then multiplying by 100%. In the both the forested and agricultural watersheds, DRP accounted for 38 % of TP and Organic/Particulate P accounted for 62% of TP (Table 2.9). P fractions were not significantly different between land uses at $\alpha = 0.05$. Figure 2.8 shows a graphic of P fractions for each land use. P fractions results from our forested and agricultural watersheds are comparable to data from McDowell et al. (2001) which show DRP and TP data from a watershed in the Appalachian Valley and Ridge Physiographic Province (AVRPP). DRP fractions (% of TP) in runoff from the watershed ranged from 22 to 47%. Land uses in the AVRPP watershed were 50% row crop, 30% forest, 20% pasture. Poultry manure was added to the AVRPP watershed annually at a rate of 85 kg P ha^{-1} yr^{-1}.

Summary and Conclusions

In general, sediment loads from the agricultural watersheds were consistent with loads observed in the Control Forest watershed when rating curves were used to calculate loads. Mean sediment loads from the forested watersheds were greater due to sedimentation occurring in the Treatment Forest watershed. Reduced stream bank stability and greater sediment concentrations in overland flow in the Treatment Forest watershed may have caused mean sediment loads in the forested land use to be larger. Mean sediment load from the mixed agricultural watershed (Cattle and Poultry) was similar to the loads observed in the Control Forest watershed, except in the last year (2005). Construction of a subdivision began on the Cattle and Poultry watershed in this year. Significant differences in SSC rating curve slopes for all three harvest periods from the treatment forest watershed indicate a harvest effect on sediment concentration and that current BMP's may not be adequately reducing runoff under silvicultural practices.

Although P loads were not significantly different between agricultural and forested land uses, P loads were much larger at the Dairy watershed. High animal occupancy at Dairy (approximately 3 animals per hectare) and the over-application of cattle and poultry manure on the watershed may have greatly contributed to the amount of P loading, as indicated by the high fractions of organic/particulate phosphorus in the agricultural land use. High organic/particulate P fractions in both the forested and agricultural land uses likely were the result of stream/wildlife interactions.

Calculating loads from samples collected during the study did not seem to be an accurate method of load estimation when compared to rating curves. However, when concentration and flow relationships were not well correlated as with DRP in the forested watersheds, this method was the only way to calculate loads. Collecting an adequate number of samples to be used for calculating loads proved difficult due to changes in annual flow and weather patterns.

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Table 2.1. Sediment rating curves.

Site/Contaminant	Period	Slope	Intercept	R^2
Control Forest	Pre-Harvest	0.519	1.46	0.67
	Harvest	0.470	1.60*	0.52
	Replant	0.570	1.45	0.56
Treatment Forest	Pre-Harvest	0.728	1.44	0.74
	Harvest	0.869*	1.35	0.55
	Replant	1.09***	1.16***	0.61
Dairy	2003-2005	1.25	1.70	0.60
Cattle and Poultry	2003-2005	1.00	1.75	0.48

*indicates a significant difference in value at p = 0.05
***indicates a significant difference in value at p = 0.001.

Table 2.2. Total P rating curves.

Site/Contaminant	Period	Slope	Intercept	R^2
Control Forest	Pre-Harvest	0.246	-1.47	0.34
	Harvest	0.203	-1.38	0.18
	Replant	0.146*	-1.46	0.07
Treatment Forest	Pre-Harvest	0.365	-1.48	0.55
	Harvest	0.309	-1.26***	0.38
	Replant	0.574***	-1.63***	0.48
Dairy	2003-2005	0.543	-0.393	0.37
Cattle and Poultry	2003-2005	0.426	-1.092	0.32

*indicates a significant difference in value at p = 0.05
***indicates a significant difference in value at p = 0.001.

Table 2.3. Forested watershed load comparisons calculated from rating curves.

Site	Period	SSC (kg ha^{-1} yr^{-1})	TP (kg ha^{-1} yr^{-1})	DRP (kg ha^{-1} yr^{-1})
Control Forest	Pre-Harvest	409	0.240	0.051
	Harvest	417	0.227	0.087
	Replant	245	0.142	0.060
Treatment Forest	Pre-Harvest	1534	0.409	0.050
	Harvest	1543	0.498	0.086
	Replant	727	0.246	0.084

*DRP was calculated from samples and not rating curves

Table 2.4. Mean contaminant concentrations for the forested watersheds.

Site	Period	SSC (mg L^{-1})	TP (mg L^{-1})	DRP (mg L^{-1})
Control Forest	Pre-Harvest	140	0.069	0.003
	Harvest	216	0.090	0.008
	Replant	139.9	0.063	0.008
Treatment Forest	Pre-Harvest	210	0.085	0.008
	Harvest	544	0.122	0.026
	Replant	208	0.063	0.013

Table 2.5. Annual sediment loads.

Watershed	Year	Watershed Area (ha)	Annual Sediment Load—Rating Curves (kg ha^{-1} yr^{-1})	Annual Sediment Load—Samples (kg ha^{-1} yr^{-1})	No. Storm Flow Days Sampled
Control Forest	2003	43	395	1263	62
	2004		461	213	11
	2005		241	179	21
Treatment Forest	2003	32	1124	1070	49
	2004		1725	635	23
	2005		1234	275	35
Dairy	2003	295	269	861	62
	2004		379	1175	10
	2005*		339	59	4
Cattle and Poultry	2003	855	455	448	65
	2004*		359	46	2
	2005		1617	219	22

* denotes a malfunction with the automated sampler resulting in missed storm flow days sampled

Table 2.6. Mean annual loads calculated from rating curves.

Land Use	Year	SSC (kg ha^{-1} yr^{-1})	TP (kg ha^{-1} yr^{-1})	DRP (kg ha^{-1} yr^{-1})
Forest	2003	760	0.267	N/A
	2004	1092	0.394	N/A
	2005	737	0.252	N/A
Agriculture	2003	362	0.804	0.052
	2004	369	0.902	0.212
	2005	978	0.549	0.256

Table 2.7. DRP rating curves.

Watershed	Period	Slope	Intercept	R^2
Dairy	2003-2005	0.522	-0.964	0.35
Cattle and Poultry	2003-2005	0.563	-1.939	0.24

Table 2.8. Annual TP loads.

Watershed	Year	Watershed Area (ha)	Annual TP Load—Rating Curve (kg ha^{-1} yr^{-1})	Annual TP Load—Samples (kg ha^{-1} yr^{-1})	No. Storm Flow Days Sampled
Control Forest	2003	43	0.234	0.641	61
	2004		0.246	0.169	10
	2005		0.145	0.144	21
Treatment Forest	2003	32	0.301	0.278	42
	2004		0.542	0.180	20
	2005		0.358	0.115	35
Dairy	2003	295	1.38	1.48	49
	2004		1.52	1.01	7
	2005*		1.87	0.494	4
Cattle and Poultry	2003	855	0.233	0.177	52
	2004*		0.280	0.082	2
	2005		0.739	0.165	22

* denotes a malfunction with the automated sampler resulting in missed storm flow days sampled

Table 2.9. Fractions of DRP and Organic/Particulate P as a percentage of Total P.

Land Use	DRP (% of TP)	Organic/Particulate P (% of TP)
Forest	38	62
Agriculture	38	62

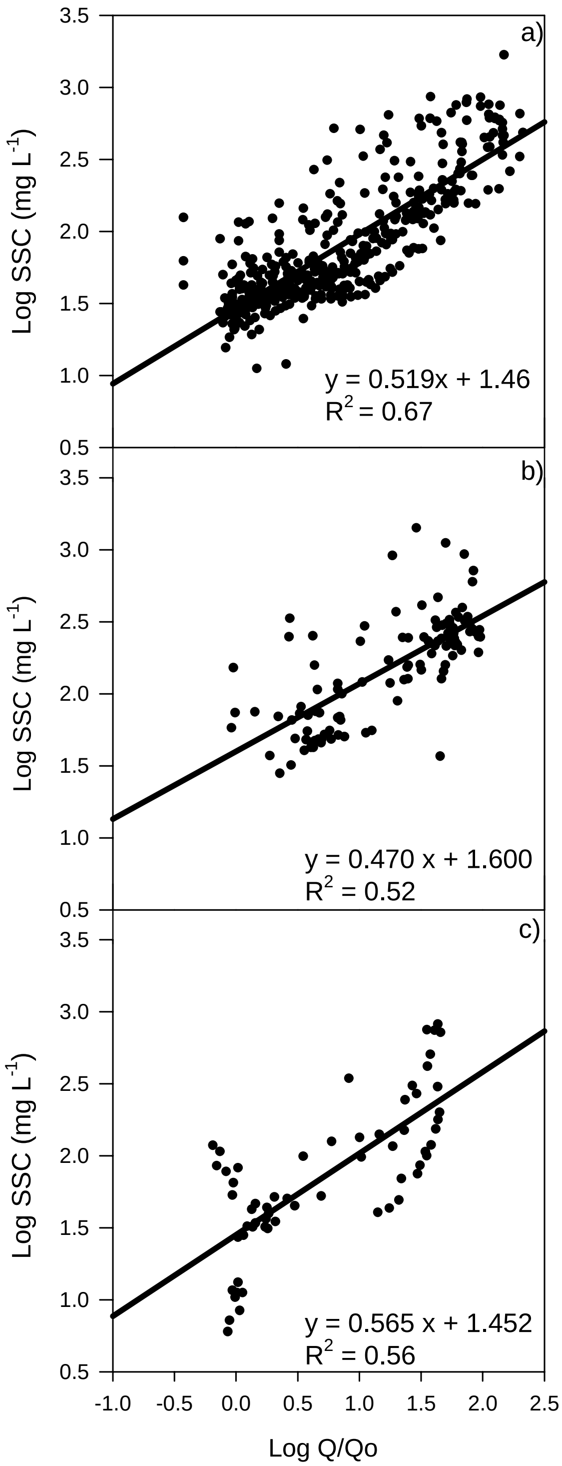

Fig 2.1. Log-log SSC rating curves for the Control Forest watershed. Pre-harvest (a), harvest (b), and replant (c) periods are shown with their respective regression equations and correlation coefficients (R^2).

