MONITORING, MODELING, AND FINGERPRINTING SUSPENDED SEDIMENT IN A SOUTHERN PIEDMONT STREAM

by

RAJITH MUKUNDAN

(Under the Direction of David E. Radcliffe)

ABSTRACT

Sediment is one of the most important non-point source pollutant impairing water bodies in the United States and around the world. The objective of this research was to develop a new approach to watersheds with high sediment loads in the southern Piedmont incorporating geomorphic analysis of fluvial systems and to determine suspended sediment sources and their relative contributions using a fingerprinting approach. The GIS based SWAT model was used to simulate suspended sediment transport in the study watershed and to test the effect of spatial resolution of soil data on model predictions for flow and sediment. Results showed that spatial resolution of soil data did not have a significant effect on the model predictions for flow and suspended sediment in this Piedmont watershed. The suspended sediment load estimates for the study watershed was high and comparable with streams in the Piedmont region with unstable channels. Geomorphic analysis indicated that mass wasting and fluvial erosion were the dominant erosion processes in the stream channels. Sediment fingerprinting showed that eroding stream banks contributed about 60% of the total suspended sediment load followed by construction sites, unpaved roads and road ditches that contributed about 23-30%.

Pastures contributed about 10-15% suspended sediment in this watershed. The SWAT model also indicated stream channels as the primary source of suspended sediment. The relative source contribution of suspended sediment predicted by the SWAT model was comparable with the results of the fingerprinting study.

INDEX WORDS: Geomorphic Assessment, Monitoring, Modeling, Suspended Sediment, Channel Erosion, Sediment Fingerprinting, Water Quality

MONITORING, MODELING, AND FINGERPRINTING SUSPENDED SEDIMENT IN A SOUTHERN PIEDMONT STREAM

by

RAJITH MUKUNDAN

B.Sc., Kerala Agricultural University, India, 2001

M.Sc., Tamil Nadu Agricultural University, India, 2003

A Dissertation Submitted to the Graduate Faculty of The University of Georgia in Partial Fulfillment of the Requirements for the Degree

DOCTOR OF PHILOSOPHY

ATHENS, GEORGIA

2009

MONITORING, MODELING, AND FINGERPRINTING SUSPENDED SEDIMENT IN A SOUTHERN PIEDMONT STREAM

by

RAJITH MUKUNDAN

Major Professor: David E. Radcliffe

Committee: William P. Miller
Todd C. Rasmussen
L. Mark Risse
Xiaobai Yao

Electronic Version Approved:

Maureen Grasso
Dean of the Graduate School
The University of Georgia
August 2009

DEDICATION

This dissertation is dedicated to my grandfather Mr. S.L. Narayana Pillai and all my family members for giving me the best in life

ACKNOWLEDGEMENTS

I express my sincere gratitude to my Major Professor Dr. David Radcliffe for providing me a wonderful opportunity to work under his able guidance. The encouragement by Dr. Radcliffe has been the greatest asset to my pursuit of an interesting project. I thank him for nurturing my interest, guiding me, motivating me, and showing faith in me throughout my study.

The constructive comments and critical suggestions provided by members of my advisory committee helped to improve the manuscripts. My sincere thanks are due to my committee members for their time and effort. I acknowledge the list of people who helped me during field and lab work at various stages of my research. The name of Robert McKinley deserves a special mention for his hard work and dedication during soil and sediment sampling and lab processing.

I thank Dr. Miguel Cabrera, graduate coordinator for his help and concern during the research program. The help and support rendered by Vivienne Sturgill and all past and present office staff at the Crop and Soil Sciences Department was excellent.

I am grateful to the Rema's, John and Molly for their care and concern and being my family in Athens, Georgia. Finally, I would like to express my gratitude to all my family members in India, my wife Deepa and my baby boy Suraj for their constant encouragement during this research endeavor.

TABLE OF CONTENTS

APPENDICES

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION AND LITERATURE REVIEW

Environmental degradation is occurring over large areas of the world. Conventional methods of resource management and land husbandry can no longer handle this ever-increasing problem. In this context, an alternate approach adopting the best possible technology is needed.

Many of the stream segments in the United States do not meet the water quality standards set forth by the EPA; 17% of the streams have excessive sediment loads (USEPA, 2006). Sediments are loose particles of sand, clay, silt, and organic material that originate from eroding soil and from decomposing plants and animals. Wind, water, and ice carry these particles great distances. Suspended sediment consists of that portion of the total sediment load of rivers that is carried in the water column (USEPA, 2008). This includes the portion of the total sediment load not represented in the bed material and consists of clay and silt fractions that are controlled more by supply than energy. Studies show that excessive suspended sediment loading in streams adversely affects aquatic life by lowering the diversity of macroinvertebrate communities, reducing fish growth, increasing the load of harmful bacteria and heavy metals.

For most of the sediment impaired streams in the southern Piedmont and other regions of the United States, it is not clear if the reason for the high levels of sediment is due to current upland erosion sources (e.g., agricultural fields, roads, ditches, construction sites) or bank erosion of *legacy sediment* deposited in the flood plains during the period of intensive cotton farming from about 1830 to 1930 (Trimble, 1974). This distinction is important because Total Maximum Daily Load (TMDL) implementation plans must identify the sources of erosion. A comprehensive study of these waters at a basin scale is required to get a complete picture of the processes taking place. Managing sediment without identifying its primary source may not be cost effective or efficient. According to the latest reports (USEPA, 2006), 45% of the assessed rivers and streams in the United States do not meet the stated water quality standards even after more than three decades of control efforts.

Under the TMDL program originated from Section 303(d) of the 1972 Clean Water Act, EPA requires states to list waters that are not meeting the standards for specific designated uses (National Research Council, 2001). It is not entirely possible to monitor every stream segment in the country because it would consume too much time and money. However, it is possible to adopt a monitoring and modeling approach where a model developed for one watershed could be applied to other watersheds of similar land use characteristics and geology.

A monitoring and modeling approach integrating land use maps, digital elevation models (DEM), soil maps, field data, and hydrographic data in a geographic information system (GIS) framework can accurately estimate the sediment loading in streams. The Soil Water Assessment Tool (SWAT) is a GIS based deterministic, physically based, semi distributed, functional, continuous time model that operates on a watershed scale (Neitsch et al., 2000). For modeling, the watershed is divided into a number of sub-basins. Each sub basin is further divided into different hydrological response units (HRUs) based on a unique combination of soil and land use.

Studies show that sediment fingerprinting can efficiently identify the source of sediment and their relative contributions based on a composite fingerprint of sediment properties (Collins and Walling, 2000). Estimating the sediment load and identifying its source and their relative contributions can be valuable inputs to a TMDL implementation plan.

In 1998, the North Fork of the Broad River (NFBR), Georgia was included in the 303(d) list for impacted biota and habitat. Sediment was determined to be the pollutant of concern. Therefore, the United States Environmental Protection Agency (USEPA) developed a Total Maximum Daily Load (TMDL) for sediment in the NFBR. The USEPA TMDL report (USEPA, 2000) recommended that additional data be collected to better define the sediment loading from non-point sources and that Best Management Practices (BMPs) be implemented in the watershed to reduce non-point sources of sediment. The stream was delisted in 2004 based on a macroinvetebrate survey although no measurements for sediment loads were made. At the same time, there was growing concern among stakeholders in this watershed over the high sediment load and sources of sediment. The objectives of this study were to:

1. Conduct a stream channel stability assessment using geomorphic assessments and suspended sediment yield analysis;
2. Use fingerprinting techniques to identify the primary source of suspended sediment;
3. Test the effect of spatial resolution of soil data and channel erosion on SWAT model predictions of flow and sediment.

The hypotheses tested were:

1. The stream channels in the North fork Broad River watershed are in an unstable stage of channel evolution
2. Stream bank erosion is the primary source of suspended sediment in the North Fork Broad River watershed
3. Spatial resolution of soil data has a significant influence in watershed scale hydrological modeling of flow and sediment in the southern Piedmont

LITERATURE REVIEW

Sediment load estimation

Quantification of sediment transport in rivers is important for TMDL calculations. Monitoring the amount of a pollutant transported through a stream over a period of time can provide useful information on rates of pollutant transport that is required for pollutant load reduction plans. Trends in water quality, especially the impact of management practices can be well understood through a monitoring approach. However, for larger watersheds it takes several years to see an actual impact. Monitoring for water quality involves an instantaneous measurement of the concentration of a pollutant and an associated volumetric flow, the product of which is an instantaneous load. Several methods are used to calculate long term loads from several instantaneous measurements of concentration and flow.

Suspended sediment transport rating curves are widely used by hydrologists for predicting sediment concentrations for unsampled periods (Walling, 1977; Walling and Webb, 1988; Asselman, 1999; Horowitz, 2003). Most of these methods involve developing a regression line relating suspended sediment concentration to discharge followed by interpolation and extrapolation. The efficiency of this empirical method depends largely on the availability of data pairs for all ranges of flow and sediment concentration (Horowitz et al., 2001). The daily sediment load, S (T d^{-1}) of a stream can be estimated using a simple power function of discharge, Q (m^3s^{-1}) (Nash, 1994):

$$S = aQ^b$$

where a and b are empirically determined constants. Nash used the same function to establish a relationship between sediment transport rate and discharge for 55 streams across the United States. A fairly good fit was obtained for the observed data for all streams. Improvement in sediment transport rate predictions can be made by constructing separate rating curves for different seasons (Walling, 1977).

A shortcoming in the use of a sediment rating curve is hysteresis, a phenomenon where the rate of sediment transport for a given discharge in the rising limb of the hydrograph will be different from that of the falling limb. In such cases the sediment transport rate frequently ranges over several orders of magnitude. Several studies discuss hysteresis and its causes (Walling and Webb, 1988; Asselman, 1999; Lenzi and Marchi, 2000). Hysteresis with a clockwise loop which is due to depletion of sediment is the most common. Counter-clockwise loops can be observed when the sediment sources are located far away from the monitoring station and include sources such as actively eroding channel banks and hill slopes. Hysteresis does not invalidate the use of a power function sediment rating curve if the variation about this function is symmetrically distributed at all discharges. However, if the predicted and observed sediment transport rates systematically diverge above a threshold value of discharge, then the rating curve cannot be used for extrapolation beyond that value (Nash, 1994).

Use of non linear models can cause bias in hydrologic predictions for specific values and long term means estimated through prediction (Koch and Smillie, 1986). The rating curve method tends to underpredict sediment concentrations during high flows and over predict during low flows (Horowitz et al, 2001). The non linear equation can be linearized by using a logarithmic transformation and applying appropriate bias correction factors (eg: Thomas 1985; Koch and Smillie, 1986; Cohn et al, 1989).

A direct method of estimating sediment load is to use the weighted sum of all instantaneous loads. The direct method has advantages over the rating curve approach in that it requires few assumptions about the underlying physical processes and is not subject to bias due to transformation of data (Cohn, 1995). The flow stratified sampling method involves assigning each sample to a stratum based on flow after dividing the hydrograph into different strata. Weights are provided based on the probability of discharge being within the bounds of the stratum. This method gives the lowest variance for estimating the total load for long periods with many peaks (Thomas and Lewis, 1995).

The flow interval method (Verhoff et al, 1980) uses the product of the pollutant concentration and the flow rate (pollutant flux) as a function of the flow rate for each instantaneous measurement. Total annual flux of pollutant is then calculated based on the dependency of flux with river flow rate. The estimation technique involves dividing the maximum flow rate into different flow intervals and calculating all observed pollutant fluxes in a flow interval by multiplying the observed concentration of the pollutant by the instantaneous flow. The sum of the mean fluxes for each flow interval is used for estimating total loads for the period of interest. It is assumed that the average pollutant flux for each flow interval is distributed normally.

Sediment loads can be determined by several different methods using flow and sediment concentration data. However, the present need is how to interpret the sediment load estimate or sediment concentration and stream discharge data for a given watershed in terms of the standard or reference conditions. Use of concentration standards may not be a very good idea as concentrations may vary with space and time due to differences in soils, land use, rainfall distribution, slope, discharge, and channel stability. Another problem that needs to be addressed is how to determine targets for sediment load reduction once a watershed is found to have high sediment loads.

Sediment source identification using fingerprinting techniques

The growing concern over the effect of fine sediment on water quality has resulted in the need for more effective sediment control strategies for water quality improvement at watershed scales. Better understanding of watershed processes and sources of sediment is required for designing effective sediment control strategies. Traditional methods of watershed monitoring for suspended sediment are inadequate to provide information on processes taking place within a watershed. In this context a fingerprinting approach has proved to be an effective way for tracking sediment movement within a watershed in terms of both source type and spatial origin (Walling, 2004). Several studies have used sediment fingerprinting to identify the relative contribution of various watershed sources to the total fine sediment load. The underlying principle is the difference in physical or chemical properties among the potential source materials and between the sources and the sediment samples collected from the watershed outlet. The properties that have been used for sediment source tracking include sediment color (Grimshaw and Lewin, 1980), plant pollen (Brown, 1985), mineral magnetic properties (Walden et al., 1997), rare earth elements (Kimoto et al., 2006), fallout radionuclides (Collins and Walling, 2002; Nagle and Ritchie, 2004; Walling, 2005) and stable isotopes of carbon and nitrogen (Papanicolaou et al., 2003; Fox and Papanicolaou, 2007).

Most studies relating to sediment fingerprinting have focused on the use of fallout radionuclides. The most commonly used radionuclide is cesium-137 (^{137}Cs) having a half life of 30.2 years. Radioactive ^{137}Cs was produced as a result of the atmospheric testing of nuclear weapons in the 1950s and 1960s. Global fallout of ^{137}Cs peaked in the early 1960s and subsequently decreased reaching zero levels in the mid 1980s (Walling, 2004).

Other fallout radionuclides commonly used include lead-210 (^{210}Pb) and beryllium-7 (^{7}Be). Unlike ^{137}Cs, these two radionuclides are natural in origin and their input into the atmosphere is fairly constant over time. Lead- 210 is a product of the ^{238}U decay series and has a half life of 22.26 years. It is a decay product of ^{222}Rn, the daughter of ^{226}Ra that occurs naturally in soils and rocks. In soil, ^{210}Pb remains in equilibrium with its parent. However, a small fraction of ^{222}Rn from the soil diffuses to the atmosphere and results in subsequent fallout of ^{210}Pb into the surface soils that is not in equilibrium with its parent. This fraction of ^{210}Pb is measured as unsupported/excess lead and is used as a tracer. With similar behavior to that of ^{137}Cs in soils, ^{210}Pb can be used as an alternative in erosion studies (Zapata, 2003).

Beryllium-7 is cosmogenic in origin through the atmospheric spallation of nitrogen and oxygen atoms in the troposphere and stratosphere by cosmic rays.

Compared to ^{137}Cs and ^{210}Pb, ^{7}Be has a short half live of 53.3 days that makes it suitable for short term soil erosion studies. Typically concentrated in the upper 5 mm of the soil profile it can provide good discrimination between sediment derived from surface soils and those from deeper layers (Zapata, 2003).

Nagle et al. (2007) used ^{137}Cs to quantify the relative contributions of bank and surface sediment sources in several watersheds in the New York region. A simple mixing model was used to derive the relative proportions and Monte Carlo simulations were used to account for the uncertainty in model predictions. They found that the median contribution of banks varied from 0 to 76%.

For fingerprinting sediment from multiple sources, several tracers may be required. No single diagnostic property may be able to discriminate sediment coming from a range of potential sediment sources in a river basin. A combination of multiple tracers improved sediment source discrimination in several contrasting river basins in the UK and Africa (Collins and Walling, 2002). However, the level of discrimination by a particular set of properties may not be consistent between different catchments.

Rhoton et al. (2008) used eleven soil properties in a multivariate mixing model to locate the spatial sources of suspended sediment in the Walnut Gulch Experimental Watershed (WGEW) in Tombstone, Arizona. The objective of the study was to identify sub-watersheds that contribute the greatest amounts of suspended sediment in the WGEW. Results indicated that three out of the six sub-watersheds contributed 86% of the total sediment loading in the watershed.

In a study to identify the major source of channel bottom sediment in the Wildhorse Creek drainage basin in northeastern Oregon, Nagle and Ritchie (2004) used carbon, nitrogen and bomb-derived ^{137}Cs as tracers to fingerprint sediment sources. Sediment samples were collected from the stream bottom and compared to samples collected from cultivated fields and channel banks. The tracer concentrations were analyzed in a simple mixing model to estimate the relative source contribution. They found that the majority of the channel bottom sediment originated from the stream banks.

Collins et al. (1998) used a composite fingerprint to determine the spatial sources of suspended sediment in two river basins in the UK. The objective of the study was to determine the relative contribution of sediment from different geologic sub-areas within each basin. They found that the contribution showed seasonal variability. Analysis of sediment samples from selected flood events revealed intra-storm variability in relative source contribution.

A review of the literature shows that sediment fingerprinting is being widely used all over the world, though most published studies were conducted in Europe. The techniques and tracers used in these studies can be applied to the southern Piedmont, which has a long history of erosion and sedimentation. Streams of the southern Piedmont experienced three major disturbances in the mid 20th century. The most important change was probably the reduction in upland erosion and runoff that occurred as cotton farming was abandoned and fields were converted to pasture land and forests during the cotton era (1830-1930) (Trimble, 1974). A second change was the construction of flood control reservoirs along tributaries to the main stem to in the 1960s and 1970s. The third disturbance was channelization in the floodplains during the same period. These disturbances increased stream power in the main stem and caused channels to go through a period of incision and accelerated bank erosion. The banks in the floodplains were especially prone to erode because they consisted of the non-cohesive historic sediment deposited during the cotton era. Whether these streams have reached a new stable equilibrium or are still unstable and in the process of transporting *legacy* sediment in response to the disturbances in the middle of the last century are not known. There has not been a published study on sediment fingerprinting to identify sources in the southern Piedmont. Therefore it will be interesting to determine the contribution of *legacy* sediment to the total sediment load in the Piedmont streams.

Modeling suspended sediment using GIS based models

Several models have been developed to simulate transport of sediment and other pollutants on a watershed scale. These models differ in their method of simulation, use of data, application and performance. Some of the recent hydrological simulation models intended to be used in non-urban areas include AGNPS, HSPF, WEPP and SWAT with each model having its own advantages and limitations. Among the different models SWAT is one of the comprehensive models that operate in a semi-distributed manner to account for the spatial differences in soils, land use, crops, topography, channel morphology, and weather conditions. The Soil and Water Assessment Tool (SWAT) model developed by the USDA-ARS has features of several ARS models and is a direct extension of the SWRRB model (Simulator for Water Resources in Rural Basins) (Williams et al., 1985; Arnold et al., 1990). Models that contributed significantly to the development of SWAT were CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems) (Knisel, 1980), GLEAMS (Groundwater Loading Effects on Agricultural Management Systems) (Leonard et al., 1987), and EPIC (Erosion-Productivity Impact Calculator) (Williams et al., 1984).

The SWAT model

SWAT is a deterministic, physically based, semi distributed, functional, continuous time model. In the SWAT model, a watershed is divided into a number of sub-basins and each sub basin is further subdivided into different Hydrological Response Units (HRUs) based on a unique combination of soil and land use. The hydrologic cycle is simulated by SWAT based on the water balance equation. Surface runoff is computed using a modification of the SCS curve number method (USDA Soil Conservation Service, 1972) or the Green and Ampt infiltration method (Green and Ampt, 1911). Erosion and sediment yield are estimated for each HRU with the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975). Flow routing through the main channels is modeled using a variable storage coefficient method developed by Williams (1969) or the Muskingum routing method. The transport of sediment in the channel is a function of deposition and degradation and the maximum amount of sediment that can be transported from a reach segment is calculated as a function of peak channel velocity.

Santhi et al. (2001) validated SWAT for flow, sediment and nutrients in the Bosque River Watershed to evaluate alternative management scenarios in controlling point and non-point sources of pollution. The study proved the utility of SWAT to predict flow, sediment and nutrients successfully and to study the effects of alternative management scenarios.

Jha et al. (2003) used SWAT to simulate flow and sediment yield in the Iowa River and Des Moines River watersheds. The model was found to simulate well on annual and monthly basis. Coefficients of determination (R^2) computed between the simulated and observed monthly stream flows and sediment yields for both the calibration and validation periods ranged from 0.65 to 0.83. Corresponding Nash-Sutcliffe model efficiency (E) statistics were between 0.46 and 0.83.

Chu et al. (2004) used six years of hydrologic and water quality data to calibrate and validate the capability of SWAT in assessing non-point source pollution for a 340 ha watershed in the Piedmont physiographic region in Maryland. Results indicated a strong agreement between yearly measured and simulated data for sediment, nitrate and soluble phosphorus loadings. However, model simulations of monthly sediment and nutrient loadings were poor. Overall, it was concluded that SWAT is a reasonable watershed-scale model for long-term simulation of different management scenarios.

Qi and Grunwald (2005) conducted a spatially distributed calibration and validation of water flow using the SWAT model in the Sandusky watershed of Ohio. The validation of total water flow showed a range in mean error of 0.03 to 4.00 $m^3\ s^{-1}$, a root mean square error of 0.06 to 2.56 $m^3\ s^{-1}$, correlation coefficients of 0.70 to 0.90, and Nash Sutcliffe coefficients of 0.40 to 0.73. Overall, simulations of water flow in the Sandusky watershed and sub watersheds were satisfactory except for winter rainfall runoff events.

The SWAT model uses the STATSGO database as the default dataset for soil information. With a little pre-processing of the higher resolution SSURGO soil database using the SSURGO SWAT 2.x extension for ArcView developed by Peschel et al. (2003), SSURGO data can be used for SWAT modeling. However, the use of the detailed soil database is more time and resource intensive.

In a study on the influences of soil data resolution on hydrological modeling in the Upper Sabinal River Watershed near Ulvade, Texas, uncalibrated SWAT model outputs from STATSGO and SSURGO data were compared. Peschel et al. (2006) showed that the SWAT model prediction of flow was higher when SSURGO data was used. The higher water yield was attributed to the larger saturated hydraulic conductivity values associated with the SSURGO database.