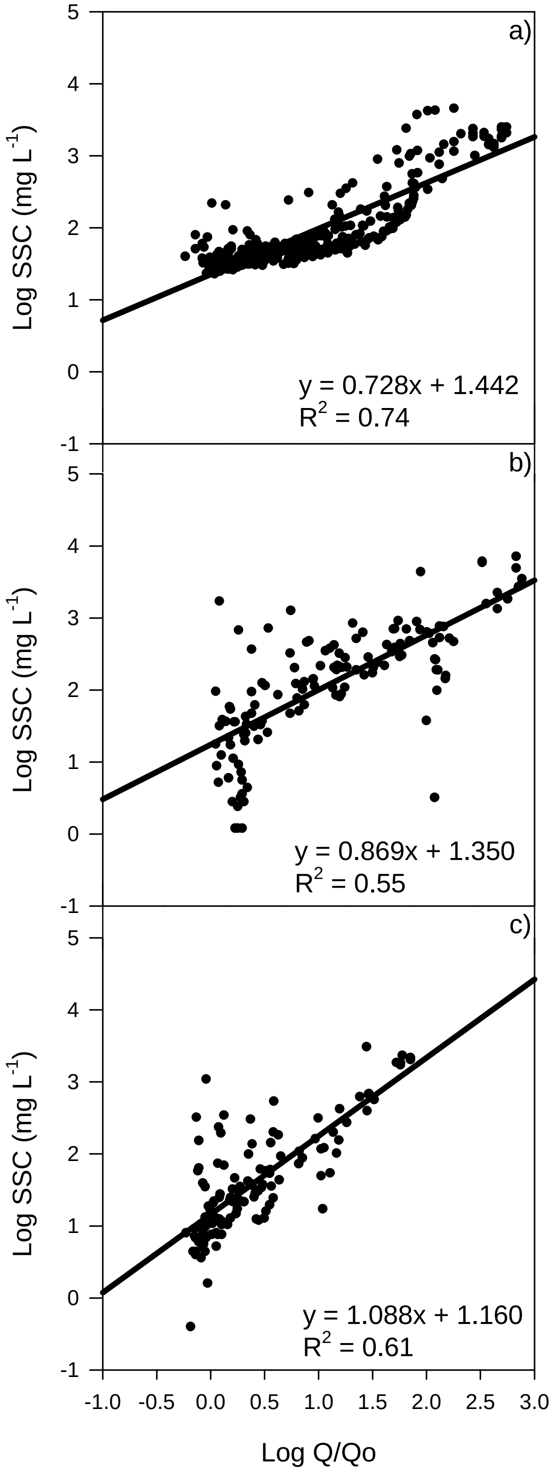

Fig 2.2. Log-log SSC rating curves for the Treatment Forest watershed. Pre-harvest (a), harvest (b), and replant (c) periods are shown with their respective regression equations and correlation coefficients (R^2).

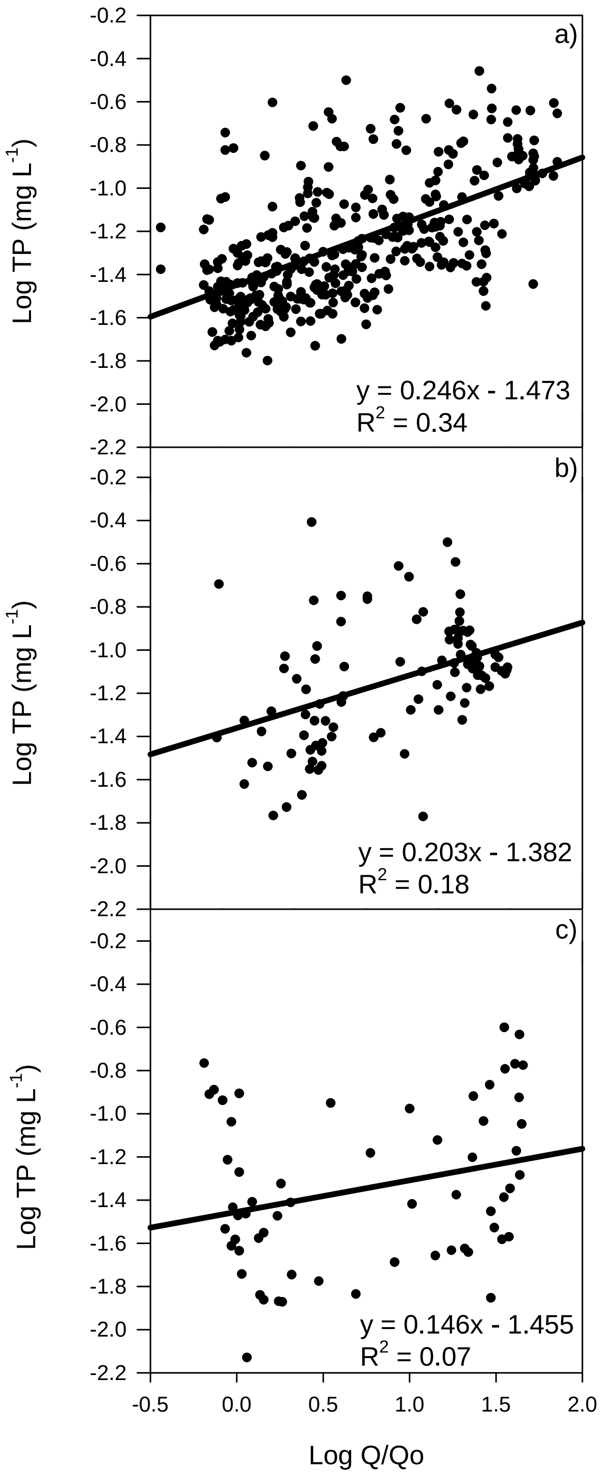

Fig 2.3. Log-log TP rating curves for the Control Forest watershed. Pre-harvest (a), harvest (b), and replant (c) periods are shown with their respective regression equations and correlation coefficients (R^2).

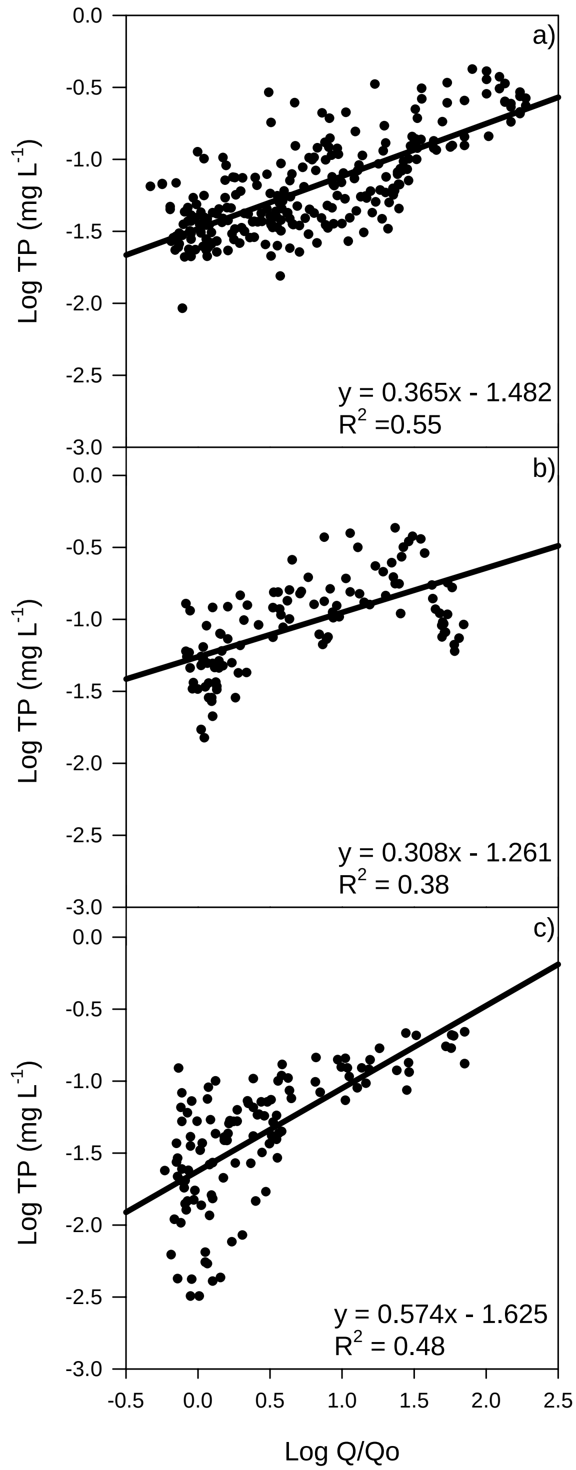

Fig 2.4. Log-log TP rating curves for the Treatment Forest watershed. Pre-harvest (a), harvest (b), and replant (c) periods are shown with their respective regression equations and correlation coefficients (R^2).

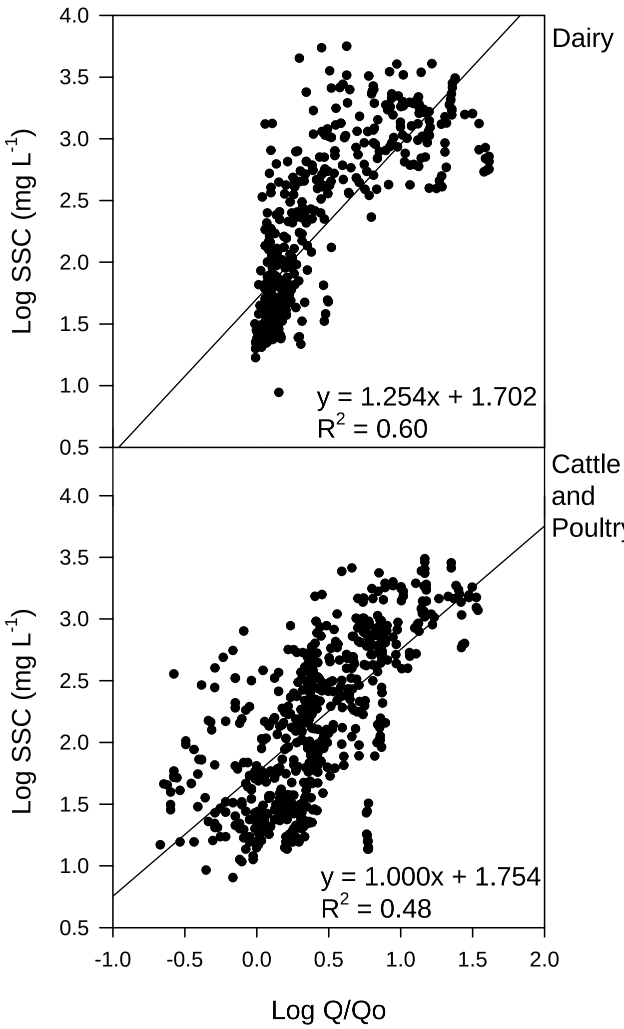

Fig 2.5. Log-log SSC rating curves for the agricultural watersheds. Dairy and Cattle and Poultry are shown with their respective regression equations and correlation coefficients (R^2).