Geza and McCray (2008) compared the effect of soil data resolution on a SWAT model prediction of flow and water quality parameters in the Turkey Creek watershed in Denver, Colorado. Comparison was made before calibration because calibration can mask differences due to soil data. They found that SSURGO data predicted more flow compared to STATSGO. However, in contrast to flow STATSGO predicted more sediment transport. This was attributed to the higher area weighted average value of k_{usle} in the STATSGO database.

Gowda and Mulla (2005) calibrated a spatial model for flow and water quality using STATSGO and SSURGO soils data for High Island Creek, an agricultural watershed in south-central Minnesota. Predicted flow and sediment transport by the STATSGO model was higher compared to SSURGO. Statistical comparison of calibration results with measured data indicated excellent agreement for both soil databases for flow and sediment. However, significant differences were observed in nitrate and phosphorus losses predicted by the two models during the evaluation period.

In a study on the effect of soil data resolution on SWAT model snow melt simulation, output from SSURGO and STATSGO models were compared using calibrated results for flow for the Elm River watershed in North Dakota (Wang and Melesse, 2006). Results indicated that SSURGO made an overall better prediction for flow although both models did a comparable job in predicting high stream flows.

However, STATSGO predicted the low stream flows more accurately and had a slightly better performance during the validation period.

Only a few studies are available on the effect of soil data resolution on watershed scale modeling of flow and water quality parameters. These studies have shown contrasting results in different physiographic regions depending on the soil type and climate. A review of the literature shows that the SWAT model has been widely used for successful modeling of flow and suspended sediment in a wide range of physiographic regions. However, the effect of soil data resolution on stream flow and sediment modeling needs to be tested in the southern Piedmont region characterized by steep slopes and highly erodible soils.

REFERENCES

Arnold, J.G., J.R. Williams, A.D. Nicks, and N.B. Sammons. 1990. SWRRB: A basin scale simulation model for soil and water resources management. Texas A&M Univ. Press, College Station, TX.

Asselman, N. E. M., 1999. Suspended sediment dynamics in a large drainage basin: the River Rhine. Hydrological Processes 13:1437-1450.

Brown A. G. 1985. The potential use of pollen in the identification of suspended sediment sources. Earth Surface Processes and Landforms. 10: 27–32. Budgets. IAHS Publication.

Chu, T.W., Shirmohammadi, A., Montas, H., Sadeghi, A.M. 2004. Evaluation Of Swat Model's Sediment And Nutrient Components In Piedmont Physiographic Region Of Maryland. Transactions of The ASAE, V. 47(5), PP.1523-1538.

Cohn, T. A., 1995. Recent advances in statistical methods for the estimation of sediment and nutrient transport in rivers. Reviews of Geophysics-Richmond Virginia then Washington- 33:1117-1124.

Cohn, T. A., L. L. DeLong, E. J. Gilroy, R. M. Hirsch and D. K. Wells, 1989. Estimating constituent loads. Water Resources Research **25**:937-942.

Cohn, T.A., 1988, Adjusted maximum likelihood estimation of the moments of lognormal populations from type I censored samples: U.S. Geological Survey Open-File Report 88-350, 34 p.

Collins, A. L., and D. E. Walling. 2002. Selecting fingerprint properties for discriminating potential suspended sediment sources in river basins. Journal of Hydrology. 261:218-244.

Collins, A. L., D. E. Walling and G. J. L. Leeks, 1998. Use of composite fingerprints to determine the provenance of the contemporary suspended sediment load transported by rivers. Earth Surface Processes and Landforms 23.

Fox, J. F. and A. N. Papanicolaou. 2007. The use of carbon and nitrogen isotopes to study watershed erosion processes. Journal of the American Water Resources Association 43:1047-1064.

Geza, M., and J. E. McCray, 2008. Effects of soil data resolution on SWAT model stream flow and water quality predictions. Journal of Environmental Management 88:393-406.

Gowda, P. H., and D. J. Mulla, 2005. Scale Effects of STATSGO VS. SSURGO Soil Databases on Water Quality Predictions. pp. 5-9.

Green, W.H. and G.A. Ampt. 1911. Studies on soil physics, 1. The flow of air and water through soils. Journal of Agricultural Sciences 4:11-24.

Grimshaw, D. L. and J. Lewin, 1980. Source identification for suspended sediments. Journal of Hydrology 47:151-162.

Horowitz, A. J., 2003. An evaluation of sediment rating curves for estimating suspended sediment concentrations for subsequent flux calculations. Hydrological Processes 17:3387-3409.

Horowitz, A. J., K. A. Elrick and J. J. Smith, 2001. Estimating suspended sediment and trace element fluxes in large river basins: methodological considerations as applied to the NASQAN programme. Hydrological Processes 15:1107-1132.

Jha, M., P.W. Gassman, S. Secchi, and J.G. Arnold. 2003. Configuration of SWAT for the upper Mississippi River Basin : An application to two sub watersheds. Pp. 317-322 in Total Maximum Daily Load (TMDL) Environmental Regulations–II Proceedings of the 8-12 November 2003 Conference (Albuquerque, New Mexico USA)

Kimoto, A., M. Nearing, M. J. Shipitalo and V. O. Polyakov. 2006. Multi-year tracking of sediment sources in a small agricultural watershed using rare earth elements. Earth Surface Processes and Landforms 31:1763-1774.

Knisel, W.G. 1980. CREAMS, a field scale model for chemicals, runoff and erosion from agricultural management systems. USDA Conservation Research Rept. No. 26.

Koch, R. W. and G. M. Smillie, 1986. Bias in hydrologic prediction using log-transformed regression models. Journal of the American Water Resources Association 22:717-723.

Lenzi, M. A. and L. Marchi, 2000. Suspended sediment load during floods in a small stream of the Dolomites (northeastern Italy). Catena 39:267-282.

Leonard, R.A., W.G. Knisel, and D.A. Still. 1987. GLEAMS: Groundwater loading effects on agricultural management systems. Trans. ASAE 30(5):1403-1428.

Nagle, G. N. and J. C. Ritchie. 2004. Wheat field erosion rates and channel bottom sediment sources in an intensively cropped northeastern Oregon drainage basin. Land Degradation and Development 15: 15-16.

Nagle, G. N., T. J. Fahey, J. C. Ritchie and P. B. Woodbury, 2007. Variations in sediment sources and yields in the Finger Lakes and Catskills regions of New York. Hydrological Processes 21:828-838.

Nash, D. B., 1994. Effective sediment-transporting discharge from magnitude-frequency analysis. Journal of Geology 102:79-96.

National Research Council, 2001. Assessing the TMDL approach to water quality management. Committee to Assess the Scientific Basis of the Total Maximum Daily Load Approach to Water Pollution Reduction. National Research Council, Washington, DC.

Neitsch, S.L., J. Arnold, J.R. Kiniry, and J.R Williams. 2000. Soil and Water Assessment Tool User's Manual Version 2000, Blackland Research Center, Texas Agricultural Experiment Station, Temple, TX.

Papanicolaou, A. N., J. F. Fox and J. Marshall, 2003. Soil fingerprinting in the Palouse Basin, USA, using stable carbon and nitrogen isotopes. International Journal of Sediment Research 18:278-284.

Peschel, J. M., P. K. Haan and R. E. Lacey, 2003. A SSURGO pre-processing extension for the ArcView Soil and Water Assessment Tool. pp. 27–30.

Peschel, J. M., P. K. Haan and R. E. Lacey, 2006. Influences of soil dataset resolution on hydrologic modeling. Journal of the American Water Resources Association 42:1371-1389.

Qi, C and S. Grunwald, 2005. GIS Based Hydrologic Modeling in the Sandusky Watershed using SWAT. Trans. ASAE 48(1): 169-180

Rhoton, F. E., W. E. Emmerich, D. A. DiCarlo, D. S. McChesney, M. A. Nearing and J. C. Ritchie, 2008. Identification of Suspended Sediment Sources Using Soil Characteristics in a Semiarid Watershed. Soil Science Society of America Journal 72:1102-1112.

Santhi, C., J. G. Arnold, J. R. Williams, W. A. Dugas, R. Srinivasan and L. M. Hauck, 2001. Validation of the SWAT model on a large river basin with point and nonpoint sources 1. Journal of the American Water Resources Association 37:1169-1188.

Thomas, R. B. and J. Lewis, 1995. An evaluation of flow-stratified sampling for estimating suspended sediment loads. Journal of Hydrology 170:27-45.

Thomas, R. B., 1985. Estimating total suspended sediment yield with probability sampling. Water Resources Research 21:1381-1388.

USDA Soil Conservation Service. 1972. National Engineering Handbook Section 4 Hydrology, Chapters 4-10.

USEPA 2008. Watershed Assessment of River Stability and Sediment Supply (WARSSS) Available at http://www.epa.gov/WARSSS/sedsource/suspend.htm verified (March 12, 2009)

USEPA 2006. Water Quality Assessment and Total Maximum Daily Loads (TMDL) information. Available at http://www.epa.gov/waters/ir/index.html verified (March 12, 2009)

USEPA, 2000 Total Maximum Daily Load Development for the North Fork Broad River Watershed, EPA region 4, Water Management Division [online] Available at http://www.gaepd.org/Files_PDF/techguide/wpb/TMDL/Savannah/EPA_NorthForkBroadRiver_Sediment_TMDL.pdf (verified 2 February 2009)

Verhoff, F. H., S. M. Yaksich and D. A. Melfi, 1980. River Nutrient and Chemical Transport Estimation. Journal of the Environmental Engineering Division 106:591-608.

Walden, J., M. C. Slattery and T. P. Burt. 1997. Use of mineral magnetic measurements to fingerprint suspended sediment sources: approaches and techniques for data analysis. Journal of Hydrology. 202, 353-372

Walling, D. E. 2005. Tracing suspended sediment sources in catchments and river systems. Science of the Total Environment. 344:159-184.

Walling, D. E. and B. W. Webb, 1988. The reliability of rating curve estimates of suspended sediment yield: some further comments. IN: Sediment

Walling, D. E., 1977. Assessing the Accuracy of Suspended Sediment Rating Curves for a Small Basin. Water Resources Research 13:531-538.

Walling, D. E., 2004. Using environmental radionuclides to trace sediment mobilization and delivery in river basins as an aid to catchment management. Proceedings of the Ninth International Symposium on River Sedimentation Oct 18-21, 2004 Yichang, China.

Walling, D. E., P. N. Owens and G. J. L. Leeks, 1999. Fingerprinting suspended sediment sources in the catchment of the River Ouse, Yorkshire, UK. Hydrological Processes 13:955-975.

Wang, X., and A. M. Melesse, 2006. Effects of STATSGO and SSURGO as inputs on SWAT model's snowmelt simulation 1. Journal of the American Water Resources Association 42:1217-1236.

Williams, J.R. 1969. Flood routing with variable travel time or variable storage coefficients. Trans. ASAE 12(1):100-103.

Williams, J.R. 1975. Sediment routing for agricultural watersheds. Water Resour. Bull. 11(5):965-974.

Williams, J.R., A.D. Nicks, and J.G. Arnold. 1985. Simulator for water resources in rural basins. Journal of Hydraulic Engineering 111(6): 970-986.

Williams, J.R., C.A. Jones and P.T. Dyke. 1984. A modeling approach to determining the relationship between erosion and soil productivity. Trans. ASAE 27(1):129-144.

Zapata, F. 2003. The use of environmental radionuclides as tracers in soil erosion and sedimentation investigations: recent advances and future developments. Soil and Tillage Research 69:3-13.

CHAPTER II

CHANNEL STABILITY ASSESSMENT THROUGH RAPID CHANNEL MORPHOLOGY AND SEDIMENT YIELD ANALYSIS

[1] Mukundan, R., D. E. Radcliffe, A. Simon, and L. Klimetz. To be submitted to the *Journal of American Water Resources Association.*

ABSTRACT

Streams can be classified as stable or unstable, depending on their stage of channel evolution. Many streams of the southern Piedmont have high sediment loads and are listed as impaired under the Total Maximum Daily Load (TMDL) program and may be unstable. It is not clear what the target (reference) load or remediation measures should be for unstable streams. The objective of this study was to determine the relative channel stability for a typical southern Piedmont stream using Rapid Geomorphic Assessments (RGAs) and sediment yield analysis. RGAs were performed along 52 reaches on the North Fork Broad River (NFBR) main stem and two tributaries. Stream discharge and suspended sediment concentration were measured at the outlet of the watershed from January 2005 to December 2007. Annual sediment yields were calculated and compared to an analysis of yields for stable and unstable streams in the southern Piedmont and other ecoregions. Almost 70% of the NFBR mainstem was found to be unstable with channel erosion processes such as mass wasting and fluvial erosion dominant. Sediment yield estimates were comparable with streams in the Piedmont that are in an unstable stage of channel evolution. The estimated average annual sediment yield was 0.91 T ha^{-1} yr^{-1}. By comparison, the median annual yield is 0.19 T ha^{-1} yr^{-1} for stable streams and 0.50 T ha^{-1} yr^{-1} for unstable streams in the Piedmont ecoregion. We conclude that the NFBR is in an unstable stage of channel evolution. This has important implications for the reference condition and remediation efforts related to stream turbidity and stream bank restoration.

INTRODUCTION

Watershed assessment for suspended sediment involves determining the condition of the watershed with respect to the amount of suspended sediment generated and transported from the watershed, as well as identifying critical sources and source areas within the watershed. Alterations to natural stream channels occurring due to natural or anthropogenic causes can affect the rates of sediment transport through the channel. Hydrologic modifications (e.g., dams) can cause sediment deficits that result in stream channel scour and destruction of habitat structure (Waters, 1995). Urbanization is one form of alteration that increases water discharge and channelization is another form that potentially increases the channel slope (Doyle and Shields, 2000). This can be explained using a proportional relationship (Lane, 1955):

$$QS \propto Q_s D_s$$

where Q = water discharge ($L^3 T^{-1}$), S = channel slope, Q_s = sediment discharge ($M T^{-1}$), and D_s = particle size of the river bed material (L). Several studies have used this relationship to explain the morphologic changes to stream channels due to disturbances (Simon, 1989; Rosgen 1996; Simon and Rinaldi, 2006).

Streams of the southern Piedmont experienced three major disturbances overtime (1830-1930). The most important change was probably farming and then the reduction in upland erosion that occurred as cotton farming was abandoned and fields were converted to pasture land and forests in the first half of the 20th century (Trimble, 1974). Other changes include construction of flood control reservoirs in the 1960's and 1970's and channelization in the floodplains in the first half of the 20th century (Barrows and Phillips, 1917). These disturbances increased stream power (QS) in the main stem and caused channels to go through a period of incision and accelerated bank erosion. The banks in the floodplains were especially prone to erode because they consisted of the non-cohesive historic sediment deposited during the cotton era from the mid 19th to early 20th centuries. These disturbances to stream channels, including enlargement of channels and longer recurrence intervals for bank full discharge, resulted in less stream-floodplain interaction (Ruhlman and Nutter, 1999). As a result, most of the stream power was dissipated within the channel accelerating bank erosion. Whether these stream channels have reached a new stable equilibrium, or are still unstable and in the process of transporting high loads of legacy sediment in response to the disturbances in the middle of the last century, is not known.

Narrative standards need to be converted to a numerical load for the purpose of 303d listing and TMDL calculation and implementation (Reckhow, 2001). The numeric target should be different for stable and unstable streams, so it is important to assess channel stability as part of the TMDL process. Stream channel stability defined in geomorphic terms "is the ability of a stream, over time, in the present climate, to transport the sediment and flows produced by its watershed in such a manner that the stream maintains its dimension, pattern and profile without either aggrading or degrading" (Rosgen, 1996). This study is part of a larger effort to develop a new approach to managing sediment TMDL watersheds by incorporating geomorphic analysis of fluvial systems in watershed assessment for suspended sediment. Once the watershed is assessed for suspended sediment through qualitative and quantitative means, appropriate targets can be determined that represent the eco-region of the assessed watershed.

Simon and Hupp (1986) developed a theory of how stream channels evolve in response to disturbances such as channelization. An undisturbed stream is considered to be in Stage I (Fig. 2.1 and Table 2.1). Channelization increases stream power which causes incision of the stream channel (Stages II and III). The steep stream banks become unstable and erode in Stage IV which leads to widening of the stream. Bank failures in the form of mass wasting, a process that involves bank collapse and slumping of soil mainly due to the force of gravity is characteristic of this stage. The other dominant channel erosion process in this stage is fluvial erosion where direct scouring of stream bank material takes place by the physical action of running water. A reduction in stream power and aggrading channel bed is characteristic of Stage V. In Stage VI, the stream finally reaches a new post-disturbance equilibrium condition.

Simon et al. (2004) and Klimetz and Simon (2007) analyzed sediment rating curves (suspended sediment load vs. stream discharge) from over 400 USGS gaging stations in the southeastern U.S. The streams were identified as stable or unstable depending on the stage of channel evolution determined using rapid geomorphic assessments or analysis of channel profile time series. Median annual sediment yield was estimated for each stream (Table 2.2). Unstable streams (Stage II-V) had significantly higher sediment yields than stable streams (Stage I or VI). The Piedmont region had among the highest yields (50^{th} percentile) of any region within the southeast: 0.19 T ha^{-1} $yr^{-1}$for stable streams and 0.50 T ha^{-1} yr^{-1} for unstable streams. Of the 103 streams analyzed in the Piedmont region, about 50% were classified as unstable.

Sediment in streams negatively impacts aquatic life by lowering diversity in macroinvertebrate communities and primary productivity (Reger and Kevern, 1981). To predict the effects of suspended sediment on aquatic organisms, it is important to know the magnitude, frequency, and duration of suspended sediment concentration in a water body. Newcombe and MacDonald (1991) found that the product of sediment concentration and duration is a better indicator of effects than just sediment concentration. Suspended sediment concentrations as low as 100 mg L^{-1} can affect fish growth and feeding responses (Gregory and Northcote, 1993), while a concentration of 705 mg L^{-1} for a six-hour duration causes reduced growth rates for some fish species (Shaw and Richardson, 2001).

Principles from fluvial geomorphology and hydrology are used to make a qualitative and quantitative watershed assessment for stage of channel evolution. The objective was to determine the stability of a typical southern Piedmont stream through a qualitative analysis using rapid geomorphic assessments, and a quantitative analysis comparing the current sediment yield to regional yields for stable and unstable streams.

MATERIALS AND METHODS

Study site

The North Fork Broad River (NFBR), Georgia, was included in the Clean Water Act Section 303(d) list (1998) for impacted biota and habitat (Fig. 2.2). Sediment was determined to be the pollutant of concern. The stream was placed on the list for TMDL development as part of a consent decree in a lawsuit filed against the United States Environmental Protection Agency (USEPA) and the Georgia Environmental Protection Division (Sierra Club *v*. Hankinson, 2003). The listing was based on an assessment of land-use in the watershed that concluded there was a high probability for impacted biota and habitat, although no sampling of the stream was conducted. Therefore, the USEPA developed a TMDL for sediment for the NFBR. The TMDL report (EPA, 2000b) recommended that additional data be collected to better define the sediment loading from non-point sources. A stakeholder group involving farmers, county agents, nonprofit groups and other local and state agencies was formed for identifying sediment sources and implementation of a watershed restoration plan. In 2004 after conducting a macro invertebrate survey, the USEPA removed the NFBR watershed from the 303(d) list and reported that “habitat concerns are present but not to an extent impacting the biota”. However, no measurements of sediment concentrations or discharge were made so the annual sediment load in the NFBR remained unknown. At the same time, growing concerns on identifying the major sources of erosion, addressing stream bank erosion and erosion from construction sites and unpaved roads led to a Clean Water Act Section 319 grant in 2004 to fund BMP implementation and a program to monitor stream sediment loads. Working with others, we received a grant in 2007 to test a 3-pronged approach for managing potentially unstable southern Piedmont streams with high sediment loads: 1) sediment yield and rapid geomorphic assessment to determine stability class, 2) sediment fingerprinting to determine sediment sources, and 3) sediment modeling of BMP scenarios. The study watershed located in Franklin and Stephens counties in the Piedmont region of Georgia drains an area of 182 km^2. The major land uses in the watershed are forest (72%), pasture (15%), row crops (7%), and residential (1%).

Rapid Geomorphic Assessment (RGA)

For evaluating channel stability and to determine the stage of channel evolution, RGA was carried out using a channel stability ranking scheme (Klimetz and Simon, 2007). RGA uses nine diagnostic criteria to determine the channel stability and the dominant channel erosion process (Table 2.3). This method is only a general guideline for watershed assessment and it does not consider the upland processes. However, stream channels, being a conduit for energy, flow and materials, can reflect upland processes. The higher the channel stability index the greater the level of instability. Channel stability indices below 10 are fairly stable, between 10-20 are unstable and above 20 are highly unstable (Simon et al, 2007). Stage of channel evolution is determined based on a range of diagnostic criteria such as degree of channel incision, stream bank erosion, and accretion. A total length of 27 km along the main channel of the North Fork Broad River was assessed at 36 reaches, each having a length ranging from 700-900 m. RGAs were also carried out on the two main tributaries that drain into the NFBR: Clarke's Creek and Tom's Creek. A total of 16 triburary reaches were assessed with lengths ranging from 300-500 m (reach length assessed is a function of channel width).

Stage of channel evolution was determined for each of the 52 reaches using the channel evolution model set forth by Simon and Hupp (1986) (Fig. 2.1 and Table 2.1).

Monitoring for flow and suspended sediment

As part of the 319 grant, monitoring for suspended sediment started in the NFBR watershed in January 2005. Storm water samples were collected using an ISCO 6712 automated water sampler (ISCO Inc, Lincoln, NE) installed at the outlet of the watershed with a pressure transducer which recorded the time, date, and stage every five minutes.

Sample collection was triggered by a predetermined stage height that was programmed into the ISCO sampler. Multiple discrete samples were collected over the course of a storm hydrograph which typically lasted one day. Base flow grab samples were collected at biweekly intervals in addition to the storm flow samples. Rainfall was measured at the outlet monitoring station (as well as three other monitoring stations) with a tipping-bucket rain gauge programmed to record precipitation amounts every five minutes.