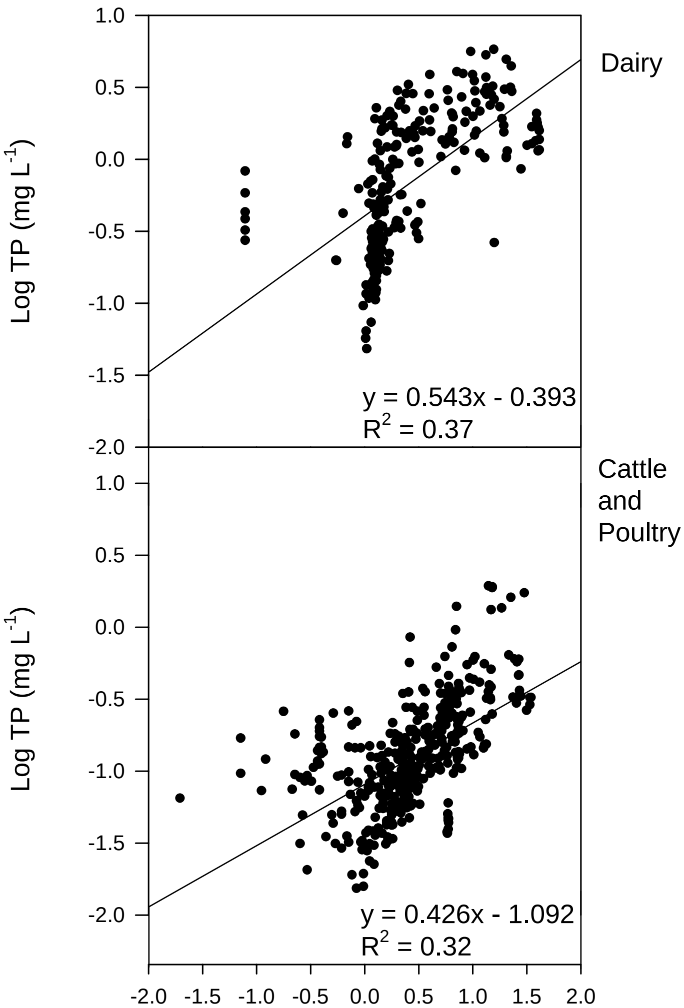

Fig 2.6. Log-log Total P rating curves for the agricultural watersheds. Dairy and Cattle and Poultry are shown with their respective regression equations and correlation coefficients (R^2).

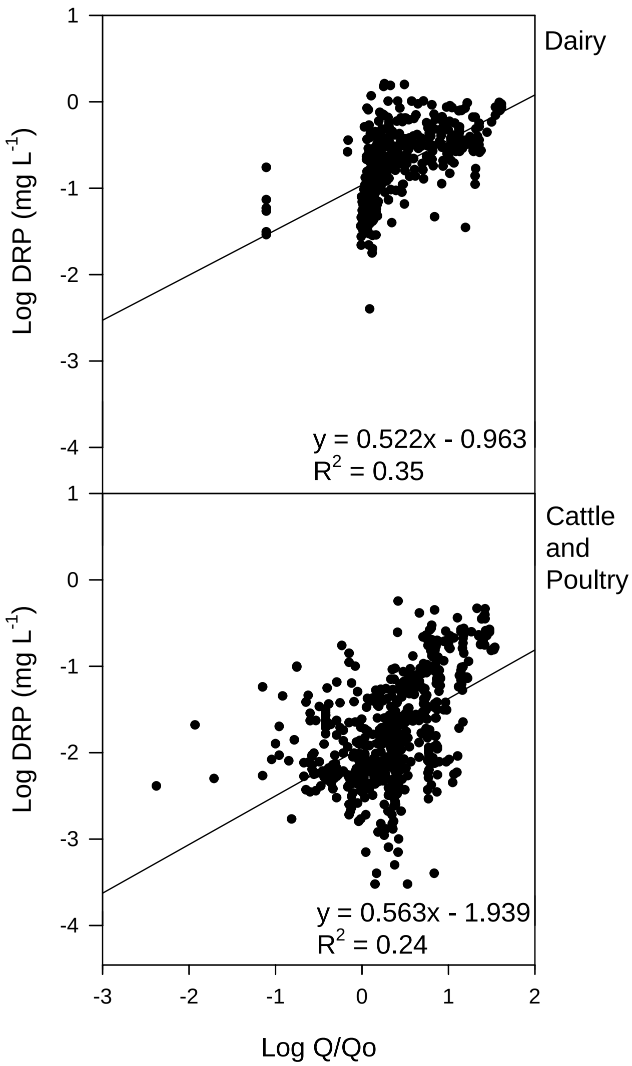

Fig 2.7. Log-log DRP rating curves for the agricultural watersheds. Dairy and Cattle and Poultry are shown with their respective regression equations and correlation coefficients (R^2).

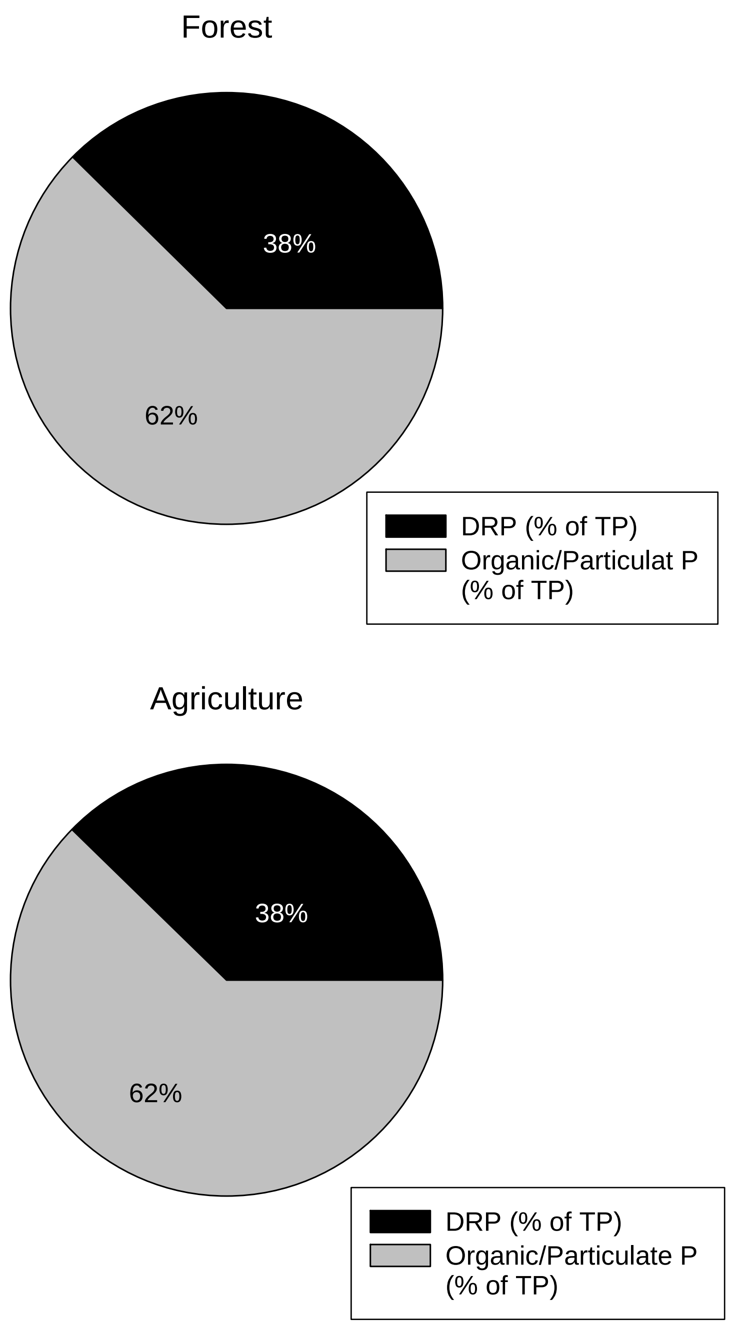

Figure 2.8. Phosphorus fractions for Forested and Agricultural land uses, shown as a percentage of Total P.

CHAPTER 3

SWAT MODEL ASSESMENT OF TWO AGRICULTURAL WATERSHEDS IN THE GEORGIA PIEDMONT, USA

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Abstract

A TMDL is the maximum amount of a pollutant that can be allowed into a stream for it to meet its assigned designated use. For TMDLs to be developed, USEPA requires that sources of contaminants be identified. Streams that do not meet their designated use are listed on the State's 303(d) list. One accepted method of contaminant identification is through the use of watershed-scale models such as the Soil and Water Assessment Tool (SWAT). SWAT has been widely used to model water quality in large watersheds. When modeling a large watershed, it is almost impossible to determine if model parameters used to calibrate SWAT represent real-world values for a specific land use. One way to determine if parameter values are accurate is to model small, single land use watersheds. A research project was initiated to study how model parameter values relate to land use. Flow, sediment, and phosphorus (P) data collected from two agricultural watersheds were used to calibrate SWAT. Base-flow and storm samples were collected and analyzed for Suspended Sediment Concentration (SSC) and total P (TP). SWAT was calibrated by the parameterization method for model calibration outlined by Mulla and Addiscott (1999). Error between predicted and observed sediment loads ranged from 7 to 367% and error between predicted and observed TP loads ranged from 47 to 70%. SWAT parameter values governing sediment and P were not always consistent between the two modeled watersheds and also varied when compared to the calibrated model of a large watershed.

Introduction

The 1972 Clean Water Act initiated the Total Maximum Daily Load (TMDL) program to aid states in meeting water quality standards. A TMDL is the maximum amount of a particular pollutant that can be allowed into a stream for it to meet its assigned designated use. For TMDLs to be developed, the USEPA requires that sources of contaminants be identified. Streams that have been identified as not meeting their designated use are listed on the State's 303(d) list. Once a stream is listed, a plan must be developed to set a TMDL for that particular stream. According to the USEPA, 15% of our nations watersheds have relatively good water quality, 36% are moderately impaired, 22% are seriously impaired, and 27% do not have enough information to be classified (USEPA, 2002). The USEPA Clean Water Action Plan of 1998 states that polluted runoff, specifically from agricultural non-point sources, is the most important source of water contamination preventing the attainment of water quality goals identified by the Clean Water Act (USEPA, 1998). Sediment and nutrients are two main contributors of poor water quality.

Models represent a scientific understanding of certain stressors with respect to water quality (Committee to Assess the Scientific Basis of the Total Maximum Daily Load Approach to Water Pollution Reduction, 2001). One accepted watershed-scale model for identifying stressors of contaminants is the Soil Water Assessment Tool (SWAT). Models must produce reasonable results, especially when predicting TMDLs (Arnold et al., 1998). In order for a watershed scale model to produce reasonable results, it must first be calibrated using data collected from the watershed being modeled or a similar watershed. Data collected from research sites, along with any other applicable data sets that are available in the literature, can be used to develop land use specific parameters (calibrate) for the SWAT model. The literature indicates that SWAT has been used worldwide to model watersheds of varying sizes, from 3.3 km^2 to 598,538 km^2 (Bosch et al., 2004). However, there is limited literature available on the use of SWAT in TMDL development. There is also limited literature on the use of SWAT to predict sediment and P in small, single-land-use watersheds. These criteria are especially true for the Piedmont region of Georgia.

SWAT parameters need to be evaluated for sediment and P predictions in small, Piedmont watersheds where there is a single land use defined. One way to do this is to collect as much data as possible from these watersheds and compare them to model predictions. If model predictions for sediment are reasonable for small watersheds, then the sediment TMDL development process will become much more efficient. The first objective of this study was to determine sediment and P loads from four watersheds in the Georgia Piedmont. The second objective of this study was to assess SWAT for TMDL applications, based on its performance when modeling sediment and P, in two small, agricultural, Georgia Piedmont watersheds.

MATERIALS AND METHODS

Study Sites

Two agricultural watersheds, located in different counties, were evaluated in this study. The Springfield Creek watershed was located in Morgan County, Georgia. This watershed (295 ha) was used primarily for dairy cattle production and we will refer to it as the Dairy watershed. There are approximately 900 cattle on the watershed. Land covers within the watershed consist of 55% pasture, 28% forest, and 6% row crops (NRSAL, 1998). [1]Davidson, [2]Lloyd, and [3]Tocooa ([1]fine, kaolinitic, thermic Rhodic Kandiudults; [2]fine, kaolinitic, thermic Kanhapludults;

[3]coarse-loamy, mixed, active, nonacid, thermic Typic Udifluvents) soil series cover approximately 96% of the watershed (USDA, 2005a). The Rose Creek watershed was located in Oconee County, Georgia. This watershed (855 ha) was used primarily for cattle and poultry production. There are 16 poultry houses and approximately 500 cattle on the watershed. Land covers within the watershed consist of 67% pasture, 31% forest, and 2% row crops (NRSAL, 1998). [1]Appling, [1]Cecil, [2]Davidson, and [1]Pacolet ([1]fine, kaolinitic, thermic Typic Kanhapludults; [2]fine, kaolinitic, thermic Rhodic Kandiudults) soils series cover approximately 90% of the watershed (USDA, 2005b). Measurements were taken at road culverts for both agricultural sites. Monitoring at the outlet of each watershed began in 2003 and continued through the end of 2005.

Sample Collection for Observed Data

Automated ISCO 6700 water samplers (ISCO Inc, Lincoln, NE) were installed at each monitoring site to collect storm samples. The ISCO samplers were powered by 12-volt, deep-cycle marine batteries connected to solar panels. The ISCO samplers were programmed to take multiple discrete samples during a storm event. This was done using a pressure transducer installed vertically in the stream through a PVC pipe with holes drilled in it. The PVC pipe was used to protect the pressure transducer from debris and sediment being transported by the stream. The pressure transducer was connected to the ISCO sampler's control unit, which recorded the date, time, stream stage, and sample number (when applicable), every 5 min for the duration of the study. Sample collection was triggered by a predetermined stage height that was manually programmed into the ISCO sampler. Samples were picked up within 24 h of collection and stored in a refrigerator until analyzed. Base flow grab samples were collected weekly in addition to the storm flow samples. Rainfall was measured in each watershed with a tipping-bucket rain gauge connected to a data logger that was programmed to record precipitation amounts every 5 min.

Rating curves were developed at each site so that stream stage could be converted to flow. To construct the rating curves, culvert dimensions were measured at each site to determine the hydraulic radius of the stream. Stream velocity for a given stage height was calculated using the Manning equation with measured values of hydraulic radius and slope and an estimate of the Mannings roughness coefficient. Individual discharge rating curves were then created using FlowMaster (Haestad Methods, Waterbury, CT). Finally, manual flow measurements were taken at each site and the roughness coefficient was adjusted to get a more accurate flow measurement.

Sample Analysis

Base and storm flow samples collected from each site were analyzed for sediment and P concentrations. Suspended sediment concentrations (SSC) were determined by filtering a sample on pre-weighed 0.45µm filter, drying the filter at 106°C for 24 hours, and reweighing the filter (Guy, 1969). Dissolved reactive P (DRP) for filtered storm and base flow samples was determined colorimetrically with an RFA-300 Series Alpkem Analytical auto-analyzer (Murphy and Riley, 1962). Total P (TP) for unfiltered storm and base flow samples was determined by persulfate digestion and measured colorimetrically with the autoanalyzer (Bremner, 1996).

Rating Curves and Load Calculations

Flow-weighted daily average concentrations for each contaminant were used for comparing model predictions from the outlet of each watershed to our observed data. Flow-weighted average contaminant concentrations were determined by summing the product of concentration, flow, and change in time between samples and dividing by the sum of the product of flow and time for the sample represented by the following equation:

$$\overline{C_Q} = \frac{\sum (C_t * Q_t * \Delta t)}{\sum (Q_t * \Delta t)}$$

where: $\overline{C_Q}$ is average daily flow-weighted concentration, C_t is the contaminant concentration at the time the sample was taken, Q_t is the flow at the time the sample was taken, and Δt is the amount of time between samples for any given day.

SWAT Model Data

Several GIS datasets are required when setting up the SWAT model for a watershed. These data sets include land cover, elevation, soils, and streams. Each data set is essential because the model uses the data within each GIS layer as parameter values that can be modified to calibrate the model. All GIS data were projected into the Universal Transverse Mercator (UTM).

The land cover data set used for both agricultural sites was developed by the University of Georgia Natural Resource Spatial Analysis Laboratory. The land cover map was produced from Landsat imagery at 1:24,000 scale with a 30-m spatial resolution. The overall statewide accuracy for the land cover data is 85% and is based on videography, digital-ortho-quarter-quads (DOQQ), and other ground referencing information (NRSAL, 1998). The digital elevation model (DEM) data used for this project was developed by the USGS Earth Resources Observation and Science (EROS) Data Center. The National Elevation Datasets (NED) were downloaded for Morgan and Oconee counties at 1:24,000 scale with a spatial resolution of 30-m. According to USGS, the NED is a much-improved base of elevation data for calculating slope and hydrologic derivatives, due to efficient data processing methods that reduce errors and improve overall accuracy in the data (USGS, 1999). The NED meets DEM standards set by the USGS (USGS, 2005).

STATSGO soils data are the default soils data layer used with the SWAT model. However, because of the variability of soils over a landscape and the small watersheds being modeled in this study SSURGO soils data were used. The SSURGO database was developed and is maintained by the USDA Natural Resource Conservation Service (NRCS). This dataset is the most detailed level of soil geographic data developed by the National Cooperative Soil Survey. However, because digitizing soils data is a time intensive process, not all counties in Georgia have digital SSURGO data available. For this study, both spatial and tabular data were available for Morgan County, but only tabular data was available for Oconee County. Data for Morgan County was downloaded from the NRCS Soil Data Mart at a scale of 1:24,000 (USDA, 2005a). Tabular data for Oconee County was also downloaded from the NRCS Soil Data Mart (USDA, 2005b). However, USDA Soil Survey paper maps had to be digitized before the data could be incorporated into the SWAT model. The process for digitizing soil data is explained later.

Finally, the streams and rivers base-map (1:24,000 scale) GIS data sets were downloaded for both, Morgan and Oconee Counties. These maps were originally developed by the Georgia Department of Transportation (GDOT) for use as a cartographic tool for developing general highway maps for each county. The maps contain linear hydrographic features, such as, rivers, streams, and artificial flow paths through water bodies. These data meet National Map Accuracy Standards of less than 0.84 mm error for 1:24,000 scale data (GDOT, 1996; USGS, 2005).

Digitizing SSURGO Data

The soil survey map for Oconee County was developed by the USDA Soil Conservation Service and published in 1968, but has not been digitized (USDA, 1968). Therefore, an

ArcView GIS script, ImageWarp 2.0, was used to register and rectify the paper soil survey map for the watershed area so that a digital soil map could be created (ESRI, 2006). First, the paper soil survey map was scanned to make a digital copy. Next, photo-editing software was used to edit the scanned images into a single map that contained the watershed. Several road intersections were identified both on the soil survey map and georeferenced DOQQs and were selected as control points. The digital soil survey map was then registered to the UTM projection by selecting individual control points from the DOQQ and rectifying them to the corresponding point on the soil survey map. Five control points were selected so the positional accuracy, based on root mean squared error (RMSE), was less than 1-m. Finally, a vector GIS layer was created in ArcView GIS and polygons on the digitized soil survey maps were traced using the editing tool. Corresponding soil map-unit symbols were assigned to each polygon in the attribute table.

SSURGO Data Processing

In order for SSURGO soil data to be used by the SWAT model, it must be written into a database file that SWAT recognizes. This database file does not accompany the original SSURGO data; therefore, tabular SSURGO data must be joined with spatial GIS data before it can be used by the model. This was done with the SSURGO SWAT 2.x ArcView GIS script (Peschel et al, 2003). After processing the SSURGO data into the appropriate database format, the SSURGO spatial data layer was integrated into the SWAT model by joining the spatial data attribute table with the processed soil data for the watershed.

SWAT Calibration

Default model parameters are automatically set by SWAT when the model is set up for a watershed. The model may be calibrated by adjusting parameters related to flow, sediment, and

P. To calibrate the model, parameters governing flow, sediment and P were adjusted to obtain the best possible fit between simulated model output and observed data. This process, called parameterization, involves selecting or estimating model input parameters and comparing model outputs to an experimental dataset (Mulla and Addiscott, 1999). For our study watersheds, SWAT was calibrated manually for flow by adjusting groundwater and runoff parameters until an adequate fit was observed. The Nash-Sutcliffe model efficiency coefficient was used to determine the goodness-of-fit for our flow calibration. Sediment and P concentrations were also manually calibrated by parameterization and then compared to observed data, using the root mean squared error (RMSE) to determine the best fit between model predictions and observed data. Predicted sediment and P concentrations were calibrated to observed flow-weighted average daily sediment and P concentrations, respectively. Sub-hourly 15-min rainfall data were used for the model simulations and the Green Ampt infiltration method was selected. Once the model was calibrated, the parameters were compared to a calibrated SWAT model from the Etowah River Basin, a 2800-km^2 watershed located in North Georgia.

SWAT requires a "warm-up" period in order to equilibrate necessary pools (i.e. groundwater and soil storage) before calibration. We allowed the model to "warm-up" for 2-yr (2001-2002). After the "warm-up" period, calibration for both watersheds was completed with data collected from 01 January 2003 to 31 December 2003.

SWAT Soil P Parameter Estimation

Several SWAT phosphorus parameters can be easily estimated from soil test data and the SSURGO soil database. These parameters include initial labile P concentration in the surface soil layer (SOL_SOLP), initial organic P concentration in the surface soil layer (SOL_ORGP), phosphorus sorption coefficient (PSP), and phosphorus soil partitioning coefficient (PHOSKD).