Manning's equation was used to calculate the flow velocity (m s^{-1}) based on stream stage from which actual discharge (m^3 s^{-1}) was calculated by multiplying by the cross sectional area (m^2) of the channel. A rating curve was developed so that stream stage could be converted directly to discharge. To construct the rating curve, channel dimensions were measured to determine the hydraulic radius of the stream. Stream velocity, hydraulic radius, slope, and an estimated roughness coefficient were used to estimate discharge for a given stage height. The stream channel at the outlet of the watershed was stable and did not change its dimension during the period of monitoring.

Estimating suspended sediment concentration in water samples

Representative suspended sediment samples were selected from each storm event based on the time of sampling. Care was taken to make sure the samples represented the entire hydrograph. FLOWLINK- Advanced Flow Data Management software was used for data analysis and sample selection (ISCO Inc, Lincoln, NE). Samples were analyzed for suspended sediment concentration (SSC) in mg L^{-1} and turbidity in Nephelometric Turbidity Units (NTU). SSC was determined using the evaporation method described by Guy (1969) which involves filtering a 250-mL subsample into a pre-weighed 45-µm filter. The filter was then kept in an oven at 110^{o}C for 24 hours and reweighed. SSC was calculated by subtracting the mass of the clean pre-weighed filter from the mass of the filter containing the filtrate. Turbidity was measured in a separate subsample using a HACH 2100P turbidimeter (HACH Company, Loveland, CO).

Analyzing suspended sediment data

The instantaneous discharge and sediment concentration data were used for sediment load estimates. Two different methods were adopted for sediment load estimation, one using the rating curve of discharge vs. sediment loads and the other using sediment flux estimates at different discharge intervals.

The rating curve method used the LOADEST program (Runkel et al., 2004) to develop a regression equation relating stream discharge (Q) to sediment loads (L). Instantaneous loads were estimated using a quadratic equation:

$$\ln(L) = a_0 + a_1 \ln(Q) + a_2 \ln(Q^2)$$

where L = suspended sediment load (kg d^{-1}), $\ln(Q)$ = normalized discharge and a_0, a_1 and a_2 are regression coefficients.

LOADEST uses three methods to account for the error in load estimates due to natural log retransformation bias: the maximum likelihood estimation (MLE) (Cohn et al., 1989), the adjusted maximum likelihood estimation (AMLE) (Cohn, 1988) and the least absolute deviation estimation (LAD) (Duan, 1983). Sediment yields in metric tons per hectare per year (T ha^{-1} yr^{-1}) were calculated by dividing the average annual load by the watershed area. Sediment yields were preferred to loads for ease of comparison with values from other watersheds.

The rating curve method tends to under-predict sediment concentrations during high flows and over-predict during low flows (Horowitz et al., 2001). Due to the wide differences in flow regimes during the period of monitoring, it was expected that the rating curve might overestimate base flow sediment concentrations and create errors in sediment load estimates. Therefore a second method was used for comparison.

The second method for sediment load estimation used sediment fluxes for different discharge intervals (Verhoff et al., 1980). The maximum discharge ($Q_{\max}$) was divided into different discharge intervals (ΔQ). For each discharge interval *i,* all observed fluxes F_{ij} were calculated by multiplying the observed SSC (mg L^{-1}) by the instantaneous discharge (m^3 sec^{-1}) at that concentration. The average flux for each interval ($\overline{F}_i$) was calculated as:

$$\overline{F}_i = \frac{\sum_{j=1}^{k_i} F_{ij}}{k_i}$$

where k_i was the number of observed data points in each interval. For computing the total sediment flux for the period of monitoring, the percentage of time a particular discharge value was equaled or exceeded was calculated from the flow duration curve for the period of monitoring. The average hourly flux ($\overline{F}$) of suspended sediment was calculated by:

$$\overline{F} = \sum_{i=1}^{n} \overline{F}_i P_i$$

where P_i is the frequency that discharge occurs in the *i* interval during a given time period.

The average duration of a storm hydrograph for the stream was about one day; therefore, the frequency of flux estimation was reduced to hourly intervals. This was made possible by converting a subset of the estimated 5-minute data to hourly data using PROCEXPAND in SAS (SAS Institute Inc, NC). This method (flow interval method) is based on the assumption that the average flux for each interval is normally distributed. However, this may not always be true in the case of sediment concentration data due to several factors such as land-use activities, differences in flow regimes, hysteresis, frequency and duration of precipitation, etc. To incorporate all these uncertainties and obtain a better estimate of sediment load, a Monte Carlo simulation approach was adopted making use of all available sediment flux values within a given flow interval. The sediment load estimation was done 20,000 times with the @Risk software (Palisade Corporation, NY) using all observed instantaneous flux values. The simulation statistics were used to obtain a better estimate of sediment load. The estimated sediment load for the NFBR was compared with the estimated loads for stable and unstable streams in the Piedmont eco-region.

The magnitude, frequency and duration of suspended sediment concentrations were calculated from the monitored data for suspended sediment and stream discharge. This information could also be used to determine trends in water quality and the effect of management practices on suspended sediment concentrations over the duration of monitoring programs. Also the duration that a given suspended concentration is maintained may be used as a metric to determine whether moderate, long duration concentrations are more harmful to aquatic organisms compared to high sediment concentrations for a relatively short duration (Klimetz and Simon, 2007).

RESULTS

Relative stability of main stream channels

The channel stability index value for the 36 stream reaches assessed ranged from 11.5 to 23.5 showing that there was a wide range in channel stability in different areas of the watershed (Fig. 2.3 and Table 2.4). The median value was 18.5 and the mean value was 17.8 indicating that a majority of the channel reaches were considered unstable. Fluvial channel erosion was found to affect approximately 40% of both banks, while mass wasting affected almost half of the right or outside banks and only 14% of left or inside banks. More that 50% accretion was observed in at least one bank in 20 stream reaches assessed, indicating excessive sediment delivery from upstream or overland sources (Table 2.4).

Almost 70% of the NFBR mainstem was found to be unstable with 58% of the channel reaches in Stage V, the aggradational phase. Dominance of Stage V is an indication that mass wasting is slowing down and that channels are in a lower energy condition for suspended sediment transport and a subsequent decrease in yield (Simon, 1989). Most of the channel segments have passed the threshold stage (Stage IV) of channel evolution and hence the peak period of sediment transport. However, the relative instability of the channel reaches shows a high potential for bank erosion being a major source of suspended sediment. All stable reaches were found to be stage VI having restabilized to a post-modification quasi-equilibrium.

Relative stability of tributary channels

The two tributaries showed marked differences in the stages of channel evolution and the dominant channel erosion processes (Fig. 2.3 and Table 2.5). Stage V was the dominant

(60%) stage in the lower tributary (Clarke's Creek) whereas Stage III was the dominant (66%) stage in the upper tributary (Tom's Creek). A dominance of degradational reaches in Tom's Creek not present in Clarke's Creek suggest that a knick point has recently passed through the NFBR main stem, to which Tom's creek is still adjusting geomorphically. Clarke's Creek bed levels have already adjusted and are now exhibiting bank adjustment features such as mass wasting and are classified in Stage V. Fluvial erosion was found to dominate the stream banks of Clarke's Creek whereas the stream banks of Tom's Creek were comparatively stable with none of the bank erosion processes dominating. Unlike many of the main channel reaches, none of the tributary reaches were dominated by mass wasting (Table 2.5). Therefore, relative to the main channel the tributary channels were found to be less significant as a source of fine sediment. However, the contribution from stream banks of Clarke's Creek seems to be relatively higher than that of Tom's Creek.

Suspended sediment yield estimates and regional values

The average annual suspended sediment yield in the NFBR watershed estimated using the rating curve approach was 0.91 T ha^{-1} yr^{-1} for the full three year monitoring period. The model gave an r^2 of 0.85 for yield prediction. The p-values for the regression coefficients of the yield variables were statistically significant ($p<0.001$). Bias correction methods did not change the mean sediment yield estimates substantially (Table 2.6). The adjusted maximum likelihood (AMLE) method was preferred as it incorporates uncertainty in yield estimates expressed in terms of standard error (SE) and standard error of prediction (SEP). The 95 percent confidence interval in mean sediment yield estimates using the SEP method was 0.72 to 1.15 T ha^{-1} yr^{-1}. The MLE and the AMLE methods produced similar average annual yield estimates whereas the LAD method produced a slightly higher average annual sediment yield estimate of 1.0 T ha^{-1} yr^{-1}. The flow interval method produced an average annual suspended sediment yield estimate of 0.92 T ha^{-1} yr^{-1}. Thus the two contrasting methods produced similar sediment yieldd estimates. Monte Carlo simulation using the whole range of flux values within a flow interval gave a mean yield estimate of 0.91 T ha^{-1} yr^{-1} with an inter-quartile range of 0.76 to 0.99 T ha^{-1} yr^{-1}. The minimum and maximum yield values estimated by this method were 0.45 and 2.31 T ha^{-1} yr^{-1} respectively.

Calculated sediment yield estimates for the NFBR watershed are comparable with the 75th percentile mean annual sediment yield estimates for unstable sites in the Piedmont region (Table 2.2). These results are based on three years of suspended sediment monitoring during which most of the sediment was transported in the year 2005 characterized by a relatively high water yield compared to the other two years (2006 and 2007) (Fig. 2.4). The mean annual stream discharge of the Broad River at the USGS gauging station near Bell, Georgia, in 2005 was 80 percent higher than the ten year average (1999-2008). In 2006 and 2007 the mean annual discharge for the river was 6% and 21% lower than the ten year average. Similar results were observed by Landers et al. (2007) where the annual sediment yield estimates (1998-2003) for six watersheds in the Georgia Piedmont varied substantially between years due to differences in precipitation.

The average annual suspended sediment yields for 2006 and 2007 were 0.26 and 0.37 T ha^{-1} yr^{-1} respectively. The sediment yields for the lower water yield years were comparable with the long-term average annual sediment yield at Murder Creek, another watershed in the Georgia Piedmont with similar land use characteristics (Jackson et al., 2005). Fitted relationships between suspended sediment and discharge (1977-2003) from a USGS gauging station on Murder Creek produced sediment yield estimates of up to 0.28 T ha^{-1} yr^{-1}. Therefore, long-term monitoring would probably result in a sediment yield estimate that is lower than the current estimate for the three years of monitoring, but higher than the yield estimates for the low water yield years. In that case the mean sediment yield for the NFBR watershed would likely fall near the 50th percentile for unstable sites (Table 2.2).

This sediment yield analysis compliments the RGA results that show the channels are in a transition phase from unstable to stable conditions (Stage V to VI). However, the high sediment yield in 2005 shows that there is no shortage of sediment supply in the NFBR watershed and that varying precipitation levels in the future may affect short-term suspended sediment yields.

Reference conditions and target yield

A long-term mean annual yield close to the median value (50th percentile) for unstable streams (0.50 T ha^{-1} yr^{-1}) would require a 60 percent reduction in yield to be in the median range for stable sites (0.19 T ha^{-1} yr^{-1}). Transformation of stream channels from unstable to stable conditions might take several decades. However, an unstable channel can have a lower sediment yield than a stable channel with a relatively high sediment yield as the confidence intervals of sediment yields for stable and unstable channels overlap (Table 2.2). This depends largely on the relative contribution of channel sources vs. upland surface sources. Significant yield reduction may be achieved once the dominant sources and source areas are identified. It may be possible to analyze the data from Klimetz and Simon (2007) and determine which of the unstable streams might be considered reference streams for the unstable condition. These would be streams that were unstable, but had dominant forest landuse where upland sources of sediment are minimal. The sediment load of these streams might correspond to a given percentile for unstable streams, for example the 25th percentile. This analysis would be similar to the approach used by USEPA to recommend ambient nutrient standards (EPA, 2000a).

The magnitude, frequency and duration analysis using the three years of monitoring data indicated values representing unstable sites in the Piedmont (Fig. 2.5 and 2.6). A suspended sediment concentration of 593 mg L^{-1} was exceeded 1% of the time which is much higher than the median reference value for the Piedmont unstable streams which is 231 mg L^{-1}. However, a concentration of 7 mg L^{-1} was exceeded 90% of the time which is lower than the median reference value of 12 mg L^{-1} for unstable streams. The duration curve (average number of consecutive hours a given concentration is exceeded) was comparable with the median values (Klimetz and Simon, 2007) for the unstable streams in the Piedmont ecoregion. The median suspended sediment concentration of 42 mg L^{-1} was much higher than the median value of 22 mg L^{-1}. These results could change with long term monitoring.

CONCLUSIONS

Stream channels in the NFBR watershed are relatively unstable as evidenced by rapid geomorphic assessments. More sediment is delivered to stream channels than can be efficiently transported from the system, evident from the large number of accretional zones within the channel. Sediment yield estimates also indicate an unstable condition in that they are comparable with median yields for unstable streams in the southern

Piedmont region. Over a period of time, one can expect a complete transition of the channel towards stable (stage VI) conditions where most of the suspended sediment will be found emanating from the tributary streams and field gullies. However, this process may take thousands of years (Jackson et al., 2005).

In our experience, the NFBR is a typical rural watershed in the southern Piedmont and hence the results may be applicable to other watersheds in the region. The vestiges of human activity in the past are still affecting channels of this stream. Stream bank erosion is likely an important source of suspended sediment in this stream and will be difficult to reduce as a source. However, there are likely to be other upland sources that could be addressed efficiently. In the second phase of this three-pronged approach to potentially unstable streams with high sediment loads, we are using sediment fingerprinting to identify and quantify the different sources in the NFBR.

Acknowledgement

The authors would like to thank the U. S. Environmental Protection Agency (USEPA) for providing a Section 319 grant through Georgia Environmental Protection Division (GAEPD) for the monitoring program. Geomorphic assessment of stream channels was conducted as part of the USDA-CSREES grant # 2007-51130-03869 (A New Approach to Sediment TMDL Watersheds in the Southern Piedmont)

REFERENCES

Barrows, H. H. and J. V. Phillips. Agricultural Drainage in Georgia 1917. 1-122. Atlanta, GA, Geological Survey of Georgia. Bulletin No. 32.

Cohn, T. A., L. L. DeLong, E. J. Gilroy, R. M. Hirsch and D. K. Wells, 1989. Estimating constituent loads. *Water Resources Research* **25**:937-942.

Cohn, T.A., 1988, Adjusted maximum likelihood estimation of the moments of lognormal populations from type I censored samples: *U.S. Geological Survey Open-File Report* 88-350, 34 p.

Doyle, M. W. and F. D. Shields, 2000. Incorporation of bed texture into a channel evolution model. *Geomorphology* **34**:291-309.

Duan, N., 1983. Smearing estimate: a nonparametric retransformation method. *Journal of the American Statistical Association* **78**:605-610.

EPA 2000. Ambient water quality criteria recommendations. Rivers and streams in nutrient ecoregion IX. Available online at www.epa.gov/waterscience/criteria/nutrient/ecoregions/rivers/rivers_9.pdf (verified April 16, 2009)

EPA, 2000 Total Maximum Daily Load Development for the North Fork Broad River Watershed, EPA region 4, Water Management Division Available at http://www.gaepd.org/Files_PDF/techguide/wpb/TMDL/Savannah/EPA_NorthForkBroadRiver_Sediment_TMDL.pdf (verified February 2009)

Gregory, R. S. and T. G. Northcote, 1993. Surface, planktonic, and benthic foraging by juvenile chinook salmon(Oncorhynchus tshawytscha) in turbid laboratory conditions. *Canadian Journal of Fisheries and Aquatic Sciences* **50**:233-240.

Guy, H., 1969. Laboratory theory and methods for sediment analysis. *Water-Resource Investigations United States Geological Survey*

Horowitz, A. J., K. A. Elrick and J. J. Smith, 2001. Estimating suspended sediment and trace element fluxes in large river basins: methodological considerations as applied to the NASQAN programme. *Hydrological Processes* **15**:1107-1132.

Jackson, C. R., J. K. Martin, D. S. Leigh and L. T. West, 2005. A southeastern piedmont watershed sediment budget: Evidence for a multi-millennial agricultural legacy. *Journal of soil and water conservation* **60**:298-310.

Klimetz, L. and A. Simon, 2007. Suspended-Sediment Transport Rates for Level III Ecoregions of EPA Region 4: The Southeast. *USDA-ARS National Sedimentation Laboratory Research Report*. No. 55. 135 pp.

Landers, M. N., P. D. Ankcorn and K. W. McFadden, 2007 Watershed Effects on Streamflow Quantity and Quality in Six Watersheds of Gwinnett County, Georgia. USGS Scientific Investigations Report 2007-5132.

Lane, E. W., 1955. The importance of fluvial morphology in hydraulic engineering. *Proceedings of the American Society of Civil Engineers* **81**:1-17.

Newcombe, C. P. and D. D. Macdonald, 1991. Effects of Suspended Sediments on Aquatic Ecosystems. *North American Journal of Fisheries Management* **11**:72-82.

Reckhow, K.H. 2001. Assessing the TMDL approach to water quality management. Committee to Assess the Scientific Basis of the Total Maximum Daily Load Approach to Water Pollution Reduction. *National Research Council*, Washington, DC.

Reger, S. J. and N. R. Kevern, 1981. Benthic macroinvertebrate diversity in three Michigan streams. J. Freshwater Ecol. **1**: 179–187.

Rosgen, D., 1996. *Applied river morphology.* Wildland Hydrology Pagosa Springs, Colo.

Ruhiman M. B. and W. L. Nutter, 1999. Channel morphology evolution and overbank flow in the Georgia Piedmont. *Journal of the American Water Resources Association* **35**:277-290.

Runkel, R. L., C. G. Crawford and T. A. Cohn, 2004. Load estimator (LOADEST): A FORTRAN program for estimating constituent loads in streams and rivers. *US Geological Survey Techniques and Methods Book* **4**:69.

Shaw, A. E. and J. S. Richardson, 2001. Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout(Oncorhynchus mykiss) growth and survival. *Canadian Journal of Fisheries and Aquatic Sciences* **58**:2213-2221.

Sierra Club *v*. Hankinson. 2003. United States Court of Appeals for the 11th Circuit. Available online at http://altlaw.org/v1/cases/1128530 (verified 29 October 2008).

Simon, A. and C. R. Hupp, 1986. Channel Evolution in Modified Tennessee Channels. *Proceedings of the Fourth Federal Interagency Sedimentation Conference March 24-27, 1986, Las Vegas, Nevada.* **2**.

Simon, A., 1989. The discharge of sediment in channelized alluvial streams. *Journal of the American Water Resources Association* **25**:1177-1188.

Simon, A., W. Dickerson and A. Heins, 2004. Suspended-sediment transport rates at the 1.5-year recurrence interval for ecoregions of the United States: transport conditions at the bankfull and effective discharge *Geomorphology* **58**:243-262.

Simon, A. and M. Rinaldi, 2006. Disturbance, stream incision, and channel evolution: The roles of excess transport capacity and boundary materials in controlling channel response. *Geomorphology* **79**:361-383.

Simon, A., M. Doyle, M. Kondolf, F. D. Shields, B. Rhoads and M. McPhillips, 2007. Critical Evaluation of How the Rosgen Classification and Associated Natural Channel Design Methods Fail to Integrate and Quantify Fluvial Processes and Channel Response. *Journal of the American Water Resources Association* **43**:1117-1131.

Trimble, S.W. 1974. Man-induced soil erosion on the Southern Piedmont, 1700-1970. *Soil Conservation Society of America*, Ankeny, IA 180.

Verhoff, F. H., S. M. Yaksich and D. A. Melfi, 1980. River Nutrient and Chemical Transport Estimation. *Journal of the Environmental Engineering Division* **106**:591-608.

Waters, T. F., 1995. Sediment in streams: sources, biological effects, and control. *American Fisheries Society Monograph* **7**:251.

Table 2.1 Description of channel evolution model (from Klimetz and Simon, 2007)

Stage	Descriptive Summary
I	*Pre-modified* – Stable bank conditions, no mass wasting, small, low angle bank slopes. Established woody vegetation, convex upper bank, concave lower bank.
II	*Constructed* – Artificial reshaping of existing banks. Vegetation often removed, banks steepened, heightened and made linear.
III	*Degradation* – Lowering of channel bed and consequent increase of bank heights. Incision without widening. Bank toe material removed causing an increase in bank angle.
IV	*Threshold* – Degradation and basal erosion. Incision and active channel widening. Mass wasting from banks and excessive undercutting. Leaning and fallen vegetation. Vertical face may be present.
V	*Aggradation* – Deposition of material on bed, often sand. Widening of channel through bank retreat; no incision. Concave bank profile. Filed material re-worked and deposited. May see floodplain terraces. Channel follows a meandering course.
VI	*Restabilization* – Reduction in bank heights, aggradation of the channel bed. Deposition on the upper bank therefore visibly buried vegetation. Convex shape. May see floodplain terraces.