SOL_SOLP, the initial labile (soluble) P concentration, is best represented by anion exchange resin extractable P, but is not directly measured by soil test laboratories (Sharpley et al., 1984). However, labile P concentration is linearly related to other P extractions methods such as Mehlich-1 (M1), Bray, and Olsen extraction methods, which are more commonly used by soil test laboratories in the United States (Sims, 2000; Bray and Kurtz, 1945; Olsen et al., 1954). The equation for "highly weathered soils" (applicable to Ultisols in the Georgia Piedmont) is:

$$P_{il} = 0.24 \times DP + 2.9$$ **Eqn. 3.1**

where P_{il} is labile P (mg kg^{-1}) and *DP* is double acid (Mehlich-1) extractable P (Sharpley et al., 1984). However, because the relationship between P_{il} and DP is among soils, we developed a relationship more specific to the Piedmont region. Mehlich-1 and anion exchange resin P (AER P) were measured on soil samples collected in the Etowah Watershed (Northwest Georgia) since soils in the region are more representative of those found in our modeled watersheds. Then, the results were plotted (AER P vs. Mehlich-1 P) to get the following equation:

$$P_{il} = 0.570 \times M1 + 7.41$$ **Eqn. 3.2**

where P_{il} is labile P and *M1* is Mehlich-1 P. SOL_LABP concentrations were calculated by applying our equation from the Georgia Piedmont to soil test phosphorus (Mehlich-1 P) data from soil samples that were collected in each study watershed (Table 3.1).

A second regression equation derived by Sharpley et al. (1984) was used to obtain values for SOL_ORGP. This equation approximates organic P in the soil based on the total N content such that:

$$P_o = 44.4 + 1130 \times N_t$$ **Eqn. 3.3**

where P_o is total organic P (mg kg^{-1}) and N_t is percentage of total nitrogen. We assumed a C:N ratio of 11:1 and estimated the total N content of the soil from organic C content in the SSURGO database. Assuming that the soil organic P in the humic pool (SOL_ORGP) was approximately the same as the total organic P in the soil (P_o) we obtained values for SOL_ORGP for each soil present in the watershed (Table 3.2).

The phosphorus soil partitioning coefficient (PHOSKD) is the ratio of labile P concentration in the soil to soluble P concentration in runoff. Schroeder et al. (2004) described dissolved reactive P (DRP) from runoff in typical Piedmont soils as a function of Mehlich-3 P (M3) in the soil with the following equation:

$$DRP = 0.0017 \times M3 + 0.15 \qquad \textbf{Eqn. 3.4}$$

where *DRP* is the DRP concentration (mg L^{-1}) and *M3* is the Mehlich-3 P concentration in the soil (mg kg^{-1}). To find the relationship between DRP in runoff and labile P in soils, the relationship between DRP and M3 must be converted to a relationship between DRP and labile P (AER P). Shuman et al. (1988) showed that, for a typical Piedmont soil, the relationship between M1 and Mehlich-2 (M2) extracted P was:

$$M1 = 0.72 \times M2 - 1.71 \qquad \textbf{Eqn. 3.5}$$

where *M1* and *M2* are the Mehlich-1 P and Mehlich 2 P concentrations in the soil. Assuming that M3 can be approximated by M2 in the previous equation, we can substitute M3 for M2 in to describe the relationship between M1 and M3. Finally, the following equation is obtained by substitution using the relationships between M1 and labile P, M1 and M3, and M3 and DRP:

$$DRP = 0.00414 \times P_{il} + 0.123 \qquad \textbf{Eqn. 3.6}$$

where *DRP* is the DRP concentration (mg L^{-1}); and P_{il} is labile P (mg kg^{-1}). Therefore, PHOSKD, obtained by taking the reciprocal of the slope term (0.00414 kg L^{-1}), is 242 m^3 Mg^{-1}.

Lastly, the phosphorus sorption coefficient (PSP) can be calculated using a regression equation developed by Sharpley et al. (1984). Given soil labile phosphorus concentration, clay content, and organic matter content the following equation can be applied:

$$PSP = 0.39 - 0.047 \times \ln CL - 0.053 \times OC + 0.0045 \times LP \qquad \textbf{Eqn. 3.7}$$

where *CL* is the percent clay in the soil; *OC* is the percent organic carbon in the soil; and *LP* is the area-weighted mean of the labile P concentrations of different soils in the respective watershed (mg kg^{-1}). PSP parameter values were 0.54 and 0.45 for the Dairy and Cattle and Poultry watersheds, respectively.

Results and Discussion

Flow Calibration

Obtaining a good flow calibration is essential to producing accurate model predictions of contaminants. Table 3.1 lists parameters that were adjusted to calibrate flow in each watershed. Several parameters (i.e. SURLAG, ALPHA_BF, GWQMN, REVAP_MN, SOL_AWC, and CH_K1) took on different values between the two small agricultural watersheds, as well as from the much larger Etowah watershed. Other parameters (i.e. GW_REVAP, RCHRG_DP, CN2, and SOL_K) were similar between two of the three watersheds, but no parameter was similar for a single parameter in all three. In some cases, a parameter was changed on two watershed calibrations but no change was required in the third (CH_N2, SOL_K, and CH_N1). The flow calibration of the Dairy watershed required that the SURLAG parameter be adjusted outside of the SWAT default boundaries. In small watersheds the time of concentration is usually less than one day, requiring that the SURLAG parameter be minimal to account for the time it takes for runoff to enter a stream and exit the basin. The Nash-Sutcliffe (N-S) model efficiency coefficient is expected to be between 0-1, where a value of 1 would be the best fit between observed and simulated values. In this case the N-S coefficients were -0.003 and 0.083 for the Dairy and Cattle and Poultry watersheds, indicating that the goodness-of-fit for predicted versus observed flow was not very good for the two watersheds. Figures 3.1 and 3.2 show the goodness-of-fit of the model calibrations for Dairy and Cattle and Poultry, respectively. Variability of parameter values among the three watersheds was evident, indicating that model calibration for flow was quite dependant upon the size, land cover, and soils of each watershed.

SWAT Sediment Calibration

SWAT simulates sediment (SSC) from two sources: loadings from HRU's/subbasins and channel degradation/deposition (SWAT Manual). Calibrating SWAT to correctly simulate flow provides a good approximation of sediment loading that must be then calibrated by adjusting USLE, sediment routing (SPCON and SPEXP) parameters, and the channel cover/erodibility factors (CH_COV and CH_EROD). Table 3.2 lists parameters that were adjusted to calibrate SWAT for sediment loading from the three watersheds. As with the flow parameters, the sediment parameter values vary greatly between the watersheds. The CH_COV and CH_EROD parameters were different between all three watersheds. Two USLE parameters were adjusted in the Dairy watershed but not changed in the Cattle and Poultry or Etowah watersheds. The SPCON parameter was much higher in the Dairy watershed (0.009) than in the other two watersheds but within the default parameter bounds. In the Cattle and Poultry and Etowah watersheds SPCON was similar but outside of the bounds, 0.0004 and 0.0001 respectively. LAT_SED (sediment concentration in lateral and groundwater flow) was calibrated to a value of

175 mg L^{-1} in the Dairy watershed but not adjusted in the other two watersheds, indicating that lateral and groundwater flow may have had an effect on sediment in the smaller watershed. Cattle have access to several streams segments in the Dairy watershed, which could have also caused more SSC variability in the stream. Figures 3.3 and 3.4 show the simulated versus observed model data from the calibrated SWAT model for each watershed. Root mean squared error for sediment was 672 and 22 for Dairy and Cattle and Poultry, respectively. Note that sediment concentration ranged from 0-4000 mg L^{-1} in Dairy, which may indicate that channel degradation also significantly influenced sediment concentration in the stream. Table 3.3 lists sediment loads calculated from the SWAT calibration and sediment loads observed from the two agricultural watersheds. Error between predicted and observed sediment loads was very high at Dairy (367%) and low at Cattle and Poultry (8%). Poor relationships between predicted and observed Dairy stream during both flow and sediment calibrations were likely the cause of such a large percent error in the sediment load prediction.

SWAT P Calibration

SWAT predicts organic P and mineral P concentrations for each reach in the watershed but not total P concentration. To calculate total P, organic P and mineral P concentrations must be summed for each day and then divided by daily flow. The SWAT sediment calibration discussed previously provides a good starting point for the P calibration because it accounts for P adsorbed to sediment particles. Parameters related to soil and stream P generation, transport, and transformation processes were adjusted to calibrate for total P (Table 3.6). Fertilizer applications of broiler litter are common in the Georgia Piedmont so we assumed that each watershed received 2000 kg ha^{-1} yr^{-1}, which is a typical rate for the region.

Several parameter values were the same for all three watersheds (PHOSKD, GWSLP, RS5, BC4, and AI2), while other parameters varied (PSP, PPERCO, RS2, MUMAX, and RHOQ). The model was very sensitive to the RS2 parameter and it was necessary to adjust the value outside of the default boundary to calibrate both the Dairy (RS2 = 20) and Cattle and Poultry (RS2 = 5) watersheds, while the RS2 parameter value for the Etowah watershed remained at the lower bound (0.001). Figures 3.5 and 3.6 show the simulated versus observed model data from the calibrated SWAT model for each watershed. Observed TP ranged from 0-2.5 mg L^{-1} at Dairy and 0-0.75 mg L^{-1} at Cattle and Poultry. Root mean squared error for TP was 0.65 and 0.21 mg L^{-1} for Dairy and Cattle and Poultry, respectively. Table 3.7 lists TP loads calculated from the SWAT calibration and TP loads observed from each watershed. Error between predicted and observed TP loads was lower at Dairy (47%) than at Cattle and Poultry (70%). The large TP concentrations at Dairy likely resulted from a combination of high animal density and over application of manure in the watershed. Underestimation of the TP load at Dairy may also be attributed to the high animal density in the watershed that SWAT was not able to account for. SWAT overestimated the TP load at Cattle and Poultry which indicates that the broiler litter application we simulated may have been too high.

Summary and Conclusions

In general, SWAT calibrations for flow in both watersheds were not good. Model predictions for storm flow in the small watersheds did not correlate well with measured flow. In most cases SWAT only predicted runoff for large storm events and did not generate enough runoff to show an increase in flow in the predicted hydrograph in instances where there were small storms. The flow calibration was slightly better in the larger Cattle and Poultry watershed, indicating that SWAT may predict flow better in larger watersheds. Sediment concentrations in the Dairy watershed were much larger than at the Cattle and Poultry watershed, which can be attributed to a much larger animal density at Dairy. The large animal density likely caused greater variability in sediment concentrations in the watershed, along with an already unsatisfactory flow calibration, made it much more difficult to achieve a good model prediction for sediment. The SWAT sediment concentration at the Cattle and Poultry watershed was good because animal densities were smaller and the flow calibration was slightly better. The P calibration for the Cattle and Poultry watershed was slightly better than at the Dairy. However SWAT underestimated TP loads at Dairy and overestimated TP loads at Cattle and Poultry. Larger animal density and greater manure applications could have caused observed TP loads to be larger at Dairy, while smaller animal density and less manure application could have also resulted in lower observed TP concentrations in the Cattle and Poultry watershed. Model parameters between the two small agricultural watersheds and the large multiple land use watershed were variable, indicating that it may not be feasible to estimate parameters for a multiple land use watershed base on the primary land use of a subbasin or hydrologic response unit. Parameter values must be calibrated for each watershed being modeled in order to produce a good model prediction of flow and contaminants in larger watersheds. Once model calibrations are acceptable they may be used for assessment of land management impacts on flow, sediment and P.

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Table 3.1. Initial SOL_LABP values for SWAT subbasins.

Watershed	SWAT Subbasin	Area Weighted Avg. DP† (lbs $acre^{-1}$)	DP (mg kg^{-1})	SOL_LABP‡ (mg kg^{-1})
Dairy	1	266	133	83
	2	68	34	27
	3	348	174	107
	4	2468	123	77
	5	148	74	50
	6	8	42	31
	7	252	126	79
	8	463	232	139
	9	231	116	73
	10	99	49	36
	11	469	234	141
Cattle and Poultry	1	226	113	72
	2	220	110	70
	3	253	127	80
	4	229	114	73
	5	214	107	69
	6	170	85	56
	7	164	82	54
	8	174	87	57
	9	54	27	23
	10	28	14	15
	11	203	102	65
	12	110	55	39
	13	110	55	39
	14	109	55	39
	15	111	56	39
	16	140	70	47
	17	36	18	18
	18	198	99	64
	19	65	33	26

† Double Acid P
‡ SWAT Soil Labile P parameter

Table 3.2. Initial SOL_ORGP values for SWAT soils.

Watershed	Soil Series	Organic Matter Low (%)	Organic Matter High (%)	Organic Carbon (%)*	Total N (%)	SOL_ORGP† (mg kg^{-1})
Dairy	Altavista-Wickham 2-6% slopes	0.5	2	0.7	0.07	119
	Cecil (scl) 2-6% slopes	0.5	1	0.4	0.04	89
	Chewacla (sl) 0-2 % slopes	1	4	1.5	0.13	194
	Davidson (l) 2-6% slopes	0.5	2	0.7	0.07	119
	Davidson (cl) 2-6% slopes	0.5	2	0.7	0.07	119
	Lloyd (sl) 2-6% slopes	0.5	2	0.7	0.07	119
	Lloyd (sl) 6-15% slopes	0.5	2	0.7	0.07	119
	Lloyd (cl) 2-6% slopes	0.5	2	0.7	0.07	119
	Lloyd (cl)6-15% slopes	0.5	2	0.7	0.07	119
	Toccoa (ls) 0-3% slopes	1	2	0.9	0.08	134
Cattle and Poultry	Pacolet (sl) 15-25% slopes	0.5	2	0.7	0.07	119
	Pacolet (scl) 6-10% slopes	0.5	1	0.4	0.04	89
	Pacolet (scl) 10-15% slopes	0.5	1	0.4	0.04	89
	Pacolet-Gullied land complex 10-15 % slopes	0.5	1	0.4	0.04	89
	Pacolet-Gullied land complex 10-25% slopes	0.5	1	0.4	0.04	89
	Wehadkee and alluvial land	2	5	2.0	0.18	253
	Appling (scl) 6-10% slopes	0.5	3	1.0	0.09	149
	Appling (sl) 2-6% slopes	0.5	3	1.0	0.09	149
	Appling (sl) 6-10% slopes	0.5	3	1.0	0.09	149
	Cecil (sl) 2-6% slopes	0.5	1	0.4	0.04	89
	Cecil (sl) 6-10% slopes	0.5	1	0.4	0.04	89
	Cecil (scl) 2-6% slopes	0.5	1	0.4	0.04	89
	Colfax (sl) 2-6% slopes	0.5	2	0.7	0.07	119
	Congaree and alluvial land	1	4	1.5	0.13	194
	Chewacla and alluvial land	1	4	1.5	0.13	194
	Davidson (cl) 6-10% slopes	0.5	2	0.7	0.07	119
	Davidson (cl) 10-15% slopes	1	3	1.2	0.11	164
	Davidson (sl) 2-6% slopes	0.5	2	0.7	0.07	119

*Organic C content was obtained by dividing the average organic matter from SSURGO by 1.72
† SWAT Soil Organic P parameter

Table 3.3. Flow related parameters in SWAT.

Parameter Name	Definition (unit)	SWAT Default Value	Bounds	Dairy Final Value	Cattle and Poultry Final Value	Etowah Final Value	SWAT File
SURLAG	Surface runoff lag time (days)	4	1-24	0.125	6.503	1	.bsn
ALPHA_BF	Base flow alpha factor (days)	0.048	0-1	0	0.280	0.014	.gw
GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur (mm)	0	0-5000	175.1	914.4	0	.gw
REVAP_MN	Threshold depth of water in the shallow aquifer for "revap" to occur (mm)	1	0-500	4.074	50.22	186	.gw
GW_REVAP	Groundwater "revap" coefficient (unitless)	0.02	0.02-0.20	0.0334	0.201	0.02	.gw
RCHRG_DP	Deep aquifer percolation fraction (fraction)	0.05	0-1	0.6	0.2	0.19	.gw
GW_DELAY	Groundwater delay (days)	31	0-500	99.81	99.98	38	.gw
CANMX	Maximum canopy storage (mm)	0	0-100	5.763	2.244	-0.41	.hru
ESCO	Soil evaporation compensation factor (unitless)	0	0-1	0.1036	0.9402	1	.hru
OV_N	Manning's "n" value for overland flow (unitless)	0.15	0.01-30	0.01	0.15	0.2	.hru
SLSOIL	Slope length for lateral subsurface flow (m)	0	0-150	150	0	0	.hru
CN2*	SCS runoff curve number for moisture condition II (unitless)	Varies	35-98	-14.1	+3.316	-11	.mgt
CH_N2	Manning's "n" value for the main channel (unitless)	0.014	-0.01-0.3	0.014	0.014	0.02	.rte
SOL_K	Saturated hydraulic conductivity (mm hr^{-1})	Varies	0-2000	0.1993	No Change	No Change	.sol
SOL_AWC*	Available water capacity of the soil layer (mm mm^{-1})	Varies	0-1	+0.043	-0.006	No Change	.sol
CH_K1	Effective hydraulic conductivityin tributary channel alluvium (mm hr^{-1})	0	0-150	0	150	41.1	.sub
CH_N1	Manning's "n" value for the tributary channel (unitless)	0.014	0.01-30	0.014	0.014	0.1	.sub

* denotes that value was added or subtracted from the SWAT default value

Table 3.4. Sediment related parameters in SWAT.

Parameter Name	Definition (unit)	SWAT Default Value	Bounds	Dairy Final Value	Cattle and Poultry Final Value	Etowah Final Value	SWAT File
APM	Peak rate adjustment factor for tributary sediment routing in the subbasin (unitless)	1	0.5-2	1	1	1.8	.bsn
PRF	Peak rate adjustment factor for sediment routing in the main channel (unitless)	1	0-2	1	1	1.8	.bsn
SPCON	Linear parameter for calculating the maximum amount of sediment that can be reentrained during channel sediment routing (Deg)	0.001	0.001-0.10	0.009	0.0004	0.0001	.bsn
SPEXP	Exponent parameter for calculating sediment reentrained in channel sediment routing (unitless)	1.5	1-1.5	1.5	1.9	1.5	.bsn
USLE_C	Minimum value of USLE C factor for water erosion applicable to the land cover/plant (unitless)	Varies	0.001-0.5	-0.125	No Change	No Change	crop.dat
LAT_SED	Sediment concentration in lateral flow and groundwater flow (mg L^{-1})	0	0-5000	175	0	0	.hru
USLE_P	USLE equation support practice (P) factor (unitless)	1	0.1-1	2.1	1	1	.mgt
CH_COV	Channel cover factor (unitless)	0	-0.001-1	0.001	0.015	0.005	.rte
CH_EROD	Channel erodibility factor (unitless)	0	-0.05-0.6	0.05	0.3	0.005	.rte

* denotes value was added or subtracted from the default SWAT value

Table 3.5. Observed vs. simulated sediment loads.

Watershed	Year	Watershed Area (ha)	Observed Sediment Load (kg ha^{-1} yr^{-1})	SWAT Simulated Sediment Load (kg ha^{-1} yr^{-1})	Error (%)
Dairy	2003	295	269	1259	367
Cattle and Poultry	2003	855	455	420	8

Table 3.6. P related parameters in SWAT.

Parameter Name	Definition (unit)	SWAT Default Value	Bounds	Dairy Final Value	Cattle and Poultry Final Value	Etowah Final Value	SWAT File
PHOSKD	Phosphorus soil partitioning coefficient (m^3 Mg^{-1})	175	50-200	242	242	242	.bsn
PSP	Phosphorus sorption coefficient (unitless)	0.4	0.01-0.7	0.54	0.45	0.31	.bsn
PPERCO	Phosphorus percolation coefficient (10 m^3 Mg^{-1})	10	10-17.5	10	10	15	.bsn
SOL_LABP	Initial labile P concentration in surface soil layer (mg L-1)	0	0-100	Varies	Varies	Varies	.chm
SOL_ORGP	Initial organic P concentration in surface soil layer (mg L-1)	0	0-4000	Varies	Varies	Varies	.chm
GWSLP	Concentration of soluble phosphorus in groundwater contribution to stream flow from subbasin (mg L^{-1})	0	0-1000	0	0	0	.gw
RS2	Benthic sediment source rate for dissolved phosphorus in the reach at 20 °C (day^{-1})	0.05	0.001-0.1	20	5	0.001	.swq
RS5	Organic phosphorus settling rate in the reach at 20 °C ($day-^{1}$)	0.05	0.001-0.1	0.1	0.1	0.1	.swq
BC4	Rate constant for mineralization of organic P to dissolved P in the reach at 20 °C (day^{-1})	0.35	0.01-0.70	0.01	0.01	0.01	.swq
MUMAX	Maximum specific algal growth rate at 20 °C (day^{-1})	2.0	1.0-3.0	2.45	2.9	2.9	.wwq
RHOQ	Algal respiration rate at 20 °C (day^{-1})	0.3	0.05-0.50	0.175	0.05	0.05	.wwq
AI2	Fraction of algal biomass that is phosphorus	0.015	0.01-0.02	0.013	0.01	0.01	.wwq

Table 3.7. Observed vs. simulated total P loads.