Table 2.2 Quartile measures describing suspended-sediment transport rates in the Piedmont ecoregion (from Klimetz and Simon, 2007)

	Mean annual yield (T ha^{-1} yr^{-1})		
Percentile	All streams	Stable	Unstable
10^{th}	0.10	0.05	0.15
25^{th}	0.17	0.11	0.26
50^{th}	0.39	0.19	0.50
75^{th}	0.56	0.33	0.86
90^{th}	1.08	0.40	1.17

Table 2.3 Channel stability ranking scheme used to conduct rapid geomorphic assessments (RGAs). The channel stability index is the sum of the values obtained. (From Klimetz and Simon 2007)

CHANNEL-STABILITY RANKING SCHEME

River__________________ **Station Description**____________________________________

Date _____________ **Crew** _______________ **Samples Taken**_________________________

Pictures (circle) upstream downstream cross section **Slope**__________ **Pattern:** Meandering Straight Braided

Value

1. Primary bed material

	Bedrock	Boulder/Cobble	Gravel	Sand	Silt Clay		Value
	0	1	2	3	4		____

2. Bed/bank protection

	Yes	No	(with)	1 bank protected	2 banks		Value
	0	1		2	3		____

3. Degree of incision (Relative ele. Of "normal" low water; floodplain/terrace @ 100%)

	0-10%	11-25%	26-50%	51-75%	76-100%	Value
	4	3	2	1	0	____

4. Degree of constriction (Relative decrease in top-bank width from up to downstream)

	0-10%	11-25%	26-50%	51-75%	76-100%	Value
	0	1	2	3	4	____

5. Streambank erosion (Each bank)

	None	fluvial	mass wasting (failures)	Value
Left	0	1	2	____
Right	0	1	2	____

6. Streambank instability (Percent of each bank failing)

	0-10%	11-25%	26-50%	51-75%	76-100%	Value
Left	0	0.5	1	1.5	2	____
Right	0	0.5	1	1.5	2	____

7. Established ripirian woody-vegetative cover (Each bank)

	0-10%	11-25%	26-50%	51-75%	76-100%	Value
Left	2	1.5	1	0.5	0	____
Right	2	1.5	1	0.5	0	____

8. Occurrence of bank accretion (Percent of each bank with fluvial deposition)

	0-10%	11-25%	26-50%	51-75%	76-100%	Value
Left	2	1.5	1	0.5	0	____
Right	2	1.5	1	0.5	0	____

9. Stage of channel evolution

	I	II	III	IV	V	VI	Value
	0	1	2	4	3	1.5	____

Total

Table 2.4 Rapid Geomorphic Assessments (RGA) summary (Main channel)

Cross-section number	Stage of channel evolution	Bed material	Bed or bank protection	Incision	Constriction	Streambank erosion		Streambank instability		Woody vegetative cover		Bank accretion		Channel stability index
						Left	Right	Left	Right	Left	Right	Left	Right	
0	VI	Sand	No	11-25%	0-10%	None	Fluvial	0-10%	0-10%	26-50%	26-50%	51-75%	11-25%	13.5
800	VI	Sand	No	11-25%	0-10%	Fluvial	None	0-10%	0-10%	26-50%	26-50%	0-10%	26-50%	14.5
1600	VI	Sand	No	11-25%	0-10%	Fluvial	Fluvial	0-10%	0-10%	26-50%	11-25%	26-50%	11-25%	15.5
2300	III	Sand	No	11-25%	0-10%	Fluvial	Fluvial	0-10%	11-25%	0-10%	0-10%	11-25%	0-10%	19.0
3100	V	Sand	No	11-25%	0-10%	Fluvial	Mass Wasting	11-25%	51-75%	11-25%	0-10%	26-50%	26-50%	20.5
4000	VI	Sand	No	11-25%	0-10%	Fluvial	None	11-25%	11-25%	26-50%	51-75%	11-25%	51-75%	14.0
5000	III	Sand	No	11-25%	0-10%	Fluvial	Fluvial	11-25%	11-25%	0-10%	0-10%	0-10%	11-25%	19.5
5900	V	Sand	No	11-25%	0-10%	None	Mass Wasting	11-25%	76-100%	26-50%	0-10%	76-100%	0-10%	19.5
6600	V	Sand/Gravel	No	11-25%	0-10%	Mass Wasting	Mass Wasting	51-75%	51-75%	11-25%	11-25%	26-50%	76-100%	20.5
7300	VI	Sand	No	11-25%	0-10%	None	Fluvial	11-25%	26-50%	51-75%	11-25%	76-100%	11-25%	14.5
8000	VI	Sand/Gravel	No	11-25%	0-10%	None	Fluvial	11-25%	11-25%	11-25%	26-50%	51-75%	0-10%	15.0
8800	V	Sand	No	11-25%	0-10%	Fluvial	None	11-25%	11-25%	11-25%	11-25%	11-25%	26-50%	17.5
9700	V	Sand	No	11-25%	0-10%	Fluvial	Mass Wasting	0-10%	51-75%	26-50%	0-10%	51-75%	11-25%	19.5
10600	V	Sand	No	11-25%	0-10%	None	Mass Wasting	11-25%	76-100%	26-50%	0-10%	76-100%	0-10%	19.5
11500	V	Sand	No	11-25%	0-10%	None	Mass Wasting	0-10%	76-100%	11-25%	11-25%	76-100%	11-25%	18.5
12400	V	Sand	No	11-25%	0-10%	None	Mass Wasting	11-25%	51-75%	26-50%	11-25%	51-75%	11-25%	18.5
13200	V	Sand	No	11-25%	0-10%	Mass Wasting	Fluvial	51-75%	26-50%	26-50%	11-25%	51-75%	11-25%	20.0
14000	V	Sand	No	11-25%	0-10%	Mass Wasting	Mass Wasting	51-75%	51-75%	11-25%	0-10%	11-25%	11-25%	23.5
14800	V	Sand	No	11-25%	0-10%	Mass Wasting	None	51-75%	11-25%	11-25%	11-25%	26-50%	26-50%	19.0
15600	VI	Sand	No	11-25%	0-10%	Fluvial	Fluvial	11-25%	11-25%	26-50%	26-50%	26-50%	26-50%	15.5
16400	V	Sand	No	11-25%	11-25%	Fluvial	Mass Wasting	51-75%	26-50%	11-25%	26-50%	11-25%	51-75%	21.0
17200	V	Sand	No	11-25%	0-10%	Mass Wasting	Mass Wasting	51-75%	51-75%	11-25%	11-25%	26-50%	11-25%	22.5
18000	V	Sand	No	11-25%	0-10%	None	Mass Wasting	0-10%	51-75%	26-50%	11-25%	76-100%	0-10%	18.0
18800	V	Sand	No	11-25%	0-10%	Fluvial	Mass Wasting	26-50%	51-75%	51-75%	51-75%	11-25%	11-25%	19.5
19600	V	Sand	No	11-25%	0-10%	Fluvial	Mass Wasting	26-50%	51-75%	26-50%	26-50%	26-50%	26-50%	19.5
20400	V	Sand	No	11-25%	0-10%	None	Mass Wasting	0-10%	76-100%	51-75%	11-25%	76-100%	11-25%	17.5
21100	V	Sand	No	11-25%	0-10%	Fluvial	Mass Wasting	11-25%	51-75%	51-75%	51-75%	11-25%	11-25%	19.0
21800	VI	Sand	1 Bank	11-25%	0-10%	None	Fluvial	0-10%	11-25%	51-75%	11-25%	51-75%	26-50%	14.5
22500	VI	Gravel	No	11-25%	0-10%	None	Fluvial	0-10%	11-25%	76-100%	51-75%	51-75%	11-25%	11.5
23200	V	Sand	No	11-25%	11-25%	None	Fluvial	11-25%	51-75%	51-75%	26-50%	51-75%	26-50%	17.0
23900	VI	Sand	No	11-25%	0-10%	None	Fluvial	0-10%	11-25%	51-75%	26-50%	76-100%	11-25%	13.0
24600	VI	Sand	No	11-25%	0-10%	None	Fluvial	0-10%	26-50%	51-75%	26-50%	76-100%	26-50%	13.0
25300	V	Sand	1 Bank	11-25%	11-25%	None	Fluvial	0-10%	51-75%	26-50%	26-50%	26-50%	0-10%	19.5
26000	V	Sand	No	11-25%	0-10%	None	Mass Wasting	0-10%	76-100%	76-100%	0-10%	76-100%	0-10%	18.0
26700	III	Boulder/Cobble	Yes	11-25%	11-25%	Fluvial	Fluvial	26-50%	11-25%	26-50%	51-75%	11-25%	0-10%	15.5
27300	IV	Sand	1 Bank	11-25%	0-10%	Fluvial	Mass Wasting	51-75%	26-50%	11-25%	51-75%	26-50%	0-10%	22.5

* Cross-sections are from downstream to upstream

Table 2.5 Rapid Geomorphic Assessments (RGA) summary (Tributaries)

Cross-section number	Stage of channel evolution	Bed material	Bed or bank protection	Incision	Constriction	Streambank erosion		Streambank instability		Woody vegetative cover		Bank accretion		Channel stability index
						Left	Right	Left	Right	Left	Right	Left	Right	
1C	V	Gravel	No	11-25%	0-10%	Fluvial	Fluvial	25-50%	11-25%	0-10%	0-10%	0-10%	0-10%	20.5
2C	V	Gravel	No	11-25%	0-10%	Fluvial	Fluvial	0-10%	0-10%	0-10%	0-10%	25-50%	11-25%	17.5
10C	III	Silt/Clay	No	0-10%	0-10%	None	None	0-10%	0-10%	11-25%	11-25%	11-25%	0-10%	16.5
4C	V	Sand	No	11-25%	0-10%	Fluvial	Fluvial	0-10%	0-10%	11-25%	11-25%	0-10%	0-10%	18.5
3C	V	Gravel	No	0-10%	0-10%	Fluvial	None	0-10%	0-10%	26-50%	26-50%	11-25%	11-25%	17.0
5C	V	Sand	No	11-25%	0-10%	Fluvial	Fluvial	0-10%	0-10%	11-25%	11-25%	26-50%	26-50%	17.0
6C	V	Sand	No	11-25%	0-10%	Fluvial	Fluvial	11-25%	11-25%	11-25%	11-25%	11-25%	51-75%	18.0
7C	VI	Sand	No	11-25%	0-10%	Fluvial	Fluvial	0-10%	0-10%	26-50%	26-50%	26-50%	26-50%	14.5
8C	VI	Sand	No	11-25%	0-10%	None	Fluvial	0-10%	0-10%	11-25%	11-25%	76-100%	0-10%	14.5
9C	III	Sand	No	0-10%	0-10%	None	None	0-10%	0-10%	51-75%	26-50%	0-10%	0-10%	15.5
16T	III	Sand	No	0-10%	0-10%	None	None	0-10%	0-10%	11-25%	26-50%	0-10%	0-10%	16.5
15T	VI	Sand	No	11-25%	0-10%	None	None	0-10%	0-10%	11-25%	26-50%	0-10%	0-10%	15.0
13T	III	Sand	No	0-10%	0-10%	None	None	0-10%	0-10%	0-10%	0-10%	0-10%	0-10%	18.0
12T	III	Sand	No	0-10%	0-10%	Fluvial	None	0-10%	0-10%	0-10%	0-10%	0-10%	0-10%	19.0
11T	IV	Sand	No	11-25%	0-10%	Fluvial	Fluvial	26-50%	11-25%	0-10%	0-10%	0-10%	0-10%	22.5
14T	III	Sand	No	11-25%	0-10%	None	None	0-10%	0-10%	0-10%	0-10%	0-10%	0-10%	18.0

* Cross-sections are from downstream to upstream; C-Clarke's Creek T-Tom's Creek

Table 2.6 Sediment yield estimates (T ha^{-1} yr^{-1})

Method	Mean annual sediment yield	2005	2006	2007
Rating curve				
AMLE	0.91	2.30	0.26	0.37
95% CI	(0.72-1.15)	(1.75-2.97)	(0.22-0.30)	(0.29-0.47)
MLE	0.91	2.30	0.26	0.37
LAD	1.00	2.54	0.26	0.39
Flow interval				
Mean flux	0.92			
Monte Carlo	0.91 Inter-quartile range (0.76-0.99)			

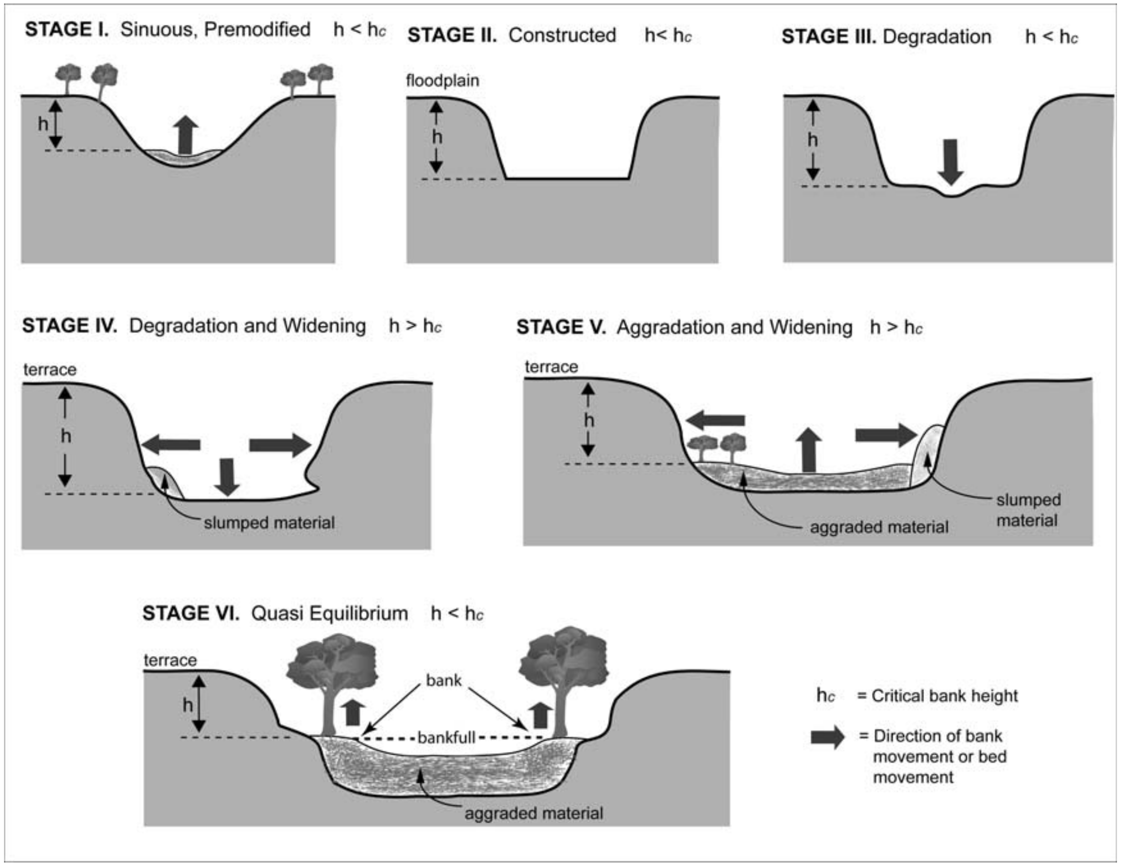

Figure 2.1 The channel evolution model of Simon and Hupp (1989)