Watershed	Year	Watershed Area (ha)	Observed Total P Load (kg ha^{-1} yr^{-1})	SWAT Simulated Total P Load (kg ha^{-1} yr^{-1})	Error (%)
Dairy	2003	295	1.38	0.724	47
Cattle and Poultry	2003	855	0.233	0.396	70

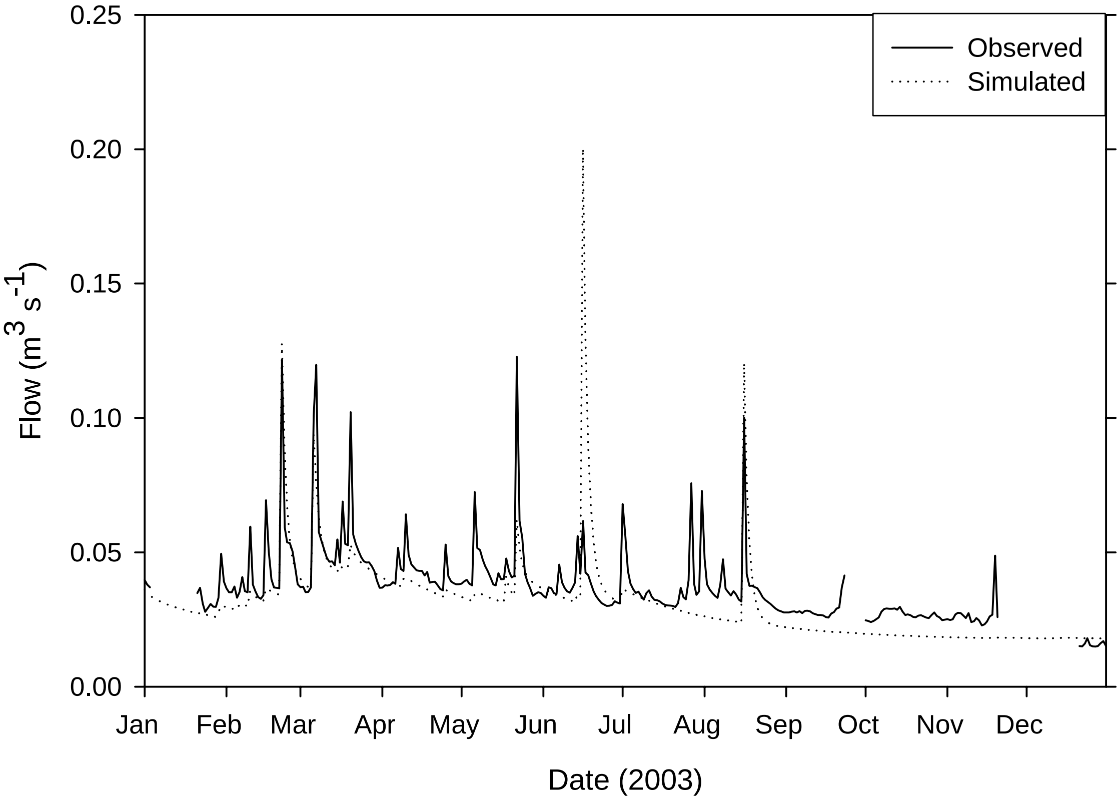

Fig. 3.1. Dairy SWAT flow calibration. The Nash-Sutcliffe model efficiency coefficient for this calibration was -0.003.

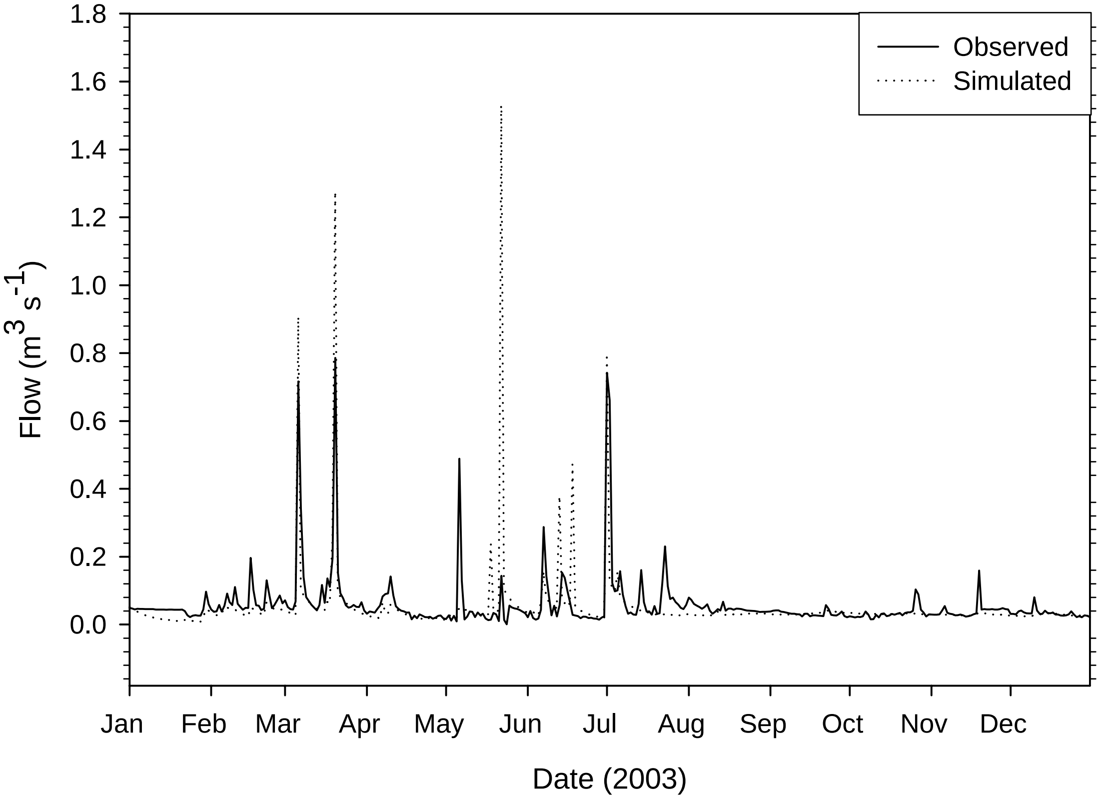

Fig. 3.2. Cattle and Poultry SWAT flow calibration. The Nash-Sutcliffe model efficiency coefficient for this calibration was 0.083.

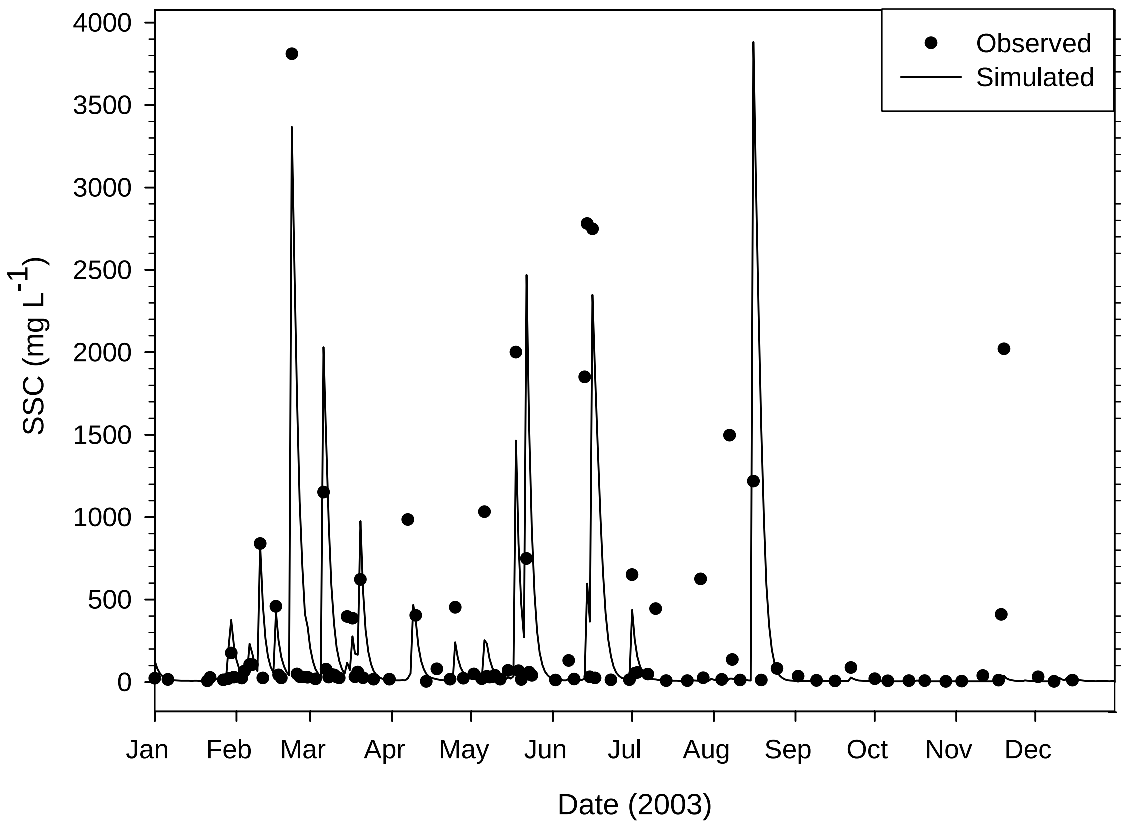

Fig. 3.3. Dairy SWAT sediment calibration (RMSE = 672).

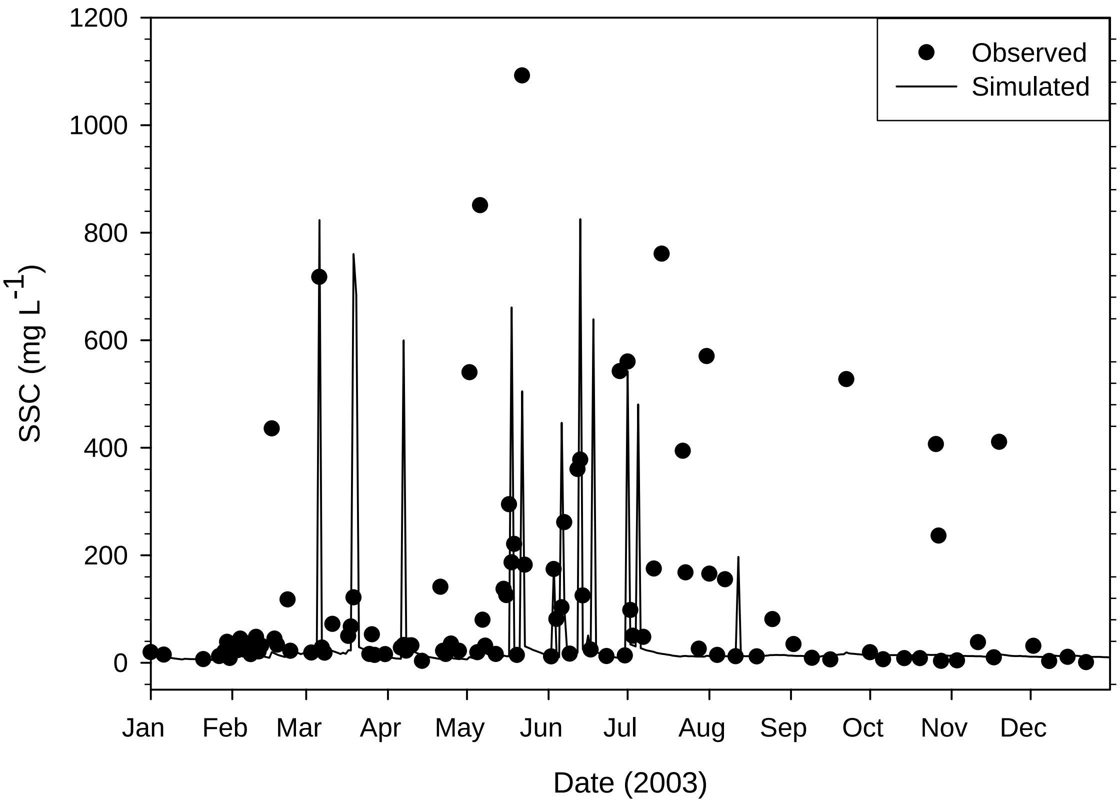

Fig. 3.4. Cattle and Poultry SWAT sediment calibration (RMSE = 22).