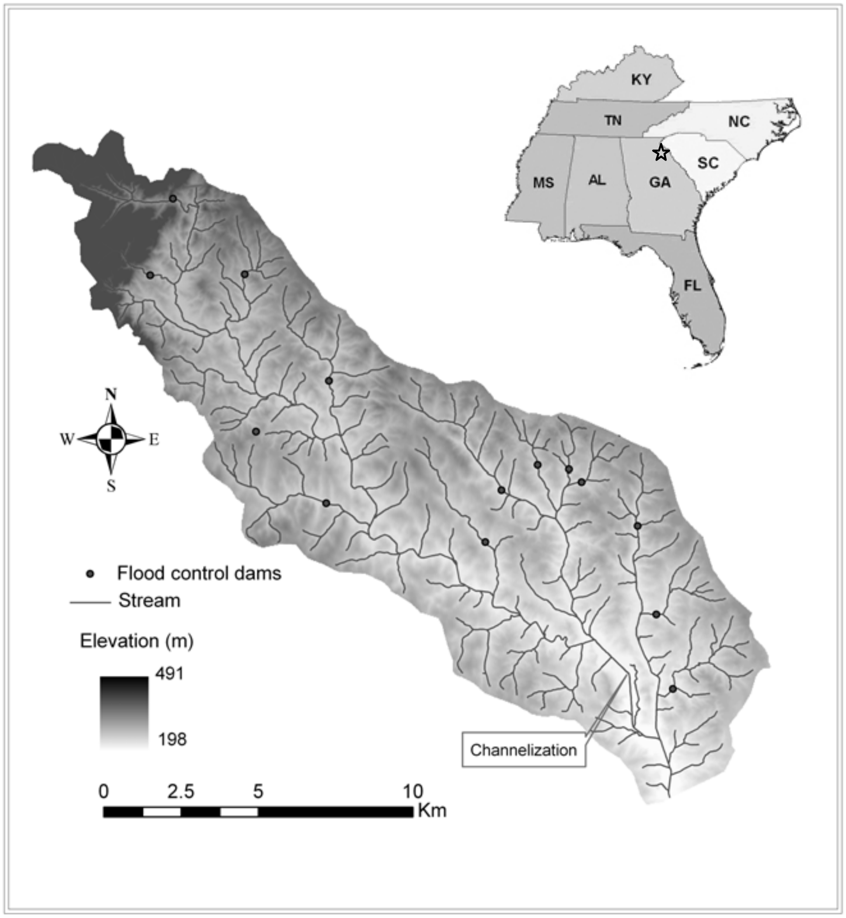

Figure 2.2 The North Fork Broad River Watershed

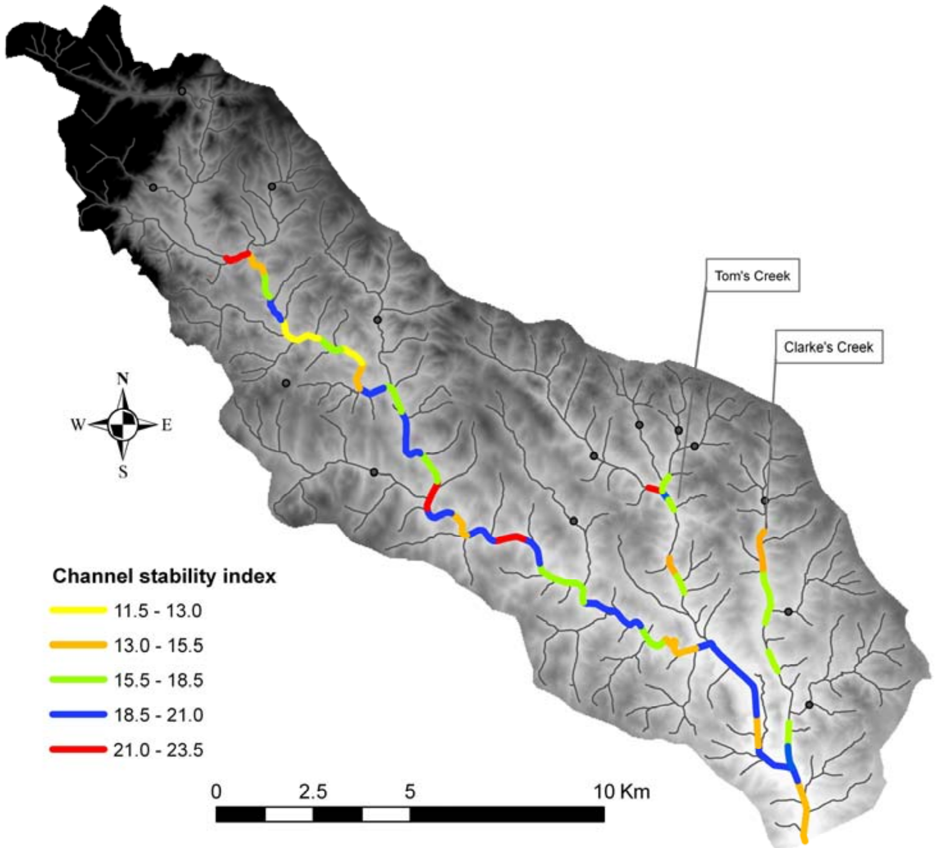

Figure 2.3 Map showing channel stability index

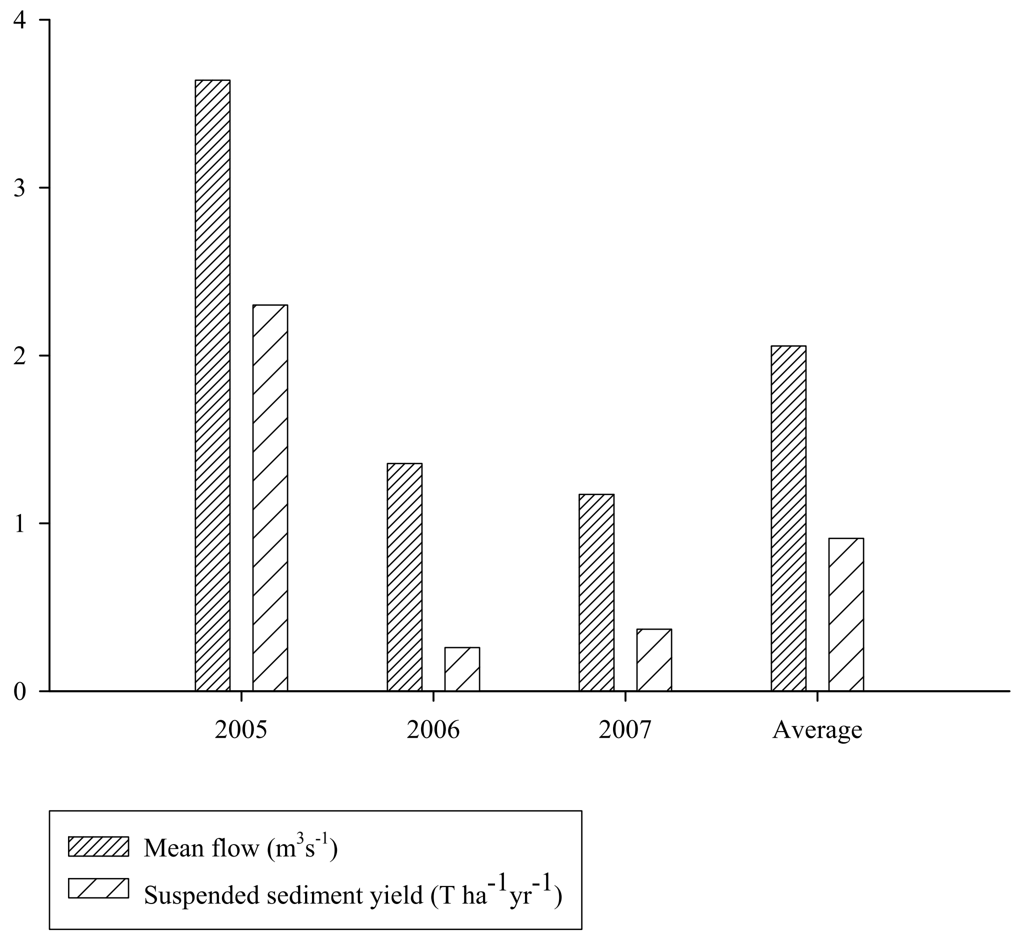

Figure 2.4 Annual flow and suspended sediment yield

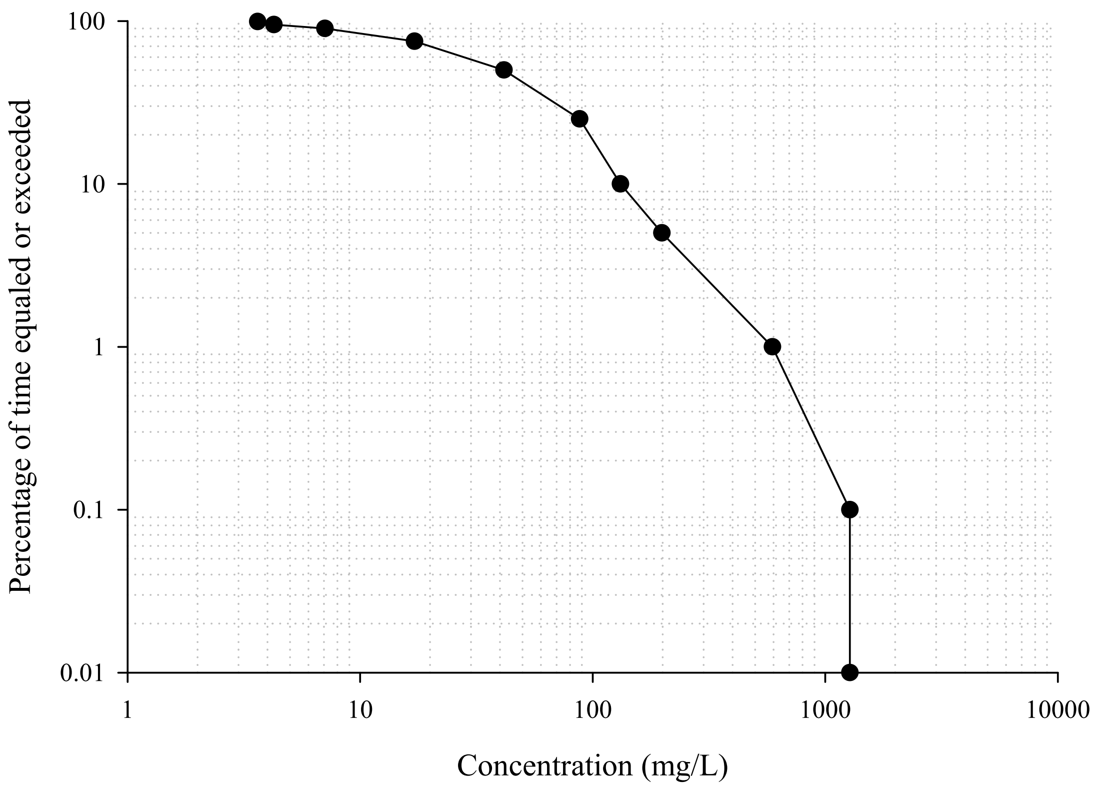

Figure 2.5 Frequency that sediment concentrations were equaled or exceeded

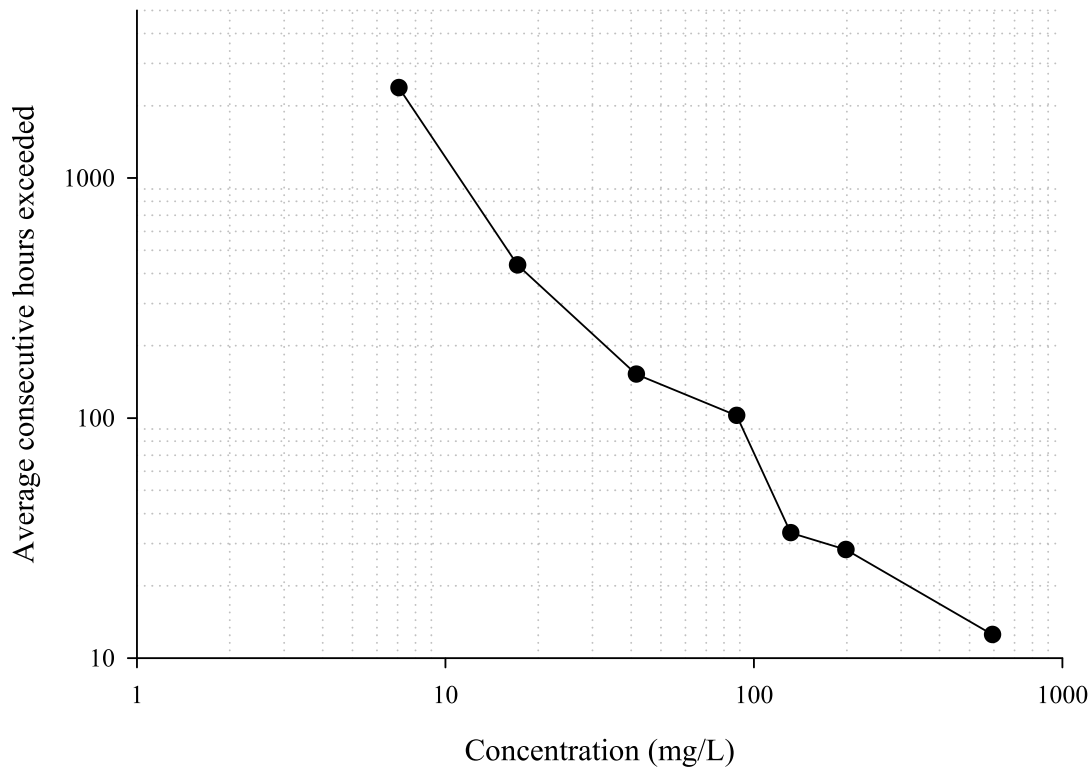

Figure 2.6 Average duration (number of consecutive hours) a given concentration is equaled or exceeded

CHAPTER III

SEDIMENT FINGERPRINTING TO DETERMINE THE SOURCE OF SUSPENDED SEDIMENT IN A SOUTHERN PIEDMONT STREAM

---

[1] Mukundan, R., D. E. Radcliffe, J.C. Ritchie, L. M. Risse and R. McKinley. To be submitted to the *Journal of Environmental quality*.

ABSTRACT

Thousands of stream miles in the southern Piedmont region are impaired because of high levels of suspended sediment. For these waters, it is not clear if the source is upland erosion from agricultural sources or bank erosion of historic sediment deposited in the flood plains between 1830 and 1930 when cotton farming was extensive. The objective of this study was to determine the source of high stream suspended sediment concentrations in a typical southern Piedmont watershed using sediment fingerprinting techniques. Twenty one potential tracers were tested for the ability to discriminate between sources, conservative behavior, and lack of redundancy. Tracer concentrations were determined in potential sediment sources (forests, pastures, row crop fields, stream banks, and unpaved roads/construction sites) and suspended sediment samples collected from the stream and analyzed using mixing models. Results indicated that ^{137}Cs and ^{15}N were the best tracers to discriminate potential sediment sources in this watershed. The δ^{15}N values showed distinct signatures in all the potential sediment sources and it was found to be a unique tracer to differentiate bank soil from upland sub-surface soils such as soil from construction sites, unpaved roads, ditches, and field gullies. Mixing models showed that about 60% of the stream suspended sediment originated from eroding stream banks, 23-30% from upland subsoil sources such as construction sites and unpaved roads and about 10-15% from pastures. The results may be applicable to other watersheds in the Piedmont depending on the extent of urbanization occurring in these watersheds. Better understanding of the sources of fine sediment has practical implications on the type of sediment control measures to be adopted. Investment of resources in improving

water quality should consider the factors causing stream bank erosion and erosion from unpaved roads/construction sites to water quality impairment.

INTRODUCTION

Many streams in the United States do not meet the water quality standards set forth by the states; 17% of the streams have high levels of suspended sediment (USEPA, 2006). Piedmont streams are no exception. Plans are being developed for reducing sediment loads in these streams under the Total Maximum Daily Load (TMDL) program (USEPA, 2008). However, it is not clear if the impairment are due to current upland erosion sources (such as agricultural fields, roads, ditches, and construction sites) or bank erosion of *legacy sediment* deposited in the flood plains during the period of intensive cotton farming from about 1830 to 1930 (Trimble, 1974). Other causes of high erosion were placer mining for gold (Leigh, 1994), eradication of beaver due to the market for pelts (Naiman et al., 1994) and construction of mill dams (Walter et al., 2008). For restoration work, it is important to know the sources of sediments and their relative contribution. Southern Piedmont streams may be an extreme example, but most other streams in the United States also experienced a period of excessive erosion during the late 19th and early 20th century when large areas of land were cleared for intensive farming (Simon and Rinaldi, 2006). Stream channels became clogged with sediment, fallen trees, and beaver dams and were prone to flooding. In response, federal agencies such as the Soil Conservation Service and local drainage districts dredged and straightened (channelized) streams. This caused a number of unforeseen changes in stream channels.

Alterations to natural stream channels occurring due to natural or anthropogenic causes can affect the rates of sediment transported through the channel. Hydrologic modifications (e.g., dams) can cause sediment deficits that result in stream channel scour and destruction of habitat structure (Waters, 1995). Urbanization is one form of alteration that increases water discharge and channelization is another form that potentially increases the channel slope (Doyle and Shields, 2000).

Streams of the southern Piedmont experienced three major disturbances in the mid 20th century. The most important change was probably the reduction in upland erosion and runoff that occurred as cotton farming was abandoned and fields were converted to pasture land and forests during the cotton era (1830-1930) (Trimble, 1974). A second change was the construction of flood control reservoirs along tributaries to the main stem to in the 1960s and 1970s. The third disturbance was channelization in the floodplains during the same period. These disturbances increased stream power in the main stem and caused channels to go through a period of incision and accelerated bank erosion. The banks in the floodplains were especially prone to erode because they consisted of the non-cohesive historic sediment deposited during the cotton era. Whether these streams have reached a new stable equilibrium or are still unstable and in the process of transporting legacy sediments in response to the disturbances in the middle of the last century are not known. Urbanization is a new form of disturbance that is affecting some watersheds through high storm flow peaks resulting in high sediment loads (Landers et al., 2007).

These disturbances produced geomorphologic changes to stream channels including enlargement of channels and longer recurrence intervals for bank full discharge especially in the upper reaches. As a result of these changes there is less stream-floodplain interaction and most of the stream power is dissipated within the channel through bank erosion processes (Ruhlman and Nutter, 1999). Jackson et al. (2005) observed that lower order stream reaches of Murder Creek, another Georgia Piedmont stream had more incised and unstable stream banks than higher order stream reaches. Ntumngia (2001) found evidence of significant channel widening in the main stem of three major rivers (Broad, Oconee and Etowah) in the Georgia Piedmont during the period from 1938 to 1999. However, stream bed elevations were found to be stable with little or no change during the period of study. The current primary source of suspended sediment and the relative proportions of bank vs. upland sources in Piedmont streams remain unknown. This is an important distinction to be made for developing sediment target loads and load reduction scenarios.

Sediment fingerprinting has proven to be an effective way to track sediment movement within a watershed in terms of both source type and spatial origin (Walling, 2005). The procedure involves characterizing the potential sediment sources in terms of their diagnostic chemical and physical properties and then comparing these properties to that of stream sediment. The fingerprint properties should be measurable in the sources as well as the stream sediment, representative of a particular source and should be conservative between sediment generation and delivery. The properties that have been used for sediment source tracking include sediment color (Grimshaw and Lewin, 1980), plant pollen (Brown, 1985), mineral magnetic properties (Walden et al., 1997), rare earth elements (Kimoto et al., 2006), fallout radionuclides (Collins and Walling, 2002; Nagle and Ritchie, 2004; Walling, 2005), and stable isotopes of carbon and nitrogen (Papanicolaou et al., 2003; Fox and Papanicolaou, 2007).

Most studies relating to sediment fingerprinting have focused on the use of fallout radionuclides. The most commonly used radionuclide is cesium-137 (^{137}Cs) having a half life of 30.2 years. Radioactive ^{137}Cs was produced during atmospheric testing of nuclear weapons in the 1950s and 1960s. Global fallout of ^{137}Cs peaked in the early 1960s and subsequently decreased reaching zero levels in the mid 1980s (Walling, 2004). In cultivated soils, the ^{137}Cs distribution tends to be uniform to the depth of tillage whereas in uncultivated soils the peak concentration occurs at a depth of about 5-8 cm and it falls to zero at about 25-30 cm. Because there are no natural sources in the environment, ^{137}Cs serves as a unique tracer for erosion and sedimentation (He and Owens, 1995).

Other fallout radionuclides commonly used include lead-210 (^{210}Pb) and beryllium-7 (^{7}Be). Unlike ^{137}Cs, these two radionuclides are natural in origin and their creation in the atmosphere is fairly constant over time. Lead-210 is a product of the ^{238}U decay series and has a half-life of 22.26 years. With behavior similar to that of ^{137}Cs in soils, ^{210}Pb can be used as an alternative in erosion studies (Zapata, 2003). Beryllium-7 is cosmogenic in origin through the spallation of nitrogen and oxygen atoms in the troposphere and stratosphere by cosmic rays. Compared to ^{137}Cs and ^{210}Pb, ^{7}Be has a half-life of 53.3 days that makes it suitable for short term soil erosion studies. Typically concentrated in the upper five millimeters of the soil profile, it can provide good discrimination between sediment derived from surface soils and those from deeper layers (Zapata, 2003).

In 1998, the North Fork Broad River (NFBR), in northeast Georgia, was included in the 303(d) list for impaired biota and habitat. Sediment was determined to be the pollutant of concern. The stream was placed on the list as part of a consent decree in a lawsuit filed against the United States Environmental Protection Agency (USEPA) and the Georgia Environmental Protection Division (Sierra Club *v*. Hankinson, 2003). The listing was based on an assessment of land-use in the watershed that concluded there was a high probability for impacted biota and habitat, although no sampling of the stream was conducted. Therefore, the USEPA developed a Total Maximum Daily Load (TMDL) for sediment for the North Fork of the Broad River. The USEPA TMDL report (2000) recommended that additional data be collected to better define the sediment loading from non-point sources. A stakeholder group involving farmers, county agents, nonprofit groups and other local and state agencies was formed to identify sediment sources and implement a watershed restoration plan. In 2004 after conducting a macro invertebrate survey, the USEPA removed the NFBR watershed from a 303(d) list and reported that "habitat concerns are present but not to an extent impacting the biota". However, no measurements of the sediment concentrations or discharge were made so the annual sediment load in the NFBR remained unknown. A Clean Water Act Section 319 grant was awarded in 2004 to fund BMP implementation and a program to monitor stream sediment loads. Working with others, we received a grant in 2007 to test a 3-pronged approach for potentially unstable southern Piedmont streams with high sediment loads: 1) sediment yield analysis and rapid geomorphic assessment to determine stability class, 2) sediment fingerprinting to determine sediment sources, and 3) sediment modeling of BMP scenarios. The watershed is located in Franklin and Stephens counties in the

Piedmont region of Georgia and drains an area of 182 km^2 (Figure 3.1). The major land uses in the watershed are forest (72%), pasture (15%), row crops (7%), and residential (1%). This is a typical land use pattern in the southern Piedmont region. In Chapter II, we reported on the sediment yield estimates and rapid geomorphic assessment of the NFBR. The sediment yield estimates for this watershed were found to be high when compared to the median value for the Piedmont region. Geomorphic assessment of stream channels indicated that majority of the stream reaches were unstable.

The objective of this study was to determine if the present load of sediment in the NFBR is from current sources (upland soils) or from historic sources (stream banks). We hypothesize that the stream channels are unstable and bank erosion is the major source of sediment. To the best of our knowledge, there has not been a study to fingerprint suspended sediment in watersheds of the southern Piedmont region.

MATERIALS AND METHODS

The radionuclide tracer used in this study was ^{137}Cs. Our hypothesis was that soil surface samples (representing current erosion sources) will have relatively high activity due to fallout and soil samples from stream banks in flood plains (representing historic sources which were buried before the bomb era) will have relatively low activity. Potential sediment sources identified in the watershed included surface soil sources (croplands, pastures, and forested areas) and sub-surface soil sources (stream banks, unpaved roads and construction sites). A total of 165 composite soil samples representing potential sediment sources were collected from spatially distributed locations in the watershed for tracer analysis (Figure 3.1). Upland soil samples were collected from the upper 0-2 cm depth. Bank samples were collected from the bank face of actively eroding regions identified in the channel. The height of the bank varied from 1 m to over 15 m at different locations of the watershed. Hence bank samples were collected by scraping soil from bank faces that are potentially erodible under the current stream flow regime. Samples were collected from regions close to the water surface to about one meter above the water surface. The samples were air-dried, sieved through a 2-mm sieve and analyzed for ^{137}Cs using a gamma ray spectrometry system with a high purity germanium detector at the USDA ARS Hydrology and Remote Sensing Laboratory, Beltsville, Maryland.

Other tracers used in this study were total C, N, P, S and trace elements (Be, Mg, Al, K, Ca, Cr, Mn, Fe, Co, Ni, Cu, Zn, As, Pb, and U). For total C, N, and S, soil samples were combusted in an oxygen atmosphere at 1350ºC, converting elemental carbon, sulfur, and nitrogen into CO_2, SO_2, and N_2. These gases were then passed through the IR (infrared) cells to determine the C and S content and a TC (thermal conductivity) cell to determine N_2. For trace elements and total P analysis, the samples were digested using nitric acid and the digest was brought to volume using deionized water and analyzed using ICP-MS. These analyses were done at the Laboratory for Environmental Analysis and the Soil Testing Lab, University of Georgia. For better discrimination between sub surface sources (bank vs. construction sites and unpaved roads) the stable isotope of nitrogen ^{15}N was used as a bio-geochemical organic fingerprint. Fox and Papanicolaou (2008) describe the applicability of ^{15}N to fingerprint sediment coming from source variables such as land-use, land management, geomorphology, and soil depth at a watershed scale. The stable isotope of nitrogen is expressed relative to the atmospheric nitrogen in “delta” (δ) notation indicating the difference between the sample isotopic ratio and the ratio in the standard as:

$$\delta^{15}N = \left[\frac{(^{15}N/^{14}N)_{\text{Sample}}}{(^{15}N/^{14}N)_{\text{Standard}}} - 1 \right] 10^3$$

where $\delta^{15}N$ is expressed in per mil (‰) or parts per thousand. Soil and sediment samples were finely ground in a ball mill and from the homogenized sample about 25 mg was analyzed for ^{15}N using mass spectrometry. This analysis was done at the Analytical Chemistry Lab, Odum School of Ecology, University of Georgia.

Soil and sediment samples collected from a wide range of locations may differ in texture. As a result, tracer concentrations can vary due to the relative proportion of the fine fraction, i.e. clay and silt. Hence textural analysis was done on all the soil and sediment samples for expressing the tracer concentration in terms of the fine fraction in the samples. This ensured that the suspended sediment samples and the soil samples collected from the banks and uplands were comparable.

Most of the stream transport of suspended sediment occurs during storms, so it is critical to sample streams during storm events. The conventional method of suspended sediment sampling involves pumping large volumes of water samples (100-400 L) from which about 20-100 g of suspended sediment is collected by centrifuging (Walling et al., 1993). This can provide a composite sample with contributions from the different sources. In this study suspended sediment samples were collected during storm events by pumping water out of the stream at the watershed outlet and passing it through a continuous flow centrifuge collector mounted at the back of a pick-up truck. The inlet of the pump hose was attached to a metal fence post installed about 3 m from the bank and water was pumped from a point about 30 cm below the water surface. This method of sampling in comparison to manual filtering ensured that sufficient mass of suspended sediment was collected for all analyses. About 100-200 g of suspended sediment was required for a complete set of all physical and chemical analysis with a high degree of confidence. From the total amount of suspended sediment collected about 50-100 g was used for ^{137}Cs analysis, 1-2 g for trace element analysis, and 40-50 g for textural analysis. For analyzing $\delta^{15}N$ about 5-10 g of the fine sediment was ground and homogenized in a ball mill from which a few milligrams were used. A larger mass of sample ensured better representation of sediment coming from various sources. A total of 20 sediment samples were collected from six different storm events.

Selection of the best suite of fingerprints for sediment source separation was a multi-step process based on based on minimization of Wilk's lambda (Collins and Walling, 2002). In the first step, all tracers were statistically tested for individual ability to separate sources using discriminant analysis (DA). In the second step, non-conservative tracers were removed based on their concentrations in stream sediment. Tracers that showed higher or lower concentrations in sediment samples when compared to all the sources were eliminated in this step. In the final step tracers that showed redundancy were removed while retaining those tracers that could explain most of the source variation. Relative source contribution of suspended sediment was estimated by using the final suite of tracers in a multivariate mixing model (Collins et al., 1998; Owens et al., 1999; Walling et al., 1999).

The method of least squares was used for deriving the source proportions by minimizing the residual sum of squares for the *n* tracer and *m* sources using,

$$RSS = \sum_{i=1}^{n} \left[\frac{C_{sed\,i} - (\sum_{s=1}^{m} C_{s\,i} \cdot P_s)}{C_{sed\,i}} \right]^2$$

where,

RSS = the residual sum of squares

$C_{sed,i}$ = the concentration of the tracer i in the sediment

$C_{s,i}$ = the mean concentration of the tracer property i in the source group s

P_s = the relative proportion from source group s

The reliability of the multivariate mixing model was tested using another method, the End Member Mixing Analysis (EMMA) (Christophersen and Hooper, 1992; Burns et al., 2001). EMMA is a widely used method in hydrology for quantifying sources of stream flow but has not been used in sediment fingerprinting. In general EMMA models are developed through principal component analysis (PCA) with conservative tracers. The median tracer concentration in the potential end-members (sources) and the sediment samples were plotted against each other after scaling the values between 0 and 1 by dividing individual values with the maximum observed value and the extent to which the sediment samples were bound by the end members was examined in 2-D space. Relative contribution of stream sediment from various sources was derived by solving the following mass-balance equation:

$$S_{st} = S_b + S_c + S_p$$

$$U1_{st}S_{st} = U1_bS_b + U1_cS_c + U1_pS_p$$

$$U2_{st}S_{st} = U2_bS_b + U2_cS_c + U2_pS_p$$

where, S represents sediment, and $U1$ and $U2$ are the scaled ^{137}Cs and $\delta^{15}N$ values; the subscripts *st*, *b*, *c*, and *p* represents stream, bank, construction/unpaved roads, and pasture respectively.

Suspended sediment samples collected were from a wide range of stream discharge and sediment concentrations (Table 3.1). Stream discharge and turbidity measurement at the time of sampling were used to calculate an instantaneous load associated with each sample assuming that a nephelometric turbidity unit (NTU) equals mg L^{-1}. A more realistic estimate of relative source contribution from various sources was obtained using the load-weighted method (Walling et al., 1999; Walling, 2005) based on the equation:

$$P_{sw} = \sum_{x=1}^{n} P_{sx}\left(\frac{L_x}{L_t}\right)$$

where, P_{sw} is the load weighted relative contribution from source type s, L_x (kg s^{-1}) is the instantaneous suspended sediment load for sample x, L_t (kg s^{-1}) is the sum of the instantaneous loads associated with the n sediment samples and P_{sx} is the relative contribution from source type s for sediment sample x.

RESULTS

Discriminant analysis showed that only 11 out of the 21 tracers were useful for sediment source separation. The stable isotope of nitrogen ^{15}N was found to be the best tracer for discriminating sediment sources as it was selected first in the stepwise selection procedure (Table 3.1). The radionuclide ^{137}Cs was selected eighth exposing its inability to distinguish between bank soil and upland sub soil material (construction sites, field gullies and unpaved roads). Also, soil samples from pastures and row crop areas showed similar ^{137}Cs signatures. Tracers that were retained included total C, N, and S, ^{137}Cs, ^{15}N, Al, Cr, Fe, Pb, Mg and U. From this list non-conservative (Al, Pb, Fe and Mg) and redundant (C, N, and S) tracers were removed. The final list had only 4 tracers (^{137}Cs, ^{15}N, Cr, and U) that could explain most of the sediment source variation (Table 3.2). However, scatter plots of ^{137}Cs against the other three tracers indicated that Cr and U were not always conservative and therefore only ^{137}Cs and $\delta^{15}N$ were used for the mixing analysis (Figure 3.2). Scatter plots using one of the redundant tracers, carbon and $\delta^{15}N$ clearly indicated that total C may be used as a viable and cost-effective alternative to ^{137}Cs for sediment fingerprinting (Figure 3.3). The tracer was found to be conservative in nature. Strong positive correlations between ^{137}C and soil organic carbon is being used in studies involving soil and soil organic carbon redistribution at the landscape scale (Ritchie and McCarty, 2003; Ritchie et al., 2007).

The concentration of ^{137}Cs was highest in forests followed by pastures suggesting that there is less erosion from forests. As expected a decrease in ^{137}Cs concentration with depth was observed in both forest and pasture soils indicating that most of it was concentrated in the upper 20 cm of the soil profile (Figure 3.4). The difference in ^{137}Cs concentration between sub surface sources (bank vs. construction/roads) was not sufficient to discriminate the two sources. However, ^{15}N showed distinct signatures in these sources (Table 3.2). The highest $\delta^{15}N$ values were found in pastures followed by bank soils. This may be due to the enrichment of ^{15}N in these sources. In pastures, enrichment of ^{15}N occurs due to plant preference for ^{14}N and removal of ^{15}N depleted biomass from the system during harvest. The nitrification process favors ^{14}N resulting in enrichment of ^{15}N in pastures where fertilizers and manures have been added (Fox and Papanicolaou, 2008). The position of banks in the landscape makes them prone to frequent anaerobic conditions that favor denitrification loss of nitrogen, a process during which isotopic fractionation and enrichment of soil ^{15}N occurs. Topographic positions in the landscape subject to wetting and anaerobic conditions can result in denitrification and residual accumulation of ^{15}N (Karamanos and Rennie, 1980). Another possible reason for enrichment of ^{15}N in stream bank soils is the age of organic matter. Generally, older organic matter is associated with relatively enriched isotopic signatures (Billings et al., 2006). Moreover, depth dependent increase in soil $\delta^{15}N$ has been widely observed (Hobbie et al., 1999; Trumbore, 2000; Amundson et al., 2003) and more pronounced in deeper soil layers due to minimal inputs and lack of recirculation of soil organic nitrogen. Similar trends in $\delta^{15}N$ values were observed in the NFBR watershed soils (Figure 3.5). Soils at construction sites and along the unpaved roads are usually exposed subsoils that have been biologically less active and hence have $\delta^{15}N$ values closer to the background. Forested soils showed a net depletion of ^{15}N as evident by a negative median $\delta^{15}N$ value. This may be due to the low input of ^{15}N in forests which is primarily from rainfall and relatively higher ^{14}N content in forest soils. The relatively stable value for $\delta^{15}N$ in forest soils reflects the incorporation of ^{15}N-depleted biomass to the soil surface that limits the enrichment of ^{15}N with time (Billings and Richter, 2006)

The concentration of uranium was highest in the bank material, perhaps indicating a lithogenic signature. Uranium concentrations vary between 2-4 ppm by weight in the earth's crust, rocks, and soils (Gavrilescu et al., 2008). However, concentrations can vary depending on the parent material. Felsic rocks contain more U compared to mafic rocks.

Albright (2004) observed higher concentrations of U in the C horizon compared to the upper layers in the Piedmont region of Georgia dominated by gneissic parent material similar to the parent material in the NFBR watershed. Chromium concentrations were highest in construction sites and unpaved roads indicating an anthropogenic signature. Chromium is used in brake linings, tires, and metal alloys used in automobile engine parts (Paul and Meyer, 2001). Used lubricating oils may contain significant amounts of lead, chromium, nickel, copper, vanadium, as well as organic phosphates (Muschack, 1990). Another possible explanation for this is the natural abundance of Cr in the sub-soils of the Georgia Piedmont. Albright (2004) reported higher background concentrations of Cr in sub soils compared to surface soils in the Georgia Piedmont. An increase in Cr content with depth may also be related to the increase in clay fraction and mobility of Cr in the soil profile (Adriano, 2001).

Both the multivariate mixing model and EMMA indicated that stream banks are the predominant source of suspended sediment in the NFBR watershed followed by upland sub surface sources (Table 3.3). The load weighted method did not produce significant change in results although contribution from upland sub-surface sources increased by 3-4% whereas contributions from stream banks and pasture decreased by 1-2% and 2-3% respectively. Although differences were not observed in the overall relative source contributions predicted by the two models, some differences were observed when individual sediment samples were compared (Figure 3.6 and 3.7). These results indicate that suspended sediment samples from several storm events covering a wide range of flows and sediment concentrations is required to produce reliable estimates of relative sediment source contributions. In that case the differences or error due to a single event/sample data value are nullified.

The suspended sediment contribution from construction sites and unpaved roads seemed to be high considering that only a small proportion of the total watershed area is occupied by these source areas. However, the rates of soil erosion from construction sites and unpaved roads are considerably higher than that of erosion from an agricultural field or pasture. Erosion rates from construction sites in urbanizing watersheds may approach 500 T ha^{-1} yr^{-1} compared to 10-40 T ha^{-1} yr^{-1} for agriculture and less than 1 T ha^{-1} yr^{-1} for undisturbed vegetation (Carpenter et al., 1998). Hayes et al. (2005) reported that the rate of soil erosion from construction sites ranges from very little to over 200 T ha^{-1} yr^{-1}. Data from one of the monitored unpaved road sites in this watershed showed that the turbidity from runoff exceeded 1000 NTU (roughly equivalent to a sediment concentration of 1000 mg L^{-1}) for almost all storms irrespective of the intensity or duration of the storm. Martin (2001) reported that unpaved roads contributed the maximum sediment per unit area in a southern Piedmont watershed and the rates were two orders of magnitude greater than that from pastures in the same watershed.

The contribution of pastures to stream sediment is less when compared to the other two sources and forests did not contribute any sediment. It was interesting to note that the particle size distribution of surface soils differed from sub-surface soils (Table 3.5). The fine fraction was significantly higher in the sub-surface sources indicating the higher potential to generate fine sediment due to anthropogenic activities. The highest clay content was observed in the construction sites and unpaved roads which were found to be a dominant source of fine sediment in this watershed. Among all sources, the highest silt content was observed in the bank soil material. This may be due to the fact that part of the bank material was formed from sediment that eroded from the upland surfaces and deposited in the floodplains during the period of erosive land use in the Piedmont. Silt being the most erodible fraction thus concentrated in the banks. The textural composition of the bank material showed large differences at different regions in the watershed (sand 17-87%, silt 10-48% and clay 3-39%) indicating that the stream channels are not entirely composed of eroded top soil from European settlement. This also indicates the differences in depositional environments/histories. Part of the channel may be composed of original in situ parent material. Textural compositions similar to the flood plains of the South Fork of the Broad River reported by Lichtenstein (2003) were mostly observed in the upper reaches characterized by clay contents of less than ten percent.

Overall, the two methods used for sediment source identification produced comparable results. The end member mixing diagram shows that the three sources are readily distinguished and that stream sediment is a mixture of the sources, i.e., it plots within the triangle of the sources (Figure 3.8).

CONCLUSIONS

The results show that there is ample scope for sediment load reduction in this Piedmont watershed. Reducing the contribution of upland sub-surface sources such as construction sites and unpaved roads through conservation measures can significantly reduce sediment loads. The results also show that legacy sediment (bank erosion) is an important source and that will be more difficult to reduce. This complements the rapid geomorphic assessment (RGA) conducted as part of this project which indicated that the majority of the stream channels were unstable. RGA results can be used to prioritize stream bank restoration efforts at watershed scale. The methods used in this study have practical applications in the total maximum daily load (TMDL) program for determining target sediment loads and load reduction scenarios. This study also brings into light the potential of ^{15}N as a tracer to identify multiple sources. A single tracer may be able to identify multiple sediment sources in some watersheds. Moreover, $\delta^{15}N$ values are not affected by particle size distribution of the sources and the sediment and hence do not require particle size correction which is an issue in sediment fingerprinting studies. Results clearly indicate that total C may be used as a viable and cost-effective alternative to the more expensive ^{137}Cs for sediment fingerprinting. By using a continuous flow mobile centrifuge system we were able to overcome one of the greatest limitations in all fingerprinting studies, i.e., the inability to collect a sufficient mass of suspended sediment for all physical and chemical analysis.

Rapid urbanization is occurring in many watersheds in the Piedmont and other regions of the United States. Tracers like ^{15}N that show a distinct signature for construction sites may be used to determine the contribution of land disturbance activities to water quality impairment in these areas. Mixing models can be used to estimate the relative source contribution provided the tracers used are conservative in nature.

We have every reason to believe that the NFBR watershed is a typical watershed in the southern Piedmont and hence the results may be applicable to other watersheds in the southern Piedmont. This will depend on the level of urbanization/land disturbance activities that result in higher sediment loads through higher storm flow peaks. Sediment source identification has practical implications on soil erosion control strategies because the methods used for surface erosion control are different from that of bank erosion control. Any investment in improving water quality should consider the contribution of sub-surface sources to water quality impairment. The holistic approach adopted in this study can be easily adapted to other watersheds or regions of varying scales depending on the availability of resources.

Acknowledgement

This study was supported by the USDA-CSREES grant # 2007-51130-03869 (A New Approach to Sediment TMDL Watersheds in the Southern Piedmont).

REFERENCES

Adriano, D. C., 2001. Trace elements in terrestrial environments: Biogeochemistry, bioavailability, and risks of metals, Springer, New York.

Albright, E. I., 2004. Background Concentrations of Trace Elements in Soils and Rocks of the Georgia Piedmont, Unpublished M.S thesis, University of Georgia.

Amundson, R., A. T. Austin, E. A. G. Schuur, K. Yoo, V. Matzek, C. Kendall, A. Uebersax, D. Brenner and W. T. Baisden, 2003. Global patterns of the isotopic composition of soil and plant nitrogen. *Global Biogeochemical Cycles* 17:1031 doi:10.1029/2002GB001903.

Billings, S. A. and D. D. Richter, 2006. Changes in stable isotopic signatures of soil nitrogen and carbon during 40 years of forest development. *Oecologia* 148:325-333.

Billings, S. A., C. M. Brewer and B. L. Foster, 2006. Incorporation of plant residues into soil organic matter fractions with grassland management practices in the North American Midwest. *Ecosystems* 9:805-815.

Brown A. G. 1985. The potential use of pollen in the identification of suspended sediment sources. *Earth Surface Processes and Landforms*. 10: 27–32.

Burns, D. A., J. J. McDonnell, R. P. Hooper, N. E. Peters, J. E. Freer, C. Kendall and K. Beven, 2001. Quantifying contributions to storm runoff through end-member mixing analysis and hydrologic measurements at the Panola Mountain Research Watershed (Georgia, USA). *Hydrological Processes* 15:1903-1924.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley and V. H. Smith, 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8:559-568.

Christophersen, N. and R. P. Hooper, 1992. Multivariate analysis of stream water chemical data: the use of principal components analysis for the end-member mixing problem. *Water Resources Research* 28 99-107.

Collins, A. L., and D. E. Walling, 2002. Selecting fingerprint properties for discriminating potential suspended sediment sources in river basins. *Journal of Hydrology*. 261:218-244.

Collins, A. L., D. E. Walling and G. J. L. Leeks, 1998. Use of composite fingerprints to determine the provenance of the contemporary suspended sediment load transported by rivers. *Earth Surface Processes and Landforms* 23:31-52.

Doyle, M. W. and F. D. Shields, 2000. Incorporation of bed texture into a channel evolution model. *Geomorphology* 34:291-309.

Fox, J. F. and A. N. Papanicolaou, 2008. Application of the spatial distribution of nitrogen stable isotopes for sediment tracing at the watershed scale. *Journal of Hydrology* 358:46-55.

Fox, J. F. and A. N. Papanicolaou. 2007. The use of carbon and nitrogen isotopes to study watershed erosion processes. *Journal of the American Water Resources Association* 43:1047-1064.

Gavrilescu, M., L. V. Pavel and I. Cretescu, 2008. Characterization and remediation of soils contaminated with uranium. *Journal of Hazardous Materials* 163:475-510.

Grimshaw, D. L. and J. Lewin, 1980. Source identification for suspended sediments. *Journal of Hydrology* 47:151-162.

Hayes, S. A., R. A. McLaughlin and D. L. Osmond, 2005. Polyacrylamide use for erosion and turbidity control on construction sites. *Journal of Soil and Water Conservation* 60:193-199.

He, Q. and P. Owens. 1995. Determination of suspended sediment provenance using caesium-137, unsupported lead-210 and radium-226: A numerical mixing model approach, pp. 207-227. in I. D. L. Foster and B. W. Webb (eds.). Sediment and water quality in river catchments. John Wiley and Sons, Ltd.

Hobbie, E. A., S. A. Macko and H. H. Shugart, 1999. Insights into nitrogen and carbon dynamics of ectomycorrhizal and saprotrophic fungi from isotopic evidence. *Oecologia* 118:353-360.

Jackson, C. R., J. K. Martin, D. S. Leigh and L. T. West, 2005. A southeastern Piedmont watershed sediment budget: Evidence for a multi-millennial agricultural legacy. *Journal of Soil and Water Conservation* 60:298-310.

Karamanos, R. E. and D. A. Rennie, 1980. Changes in natural ^{15}N abundance associated with pedogenic processes in soil. II. Changes on different slope positions. *Canadian Journal of Soil Science* 60:337–344 .

Kimoto, A., M. Nearing, M. J. Shipitalo and V. O. Polyakov. 2006. Multi-year tracking of sediment sources in a small agricultural watershed using rare earth elements. *Earth Surface Processes and Landforms* 31:1763-1774

Landers, M. N., P. D. Ankcorn and K. W. McFadden, 2007 Watershed Effects on Streamflow Quantity and Quality in Six Watersheds of Gwinnett County, Georgia. USGS Scientific Investigations Report 2007-5132.

Leigh, D.S. 1994. Mecury contamination and floodplain sedimentation from former gold mines in north Georgia. *Water Resources Bulletin*. 30:739-748.

Lichtenstein, K. P., 2003. Historic Sedimentation and Allostratigraphy of the South Fork Broad River, Northeast Georgia, Unpublished M.S thesis, University of Georgia.

Martin, J.K. 2001. Sediment Flux and Storage in a Southeastern Piedmont River System. Unpublished M.S. Thesis, University of Georgia, Athens, Georgia. 84 pp.

Muschak, W. 1990. Pollution of street runoff by traffic and local conditions. *Science of the Total Environment* 93:419–31

Nagle, G. N. and J. C. Ritchie. 2004. Wheat field erosion rates and channel bottom sediment sources in an intensively cropped northeastern Oregon drainage basin. *Land Degradation and Development* 15: 15-16.

Naiman, R.J., S.R. Elliott, and J.M. Helfield. 1999. Biophysical interactions and the structure and dynamics of reverine ecosystems: the importance of biotic feedbacks. *Hydrobiologica* 410:79-86.

Ntumngia, N. G., 2001. Analysis of Channel Change in Rivers of the Georgia Piedmont (1938/42/44-1993/99), Unpublished M.S thesis, University of Georgia.

Owens, P. N., D. E. Walling and G. J. L. Leeks, 1999. Use of floodplain sediment cores to investigate recent historical changes in overbank sedimentation rates and sediment sources in the catchment of the River Ouse, Yorkshire, UK. Catena 36:21-47.

Papanicolaou, A. N., J. F. Fox and J. Marshall, 2003. Soil fingerprinting in the Palouse Basin, USA, using stable carbon and nitrogen isotopes. *International Journal of Sediment Research* 18:278-284.

Paul, M. J. and J. L. Meyer, 2001. Streams in the urban landscape. *Annual Review of Ecology and Systematics* 32:333-365.

Ritchie, J. C. and G. W. McCarty, 2003. 137Cesium and soil carbon in a small agricultural watershed. *Soil & Tillage Research* 69:45-51.

Ritchie, J. C., G. W. McCarty, E. R. Venteris and T. C. Kaspar, 2007. Soil and soil organic carbon redistribution on the landscape. *Geomorphology* 89:163-171.

Ruhlman M. B. and W. L. Nutter, 1999. Channel morphology evolution and overbank flow in the Georgia Piedmont. *Journal of the American Water Resources Association* 35:277-290.

Sierra Club v. Hankinson. 2003. United States Court of Appeals for the 11th Circuit. Available online at http://altlaw.org/v1/cases/1128530 (verified 29 October 2008).

Simon, A. and M. Rinaldi, 2006. Disturbance, stream incision, and channel evolution: The roles of excess transport capacity and boundary materials in controlling channel response. *Geomorphology* 79:361-383.

Trimble, S.W. 1974. Man-induced soil erosion on the Southern Piedmont, 1700-1970. Soil Conservation Society of America, Ankeny, IA., 180 pp.

Trumbore, S., 2000. Age of soil organic matter and soil respiration: radiocarbon constraints on belowground C dynamics. *Ecological Applications* 10:399-411.

USEPA 2008. Overview of impaired waters and Total Maximum Daily Loads (TMDL) program. Available at http://www.epa.gov/owow/tmdl/intro.html verified (April 24, 2009)

USEPA 2006. Water Quality Assessment and Total Maximum Daily Loads (TMDL) information. Available at http://www.epa.gov/waters/ir/index.html verified (April 13, 2009)

USEPA, 2000 Total Maximum Daily Load Development for the North Fork Broad River Watershed, EPA region 4, Water Management Division Available at http://www.gaepd.org/Files_PDF/techguide/wpb/TMDL/Savannah/EPA_NorthForkBroadRiver_Sediment_TMDL.pdf (verified February 2009)

Walden, J., M. C. Slattery and T. P. Burt 1997. Use of mineral magnetic measurements to fingerprint suspended sediment sources: approaches and techniques for data analysis. Journal of Hydrology. 202, 353-372

Walling D. E., J. C. Woodward, and A. P. Nicholas, 1993. A multi-parameter approach to fingerprinting suspended-sediment sources. In Peters, N.E. (ed.), Tracers in Hydrology,. IAHS Publ. No. 215, Wallingford, UK.

Walling, D. E., 2005. Tracing suspended sediment sources in catchments and river systems. *Science of the Total Environment* 344:159-184.

Walling, D. E., 2004. Using environmental radionuclides to trace sediment mobilization and delivery in river basins as an aid to catchment management, p. 120-135. Proceedings of the Ninth International Symposium on River Sedimentation Oct 18-21, 2004 Yichang, China.

Walling, D. E., P. N. Owens and G. J. L. Leeks, 1999. Fingerprinting suspended sediment sources in the catchment of the River Ouse, Yorkshire, UK. *Hydrological Processes* 13:955-975.

Walter, R. C. and D. J. Merritts, 2008. Natural streams and the legacy of water-powered mills. *Science* 319:299.

Waters, T. F., 1995. Sediment in streams: sources, biological effects, and control. American Fisheries Society Monograph 7, American Fisheries Society, Bethesda, MD. pp. 251.

Zapata, F., 2003. The use of environmental radionuclides as tracers in soil erosion and sedimentation investigations: recent advances and future developments. *Soil and Tillage Research* 69:3-13.