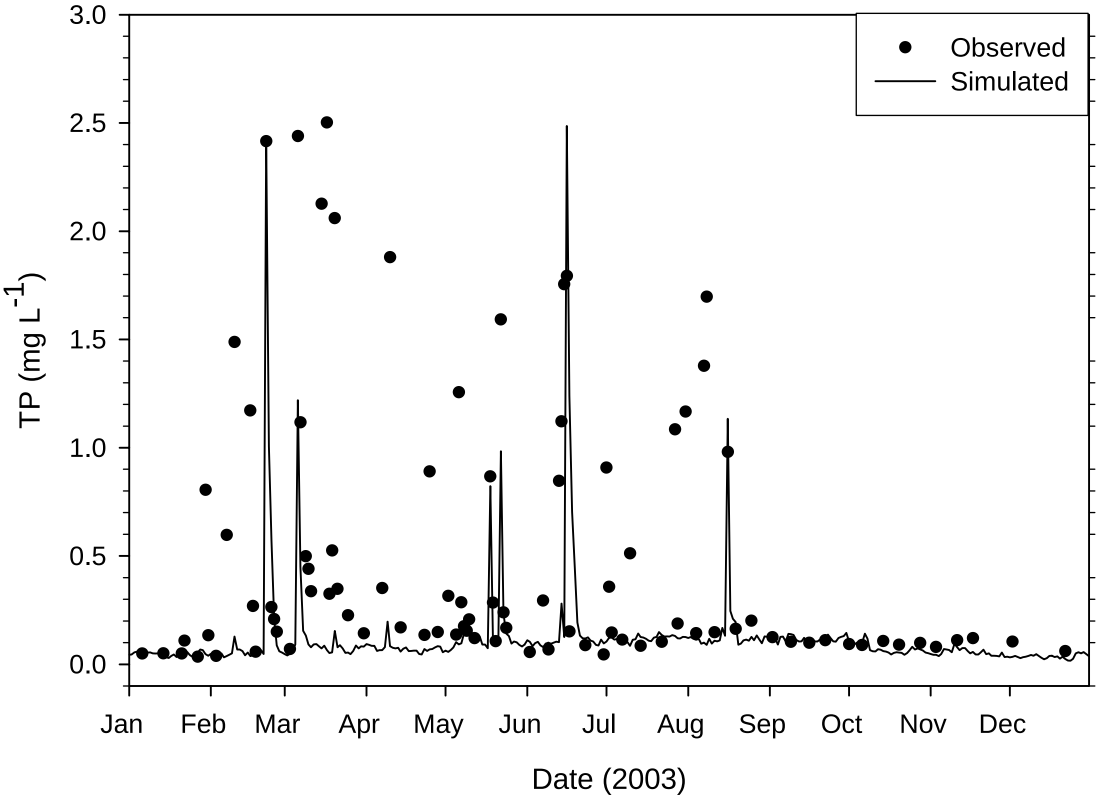

Fig. 3.5. Dairy SWAT phosphorus calibration (RMSE = 0.65 mg L^{-1}).

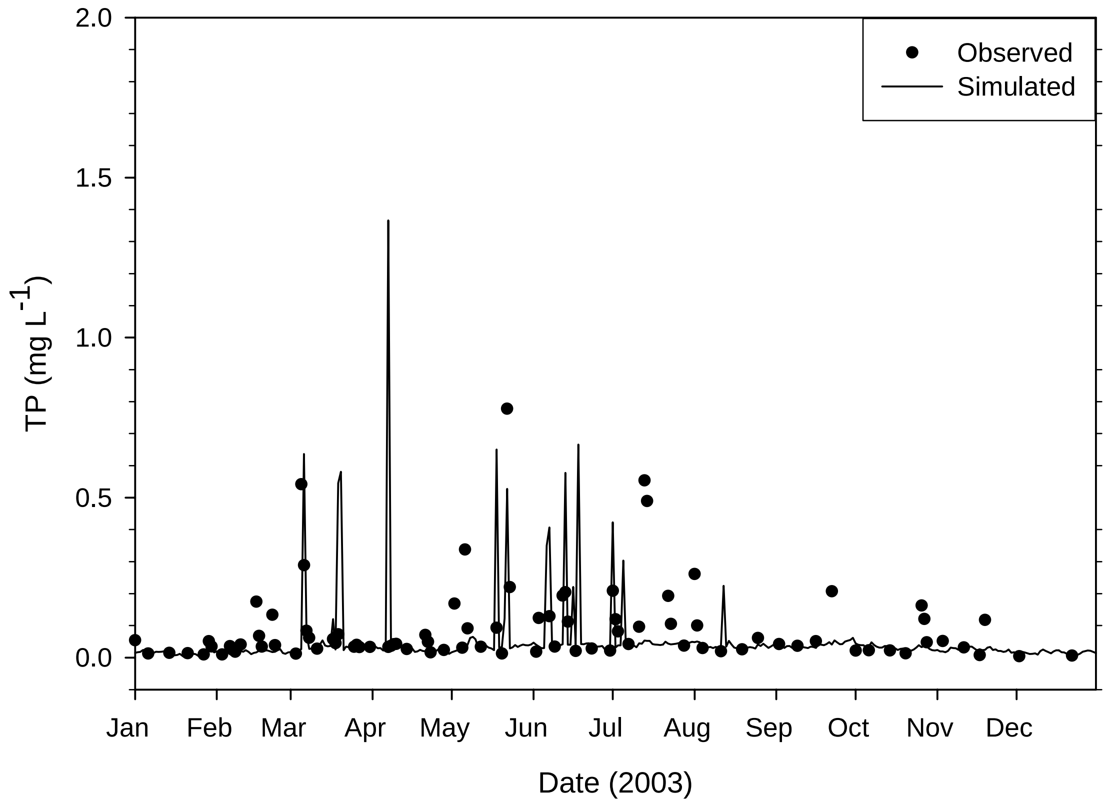

Fig. 3.6. Cattle and Poultry SWAT phosphorus calibration (RMSE = 0.21 mg L^{-1}).

CHAPTER 4

CONCLUSIONS AND IMPLICATIONS

In general, suspended sediment loads from the agricultural watersheds were similar to loads observed in the non-harvested watershed. Overall mean sediment loads from the forested watersheds were higher due to sedimentation occurring in the harvested watershed. Reduced stream bank stability and higher sediment concentrations in overland flow in the harvested watershed likely caused mean sediment loads for the forested land use to be higher. Mean sediment load from the mixed agricultural watershed (Cattle and Poultry) was similar to the loads observed in the non-harvested watershed, except in the last year (2005). Construction of a subdivision began on the Cattle and Poultry watershed in this year. Significant differences in SSC rating curve slopes for all three harvest periods from the treatment forest watershed indicate a timbering affect on sediment concentration and that current BMP's may not be adequately reducing runoff under silvicultural practices.

Although phosphorus loads were not significantly different between agricultural and forested land uses, phosphorus loads were much higher at the Dairy watershed. High animal occupancy and the application of manure on the watershed greatly contributed to the amount of phosphorus loading, as indicated by the high fractions of organic/particulate phosphorus in the agricultural land use. High organic/particulate P fractions in the forested land use were likely the result of stream/wildlife interactions.

In general, SWAT calibrations for flow in both watersheds were not good. Model predictions for storm flow in the small watersheds did not correlate well with measured flow. In most cases SWAT only predicted runoff for large storm events and did not generate enough runoff to show an increase in flow in the predicted hydrograph in instances where there were small storms. The flow calibration was slightly better in the larger Cattle and Poultry watershed, indicating that SWAT may predict flow better in larger watersheds. Sediment concentrations in the Dairy watershed were much higher than at the Cattle and Poultry watershed, which can be attributed a much higher animal density at Dairy. The high animal density likely caused greater variability in sediment concentrations in the watershed, along with an already unsatisfactory flow calibration, made it much more difficult to achieve a good model prediction for sediment. The SWAT estimate of sediment concentration at the Cattle and Poultry watershed was good because animal densities were lower and the flow calibration was slightly better. The phosphorus calibration for the Dairy watershed was slightly better than at Cattle and Poultry. Observed TP concentrations at the Dairy were high and SWAT overestimation of sediment at Dairy likely produced a better estimation of TP by predicting a higher amount of P adsorbed to sediment. SWAT over predicted TP in the Cattle and Poultry watershed, likely for the same reason. Too much P was predicted because of the overestimation of sediment. Lower animal density and less manure application could have also resulted in lower TP concentrations in the Cattle and Poultry watershed. Model parameters between the two small agricultural watersheds and the large multiple land use watershed were variable, indicating that it may not be feasible to estimate parameters for a multiple land use watershed base on the primary land use of a subbasin or hydrologic response unit. Parameter values must be calibrated for each watershed being modeled in order to produce a good model prediction of flow and contaminants in larger watersheds.

Sediment loads were very similar between the two agricultural streams and the non-harvested reference forest stream indicating that agricultural BMP's (i.e. riparian buffer zones) may be effectively removing sediment from runoff. However, even when BMP's were used in forest harvest practices, higher suspended sediment loads than the control forest watershed were observed. Comparing phosphorus loads between reference streams and agricultural land use proved to be an excellent way to determine what background P concentrations should be. It was obvious that phosphorus levels in the agricultural land uses were much too high and better BMP's should be implemented to reduce P loading, especially in watersheds with high animal densities.

The SWAT model did not seem to perform well when calibrated to data collected from the two agricultural watersheds making it impossible to evaluate parameter values for a specific land use. Flow, sediment, and phosphorus were extremely variable in the agricultural land use making it difficult to achieve good calibrations for each. Both watersheds were relatively small and could have also been a factor for producing good calibrations. Large variability in annual fluctuations of flow and contaminants (sediment and phosphorus) are common, thus evaluating SWAT over a longer period may prove to be more effective for determining the relationship between model parameters and land use.