Table 3.1 Stream discharge, turbidity, and hydrographic position at the time of sediment sampling

Sample No	Storm No	Date	Hydrograph	Discharge (m^3s^{-1})	Turbidity (NTU)
1	1	8/26/2008	Rising	1.5	91
2	1	8/26/2008	Rising	1.9	126
3	1	8/26/2008	Rising	2.3	157
4	1	8/27/2008	Falling	11.3	557
5	1	8/27/2008	Falling	10.0	456
6	2	10/9/2008	Falling	1.7	96
7	2	10/9/2008	Falling	1.7	86
8	3	1/7/2009	Falling	32.5	299
9	3	1/7/2009	Falling	29.9	279
10	3	1/7/2009	Falling	26.3	267
11	3	1/7/2009	Falling	22.6	236
12	3	1/7/2009	Falling	20.5	207
13	4	3/1/2009	Rising	7.8	250
14	5	3/16/2009	Falling	2.1	38
15	6	3/28/2009	Rising	9.9	166
16	6	3/28/2009	Rising	10.1	164
17	6	3/28/2009	Rising	11.4	145
18	6	3/28/2009	Rising	13.2	164
19	6	3/29/2009	Falling	8.4	131
20	6	3/29/2009	Falling	7.7	109

Table 3.2 Stepwise Discriminant Analysis (DA) used for tracer selection

No. of variables	Variables	Variable IN	Partial R^2	F	Wilks' Lambda	Pr < Lambda
1	N15	N15	0.635	62.205	0.365	< 0.0001
2	N / N15	N	0.522	38.815	0.174	< 0.0001
3	C / N / N15	C	0.335	17.757	0.116	< 0.0001
4	C / N / S / N15	S	0.315	16.100	0.079	< 0.0001
5	C / N / S / Cr / N15	Cr	0.213	9.404	0.062	< 0.0001
6	C / N / S / Cr / U / N15	U	0.167	6.899	0.052	< 0.0001
7	C / N / S / Al / Cr / U / N15	Al	0.166	6.835	0.043	< 0.0001
8	Cs / C / N / S / Al / Cr / U / N15	Cs	0.190	7.984	0.035	< 0.0001
9	Cs / C / N / S / Mg / Al / Cr / U / N15	Mg	0.081	2.958	0.032	< 0.0001
10	Cs / C / N / S / Mg / Al / Cr / Pb / U / N15	Pb	0.072	2.599	0.030	< 0.0001
11	Cs / C / N / S / Mg / Al / Cr / Fe / Pb / U / N15	Fe	0.073	2.607	0.028	< 0.0001

Table 3.3 Median tracer concentrations*, standard errors (SE) and coefficients of variation (CV)

Tracer	Bank n=60	Construction/ Unpaved roads n=30	Forest n=30	Pasture n=30	Row crops n=15
^{137}Cs(Bq kg^{-1})	3.04	1.82	73.83	14.23	17.40
SE	0.61	0.82	5.70	1.38	1.62
CV(%)	124	133	44	51	36
$\delta\ ^{15}N$(‰)	5.61	0.89	-0.35	8.61	6.29
SE	0.24	0.36	0.36	0.56	0.62
CV(%)	31	145	-	42	41
Cr(mg kg^{-1})	43.92	80.74	53.56	13.79	28.03
SE	4.77	10.74	11.61	3.70	8.72
CV(%)	72	74	87	96	83
U(mg kg^{-1})	4.33	3.07	2.34	3.00	2.57
SE	0.47	0.26	0.43	0.31	0.19
CV(%)	69	45	77	52	29

* expressed in terms of fines (silt + clay)

Table 3.4 Relative sediment source contribution estimated by multivariate mixing model and end member mixing analysis (EMMA) [1]Load weighted method [2]Average of individual sediment samples

	Multivariate mixing model[1]	Multivariate mixing model[2]	EMMA[1]	EMMA[2]
Bank	60	62	60	61
Construction, unpaved roads, field gullies and ditches	27	23	30	27
Pasture	13	15	10	12

Table 3.5 Particle size distribution in source soil samples (Numbers are median values and may not add up to 100)

	Bank n=60	Construction/ Unpaved roads n=30	Forest n=30	Pasture n=30
Sand	59	51	72	70
Silt	28	17	18	20
Clay	18	32	8	10

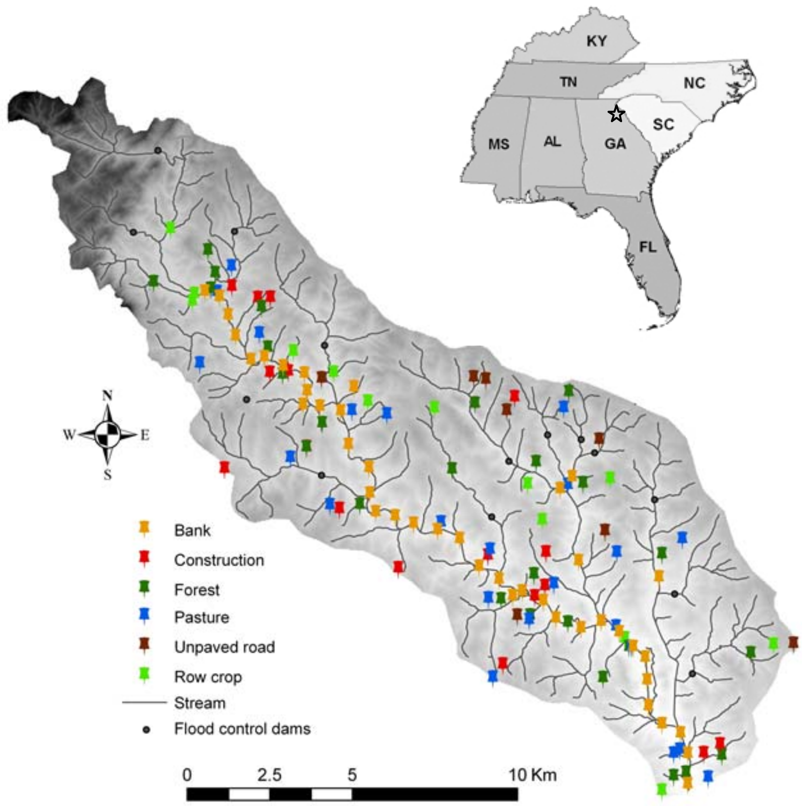

Figure 3.1 Location of the watershed showing sediment source sampling points

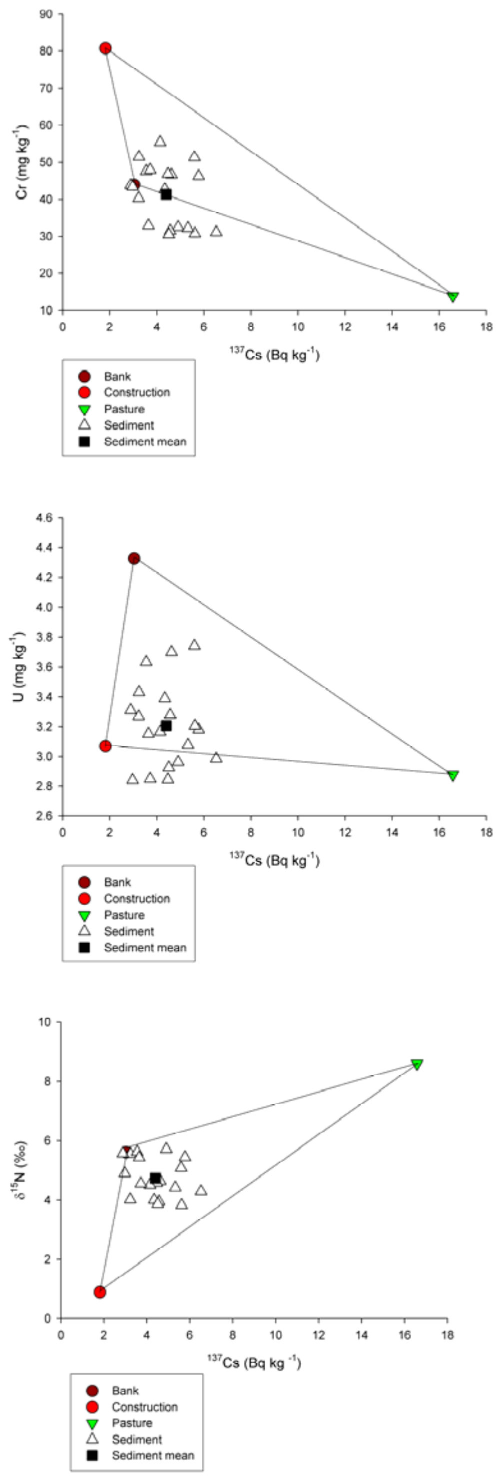

Figure 3.2 Scatter plots of ^{137}Cs against Cr, U, and δ^{15}N in sources and sediment

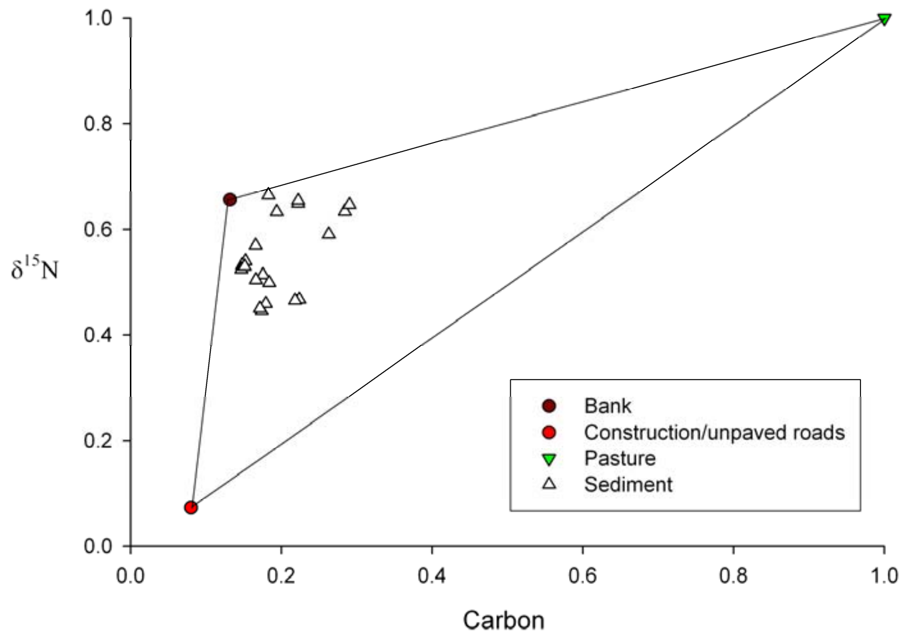

Figure 3.3 Scatter plot of carbon against $\delta^{15}N$

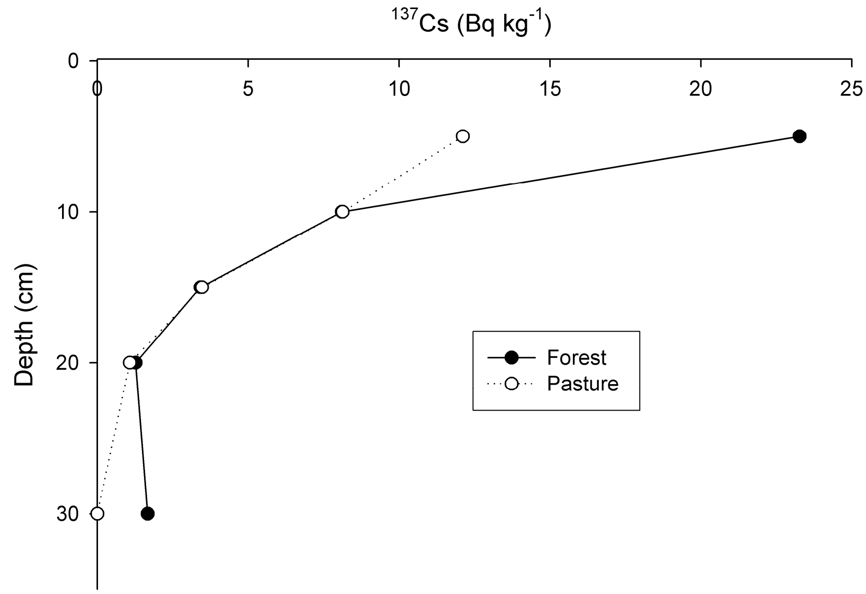

Figure 3.4 Variation in ^{137}Cs along a soil profile

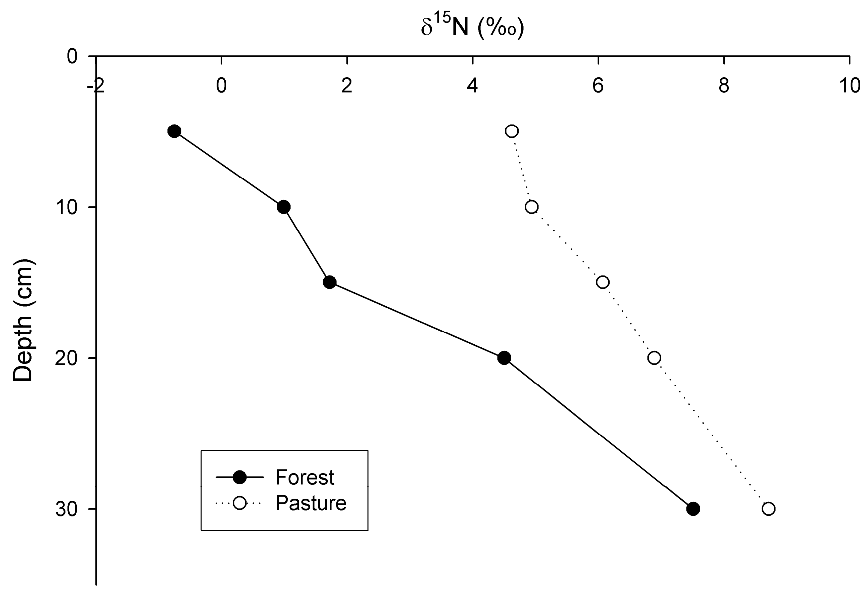

Figure 3.5 Variation in $\delta^{15}N$ along a soil profile

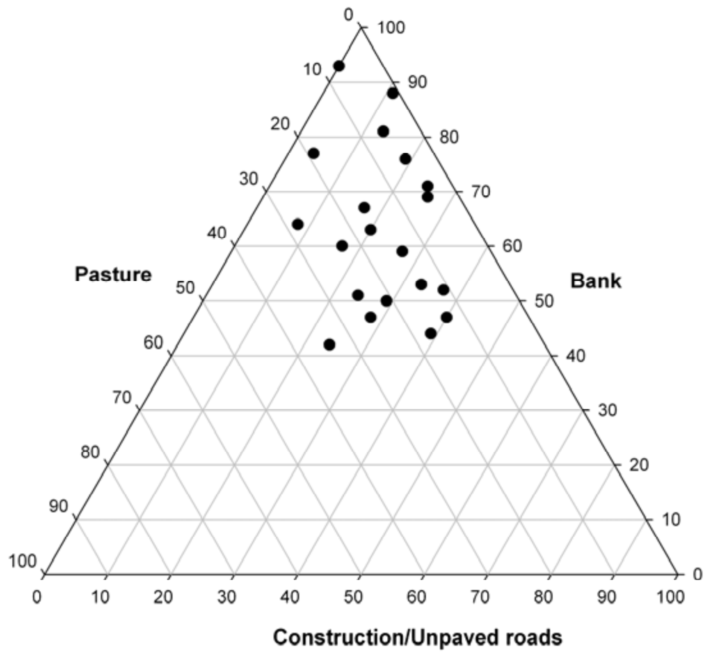

Figure 3.6 Relative source contributions predicted by the multivariate mixing model

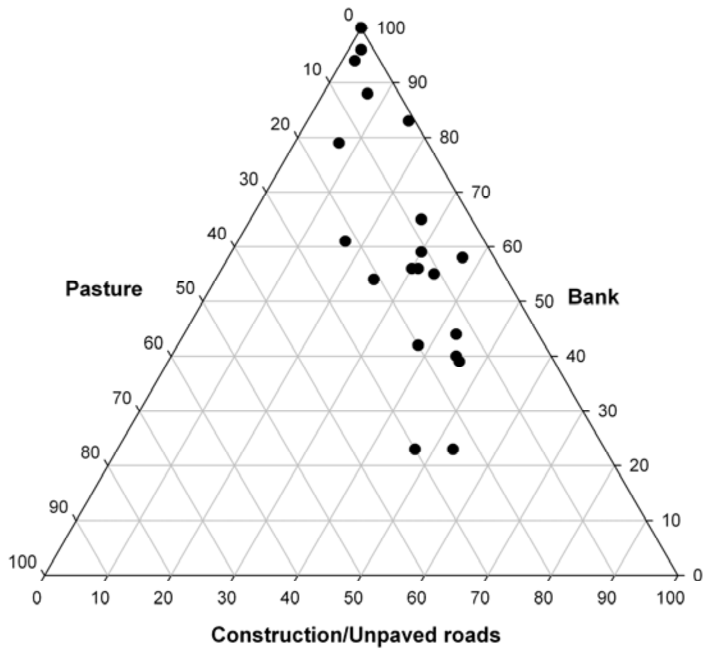

Figure 3.7 Relative source contributions predicted by EMMA

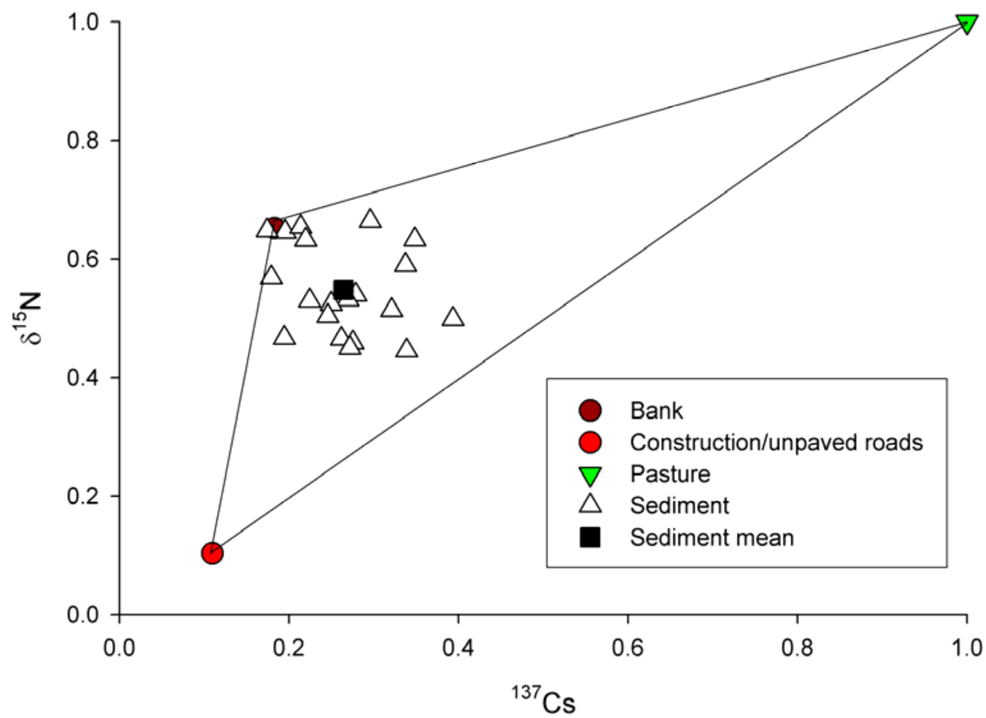

Figure 3.8 Three end member mixing diagram

CHAPTER IV

SPATIAL RESOLUTION OF SOIL DATA AND CHANNEL EROSION EFFECTS ON SWAT MODEL PREDICTIONS OF FLOW AND SEDIMENT

---

[1] Mukundan, R. and D. E. Radcliffe. Submitted to the *Journal of Soil and Water Conservation,* 02/05/2009.

ABSTRACT

Water quality modeling efforts for developing total maximum daily loads (TMDLs) often use GIS data of varying quality in watershed-scale models and have shown varying impacts on model results. Several streams in the southern Piedmont are listed for sediment TMDLs. The objective of this study was to test the effect of spatial resolution of soil data on SWAT model predictions of flow and sediment and to calibrate the SWAT model for a watershed dominated by channel erosion. The state soil geographic database (STATSGO) mapped at 1:250,000 scale was compared with the soil survey geographic database (SSURGO) mapped at 1:12,000 scale in an ArcSWAT model of the North Fork Broad River in Georgia. Model outputs were compared for the effect due to soil data before calibration using default model parameters as calibration can mask the effect of soil data. The model predictions of flow and sediment by the two models were similar and the differences were statistically insignificant (α=0.05). These results were attributed to the similarity in key soil property values in the two databases that govern stream flow and sediment transport. The two models after calibration had comparable model efficiency in simulating stream flow and sediment loads. The calibrated models indicated that channel erosion contributed most of the suspended sediment in this watershed. It is important to know that less detailed soil data can be used because more time, effort, and computational resources are required to set up and calibrate a model with more detailed soil data, especially in a larger watershed.

INTRODUCTION

Soil data is a crucial input for any hydrologic simulation model. Soil properties such as erodibility and hydraulic conductivity affect processes such as infiltration and surface transport of water and pollutants. Accuracy of a hydrologic model depends on the scale at which these soil properties are represented, provided that there is considerable spatial variability in these properties across the landscape being modeled. The commonly available soil databases for the United States are STATSGO and SSURGO. Developed and distributed by the Natural Resources Conservation Services (NRCS) in digital format, these databases can be used to derive soil information for watershed-scale modeling of stream flow and pollutants. The State Soils Geographic (STATSGO) database is mapped at 1:250,000 scale with the smallest mapping unit of about 625 ha and is used for large scale planning (USDA-SCS, 1994). The Soil Survey Geographic (SSURGO) database is mapped at 1:12,000 to 1:63,000 scale with the smallest mapping unit represented at 2 ha and based on a detailed soil survey (USDA-NRCS, 2004).

The Soil and Water Assessment Tool (SWAT), a widely used, physically based watershed-scale model for water and pollutants, uses the STATSGO database as the default dataset for soil information. With a little pre-processing of the SSURGO soil database using the SSURGO SWAT 2.x extension for ArcView by Peschel et al. (2003), SSURGO data can be used for SWAT modeling. However, the use of the detailed soil database is more time and resource intensive.

In a study on the influences of soil data resolution on hydrologic modeling in the Upper Sabinal River Watershed near Ulvade, Texas, uncalibrated SWAT model outputs from STATSGO and SSURGO data were compared (Peschel et al., 2006). Results showed that the SWAT model predictions of flow were higher when SSURGO data was used. The higher water yield was attributed to higher saturated hydraulic conductivity values associated with the SSURGO database. Geza and McCray (2008) compared the effect of soil data resolution on the SWAT model prediction of flow and water quality parameters in the Turkey Creek watershed, a mountainous watershed in Denver, Colorado. The surface elevation ranged from about 1800 to 3200 m and the soils had low infiltration capacity. Comparison was made before calibration because calibration may mask the differences due to soil data resolution. Like Peschel et al. (2006), they found that SSURGO data predicted more flow compared to STATSGO. However, in contrast to flow, STATSGO predicted more sediment and this was attributed to the higher area-weighted average value of soil eroditility (k_{usle}) in the STATSGO database. Gowda and Mulla (2005) calibrated a spatial model for flow and water quality parameters using STATSGO and SSURGO data for High Island Creek, an agricultural watershed in south-central Minnesota characterized by flat topography and poorly drained soils. Statistical comparison of calibration results with measured data indicated excellent agreement for both soil databases.

In a study on the effect of soil data resolution on SWAT model snowmelt simulation, output from SSURGO and STATSGO models were compared using calibrated results for flow for the Elm River watershed in North Dakota characterized by clay and clay loam soils and low topographic relief (Wang and Melesse, 2006). Results indicated that the SSURGO model resulted in an overall better prediction for flow although both models did a comparable job in predicting storm flows. However, the STATSGO model predicted the base flows more accurately and had a slightly better performance during the validation period. Di Luzio et al. (2005) found that the effect of soil data input on SWAT model simulation of flow and sediment was limited compared to the effect due to DEM resolution and land use maps. Their study was based on a watershed in Mississippi dominated by silt loam soils and surface elevations ranging from 78 to 128 m above the mean sea level. It was concluded that further investigation is required to determine the role of GIS input data on different watersheds of varying sizes in different climatic and land resource regions.

Romanowicz et al. (2005) reported that the hydrologic response of the SWAT model to soil data input was significant in an agriculture dominated watershed situated in the central part of Belgium. Use of a detailed soil map improved the model performance considerably. Juracek and Wolock (2002) found that the differences in soil properties between a detailed and less detailed soil database will become less significant with increase in size of the study area. Chaplot (2005) conducted a study to compare the effect of soil map scale on water quality prediction by the SWAT model in a small watershed in central Iowa characterized by flat topography and poorly drained soils. Results showed that there was a significant difference in model prediction of water quality parameters due to soil map scale although the effect was less significant for runoff predictions. Detailed scale maps made better prediction of water quality parameters compared to a less detailed map.

Previous studies have reported that model performance improves with high resolution GIS data. However, only a few studies are available on the exclusive effect of soil input data on watershed-scale modeling of flow and water quality parameters. These studies have shown contrasting results in different physiographic regions. To our knowledge there has not been a study of this kind in the southern Piedmont region, which is characterized by steep slopes, highly erodible soils and intensive rainfall patterns. Several streams in the Piedmont region are listed for sediment TMDL development and it is important to know the effect of soil input data on model results. A large difference in model results would imply that modelers may use the soil database that supports their interests while developing TMDLs.

In Chapter II, we reported on the sediment yield estimates and rapid geomorphic assessment of the North Fork Broad River (NFBR) located in the Piedmont region of Georgia. The sediment yield estimates for this watershed were found to be high when compared to the median value for the Piedmont region. Geomorphic assessment of stream channels indicated that majority of the stream reaches were unstable. Sediment fingerprinting showed that almost 60% of the stream sediment originated from eroding stream banks.

The objectives of this study were:

1. To test the influence of spatial resolution of soil data in modeling flow and sediment in a southern Piedmont watershed
2. To calibrate the SWAT model for flow and sediment in a watershed dominated by channel erosion.

MATERIALS AND METHODS

Study site

The study area was the NFBR watershed. The watershed drains an area of about 182 km^2 (Figure 4.1). The land use of the study area is predominantly forested (deciduous, evergreen and mixed), occupying about 72% of the watershed, followed by pasture (15%) and row crops (7%). The elevation of the watershed ranges from 200 m near the watershed outlet to about 500 m near the headwaters. Madison and Pacolet (Fine, kaolinitic, thermic Typic Kanhapludults) soil series cover approximately 98% of the watershed. The soils are mostly well drained and moderately permeable. The average annual rainfall of the region is about 1400 mm.

Under the TMDL program originating from Section 303(d) of the 1972 Clean Water Act, EPA requires states to list waters that are not meeting the standards for specific designated uses (National Research Council, 2001). In 1998, the NFBR was included in the 303(d) list for impacted biota and habitat. Sediment was determined to be the pollutant of concern. The stream was placed on the list as part of a consent decree in a lawsuit filed against the United States Environmental Protection Agency (USEPA) and the Georgia Environmental Protection Division (Sierra Club *v*. Hankinson, 2003). The listing was based on an assessment of land-use in the watershed that concluded there was a high probability for impacted biota and habitat, although no sampling of the stream was conducted. Therefore, the USEPA developed a TMDL for sediment for the NFBR which was a calculation of the maximum amount of sediment that could be transported from the watershed outlet (where the river crosses highway 59) without affecting the designated uses of the waterbody. The TMDL report (USEPA, 2000) recommended that additional data be collected to better define the sediment loading from non-point sources. A stakeholder group involving farmers, county agents, nonprofit groups and other local and state agencies was formed to identify sediment sources and implement a watershed restoration plan. In 2004 after conducting a macro invertebrate survey, the USEPA removed the NFBR watershed from the 303(d) list and reported that "habitat concerns are present but not to an extent impacting the biota". However, no measurements of sediment concentrations and discharge were made so the annual sediment load in the NFBR remained unknown. At the same time, growing concerns on identifying the major sources of erosion, addressing stream bank erosion and erosion from construction sites and unpaved roads led to a Clean Water Act 319 grant in 2004, as part of which we initiated a monitoring and modeling approach.

The Soil and Water Assessment Tool (SWAT) model

SWAT is a physically based, semi distributed, 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, land use and management conditions (Neitsch et al., 2000). Major inputs for setting up the model include elevation, land use, and soil datasets. Each input GIS data layer provides various parameter values required by the model that can be modified to calibrate the model. The ArcSWAT data model is a geodatabase that stores SWAT geographic, numeric, and text input data and results in such a way that a single comprehensive geodatabase is used as a repository of a SWAT simulation (Olivera et al., 2006).

SWAT estimates surface runoff with the SCS curve number method. Erosion caused by rainfall and runoff is calculated with the Modified Universal Soil Loss Equation (MUSLE) as :

$$sed = 11.8 \cdot (Q_{surf} \cdot q_{peak} \cdot area_{hru})^{0.56} \cdot K \cdot C \cdot P \cdot LS \cdot CFRG \qquad (1)$$

where sed is the sediment yield on a given day (metric tons), Q_{surf} is the surface runoff volume (mm ha^{-1}), q_{peak} is the peak runoff rate (m^3 s^{-1}), $area_{hru}$ is the area of the

HRU (ha), K is the USLE soil erodibility factor (T h MJ^{-1} mm^{-1}), C is the USLE cover and management factor (dimensionless), P is the USLE support practice factor (dimensionless), LS is the USLE topographic factor (dimensionless) and $CFRG$ is the coarse fragment factor (dimensionless).

The peak runoff rate is calculated with the modified rational formula:

$$q_{peak} = \frac{\alpha_{tc} \cdot Q_{surf} \cdot Area}{3.6 \cdot t_{conc}} \qquad (2)$$

where q_{peak} is the peak runoff rate (m^3 s^{-1}), α_{tc} is the fraction of daily rainfall that occurs during the time of concentration (dimensionless), Q_{surf} is the surface runoff (mm), $Area$ is the subbasin area (km^2), t_{conc} is the time of concentration for the sub-basin (h) and 3.6 is a unit conversion factor.

In the SWAT model the transport capacity of a channel segment is estimated as a function of the peak channel velocity:

$$T_{ch} = a \cdot v^b \qquad (3)$$

where T_{ch} (T m^{-3}) is the transport capacity of a channel segment, a and b are user defined coefficients (SP_CON and SP_EXP in SWAT), and v (m s^{-1}) is the peak channel velocity. The peak velocity in a reach segment at each time step is calculated as:

$$v = \frac{\alpha}{n} \cdot R_{ch}^{2/3} \cdot S_{ch}^{1/2} \qquad (4)$$

where α is the peak rate adjustment factor for sediment routing in the main channel (PRF in SWAT), n is Manning's roughness coefficient, R_{ch} is the hydraulic radius (m) and S_{ch} is the channel slope (mm^{-1}). Occurrence of channel degradation or aggradation will depend on the transport capacity of the channel segment. Higher transport capacities can cause channel degradation (erosion). SWAT does not distinguish between bank and bed channel deposition or erosion.

A digital elevation model (DEM) of 30-m spatial resolution developed by the U.S. Geological survey (USGS) was used for the study. A land use map from 1998 with 18 classes developed at the Natural Resources Spatial Analysis Laboratory (NARSAL), University of Georgia was used for land cover information. This land cover map was produced from Landsat TM imagery with a spatial resolution of 30-m and an overall state-wide accuracy of 85% (NARSAL, 1998). The minimum resolution for input GIS data to achieve less than ten percent model output error depended upon the output variable of interest. For flow, sediment, nitrate nitrogen and total P predictions, the minimum DEM data should range from 30 to 300 m, whereas, minimum land use and soils data resolution should range from 30 to 500 m (Cotter et al., 2003). The STATSGO dataset is the default soil database in the SWAT model and was used directly. To compare the effect of spatial resolution of soil data on hydrologic modeling, the more detailed SSURGO data was downloaded from the NRCS soil data mart at a scale of 1:12,000-scale (USDA, 2006 and USDA, 2007). However, the data had to be processed into a database file format that SWAT recognizes. This was done using the SSURGO SWAT 2.x extension for Arc-View developed by Peschel et al. (2003). The SSURGO dataset for the two counties (Franklin and Stephens) falling within the watershed was processed using the extension and the attribute table required by SWAT was created.

Automatic watershed delineation

The NFBR watershed was delineated from the DEM into sub-basins using the automatic delineation tool in the ArcSWAT interface. A default threshold of 382 ha was specified as the minimum size of the sub-basin delineated. A watershed outlet was manually added corresponding to the location of the gauging station where the river crosses highway 59 and where the sediment TMDL has been developed. A total of 25 sub-basins were delineated for the watershed based on topographic and stream network data and threshold specification.

Land use and soils definition

The land use map was input in grid format using the land use and soils definition option in the Arc SWAT interface. The SWAT land use classification table was created automatically by the interface based on the grid values. The land use/land cover code generated was manually edited and converted to the SWAT land cover/plant code. The SSURGO soils database was input in shape file format and converted to grid format by the interface.

Hydrologic Response Unit (HRU) distribution

For comparing the influence of spatial resolution of soil data on model output, uncalibrated STATSGO and SSURGO models were run using default model parameters with one HRU per sub-basin by choosing the "dominant HRU" option. Otherwise, the number of HRUs would be different in the STATSGO and SSURGO models and this could affect flow and especially sediment predictions (Fitzhugh and Mackay, 2000; Chen and Mackay, 2004). For the calibrated models, the sub watersheds were divided into one or more HRUs (using the "multiple HRU" option) based on a unique combination of land use and soils in order to incorporate the spatial variability in land use and soils and account for the differences in evapotranspiration, surface runoff, infiltration and other processes in the hydrologic cycle. A threshold value of 10% was applied for both soils and land use. Minor soil types were eliminated by applying the threshold and a reasonable number of HRUs were created. A total of 119 HRUs were created using the STATSGO database and 248 HRUs were created using the SSURGO database for the calibrated models.

Weather data input

All weather parameters except precipitation data were simulated by the model. Daily precipitation data obtained from the National Climate Data Center (NCDC) of the National Oceanographic and Atmospheric Administration (NOAA) and observed data from ISCO tipping bucket rain gauges were used. Rainfall is the driving force for any hydrologic simulation model. Therefore, in order to provide a better input and spatial representation of precipitation, data from two weather stations in the Cooperative Observer Program (COOP) network of the National Weather Service (NWS) were used. The weather stations were located near the upstream and down stream region of the watershed. Each sub-basin used data from the nearest weather station estimated based on the proximity of the station to the centroid of each sub-basin. Precipitation data were converted into a format that was compatible with ArcSWAT. Weather stations were manually added to the interface and linked to the precipitation data for the corresponding station.

Flood control dams

Many flood control dams were constructed in the Piedmont during the 1950s and 1960s. A total of 14 flood control dams present in this watershed were expected to have an impact on sediment transport. Therefore details about the dams were added in the SWAT sub-basin input file. A GIS layer of the USEPA National inventory of dams was downloaded from the Georgia GIS clearinghouse data library. From this layer, parameters related to dams in the watershed such as area, storage capacity, and fraction of the sub-basin draining into the dam was obtained.

Flow and sediment measurement

Storm water samples were collected using an ISCO 6712 automated water sampler (ISCO Inc, Lincoln, NE) installed at the outlet of the watershed. The sampler was programmed to collect multiple discrete samples during a storm event. A pressure transducer installed vertically in the stream through a PVC pipe recorded the date, time and stage every five minutes. Sample collection was triggered by a predetermined stage height that was manually programmed into the ISCO sampler. The average duration of a hydrograph was about a day. The sampler was programmed to collect a sample every hour once it was triggered so that the collected samples would represent the entire hydrograph. However, the predetermined stage height was changed depending on the flow conditions and time of year. Base flow grab samples were collected at biweekly intervals in addition to the storm flow samples. Rainfall was measured at the monitoring station with a tipping-bucket rain gauge connected to the ISCO sampler's controller that was programmed to record precipitation amounts every five minutes.

Representative suspended sediment samples were selected from each storm event based on the time of sampling. Care was taken to make sure the samples represented the entire hydrograph. FLOWLINK- Advanced Flow Data Management software was used for data analysis and sample selection (ISCO Inc, Lincoln, NE). Samples were analyzed for suspended sediment concentration (SSC) in mg L^{-1} and turbidity in Nephelometric Turbidity Units (NTU). SSC was determined using the evaporation method described by

Guy (1969) which involves filtering a 250-mL sub sample into a pre-weighed 45-µm filter. The filter was then kept in an oven at 110°C for 24 hours and reweighed. SSC was calculated by subtracting the mass of the clean pre-weighed filter from the mass of the filter containing the filtrate. Turbidity was measured in a separate sub sample using a HACH 2100P turbidimeter (HACH Company, Loveland, CO).

Manning's equation was used to calculate the flow velocity (m s^{-1}) based on stream stage from which actual discharge (m^3 s^{-1}) was calculated by multiplying by the cross sectional area (m^2) of the channel. A rating curve was developed so that stream stage could be converted directly to discharge. To construct the rating curve, channel dimensions were measured to determine the hydraulic radius of the stream. Stream velocity, hydraulic radius, slope, and an estimated roughness coefficient were used to estimate discharge for a given stage height. The stream channel at the outlet of the watershed was stable and did not change its dimension during the period of monitoring.

The instantaneous discharge and sediment concentration data were used for annual sediment load estimates. A rating curve was developed using the LOADEST program (Runkel et al, 2004). This was a quadratic equation relating normalized stream discharge with instantaneous sediment loads:

$$\ln(L) = a_0 + a_1 \ln Q + a_2 \ln Q^2 \qquad (5)$$

where:

L = suspended sediment load [kg d^{-1}]

Q = discharge normalized by dividing by the long-term average

a_0, a_1 and a_2 are regression coefficients

The model gave an R^2 of 0.85 for load prediction. The p-values for the regression coefficients were statistically significant ($p<0.001$). Sediment yields in metric tons per hectare per year (T ha^{-1} yr^{-1}) were calculated by dividing the average annual load by the watershed area.

SWAT model simulation

The model was simulated on a daily time step for the period from January 1, 2005 to December 31, 2007. For flow and sediment calibration, the model output was compared with the observed data for flow and sediment at the gauging station located at the watershed outlet. Too many parameters can make hydrologic model calibration a difficult task. Therefore a sensitivity analysis was performed to identify the flow and sediment parameters that had a significant influence on the model output and eliminate the unimportant ones. The SWAT model uses the LH-Oat method (van Griensven et al., 2006) for sensitivity analysis that combines Latin Hypercube sampling to cover the full range of all parameters and the one factor at a time sampling method to ensure that changes in model output correspond to the parameter changed. The method was successfully used for SWAT modeling of the Sandusky River basin in Ohio and the Upper Bosque River basin in central Texas (van Griensven et al., 2006).

The five most sensitive parameters affecting stream flow were used for auto-calibration of flow. The auto-calibration tool in SWAT uses the Shuffled Complex Evolution Uncertainty Analysis (SCE-UA), an optimization method in which an objective function is defined for each output parameter for which observations are available. This objective function is an indicator of the difference between the observed and the simulated values (Green and van Griensven, 2008). The procedure involves random sampling of feasible parameters to be optimized, decided by the given parameter range from an initial population. The initial population is portioned into several complexes that evolve independently using a simplex algorithm. The complexes are shuffled to form new complexes and share the gained information. The method has been successfully used in hydrologic and water quality modeling (Eckhardt and Arnold 2001; van Griensven et al., 2002).

Once flow was calibrated using auto calibration tools, sediment calibration was done manually by changing one sensitive parameter at a time until a reasonable model output was obtained. Manual calibration was done because automatic calibration did not produce reasonable model output for sediment.

The calibrated model performance for flow and sediment were evaluated using the Nash- Sutcliffe model efficiency coefficient (*E*) given as:

$$E = 1 - \frac{\sum (O - P)^2}{\sum (O - \overline{P})^2} \tag{6}$$

where, *O* is the observed value, *P* is the predicted value, and $\overline{P}$ is the average of observed values.

RESULTS

Effect of spatial resolution of soil data

The average annual water yield and sediment yield predicted by the STATSGO and SSURGO models were compared for each of the 25 sub-basins before calibration. Results showed that the flow prediction by the two models were comparable, although small differences were observed in sediment predictions. STATSGO predicted more sediment compared to SSURGO in several sub-basins (Fig 4.2). However, a paired t-test showed that the differences were not statistically significant for either flow or sediment (α=0.05).

There are two possible effects of higher soil data resolution. One is the direct effect of the soil data parameters (Table 4.1) and the other is the indirect effect on derived parameters such as slope, slope length, and condition II curve number (CN II) (Table 4.2). By using a single HRU per sub-basin, the influence of topographic factors (slope and slope length) on uncalibrated models was eliminated. However, CN II did influence the model output. Sensitivity analysis on daily flows showed that SOL_AWC and SOL_K were the most sensitive soil data parameters and CN II was the most sensitive derived parameter. Table 4.1 and 4.2 explains the possible reason for the lack of significant differences between the two model outputs of predicted flow. Paired t-tests were conducted on all the soil parameters used in the SWAT model calibration. Though statistically significant differences existed in bulk density (BD) and available water capacity (AWC) values in the two databases at the sub-basin level, the spatial variability was low as shown by the coefficient of variation being < 10% for all of the variables except saturated hydraulic conductivity (SOL_K), silt content, and clay content. The SOL_K values in the databases at the sub-basin level were not statistically significant. This probably resulted in similar flow prediction by the two databases. The average values for SOL_K for the entire basin were 75.6 mm h^{-1} and 78.2 mm h^{-1} for STATSGO and SSURGO respectively. Though USLE_K values were statistically significant, the silt content in the two databases were not statistically significant and the average values for STATSGO and SSURGO were similar (20 vs. 26%). Soils become less erodible with a decrease in the silt content, regardless of whether there is a corresponding increase in the sand or clay content (Wischmeier and Smith, 1978). Sediment in runoff is predominantly determined by the silt content which is an important factor in determining the USLE_K values used by the SWAT model. This explains the similarity in sediment prediction by the STATSGO and SSURGO models.

Most of the watershed was dominated by hydrologic group B soils characterized by sandy loam or loamy sand texture. Higher flow prediction by the SSURGO model in sub-basins 23 and 24 can be attributed to the hydrologic group C soils, which are characterized by low infiltration rates and higher runoff potential (Figure 4.2 and Table 4.2). In the STATSGO model, most of the watershed was dominated by hydrologic group B soils except for sub-basin 6 which was dominated by hydrologic group D soils. A higher CN II of 83 compared to 66 in SSURGO explains the relatively higher sediment yield in this sub-basin. However, the differences in CN II values between sub-basins were not statistically significant (Table 4.2). Similar results were observed by Di Luzio et al (2005) in an effort to compare the effect of various GIS data layers on stream flow and sediment prediction in Goodwin Creek watershed, Mississippi. They found that using a coarser soil map (STATSGO) had little impact on model predictions when compared to a detailed county soil survey map.

Model calibration for flow and sediment

Automatic calibration determined the best parameter values for flow (Table 4.3). The model was rerun with the best parameter values and compared with the observed values. The model was run for a period of five years (2003-2007) and the first two years being the “warm-up” period; actual calibration was based on data for the period 2005-2007 (Fig. 4.3). For daily flow the calibrated STATSGO model had an R^2 value of 0.50

and the calibrated SSURGO model had an R^2 value of 0.42. The Nash-Sutcliffe (NS) model efficiency coefficient for daily flow was 0.24 for both the models. Once satisfactory calibration was obtained for flow, model parameters related to sediment were changed manually. The model predicted daily sediment load was compared with the measured daily sediment load at the watershed outlet (Fig. 4.4). The observed yearly sediment yield was also used for comparing the model performance (Table 4.4). The parameter values were finalized once satisfactory results were obtained (Table 4.3). The daily sediment load for the final calibrated STATSGO model had an R^2 value of 0.37 and the SSURGO model had an R^2 value of 0.41. The NS model efficiency coefficient for sediment was 0.21 and 0.31 for STATSGO and SSURGO models respectively. Annual sediment yield predictions by SSURGO were higher than STATSGO (Fig. 4.5).

The calibrated model prediction for average water yield (mm) and sediment yield (T ha^{-1}) for the simulation period using STATSGO for each sub-basin is presented in Fig. 4.6 and 4.8. The corresponding model output with SSURGO is presented in Fig. 4.7 and 4.9. Results showed that both water yield and sediment yield increased with distance from the watershed outlet indicating the influence of topography on flow and sediment yields. Sub-basins farthest from the outlet had the highest slopes (Table 4.2). A similar trend was observed when SSURGO was used. Sub-basins 2, 12, and 14 had relatively higher sediment yield compared to the neighboring sub-basins (Fig. 4.8 and 4.9). Sub-basin 14 had 12 % area under agriculture which was the highest in this watershed followed by sub-basin 5 which had 11%. This is the likely the reason for a higher sediment yield in these sub-basins. The higher sediment yield in sub-basin 12 was probably due to the pastures in this sub-basin that occupied 18% of the area compared to the neighboring upper sub-basins with no pasture lands.

Sediment calibration and sediment source identification

Sediment calibration was done manually as auto calibration did not produce reasonable results. This was due to the sensitivity of the model to the peak rate adjustment factor for sediment routing in the main channel (PRF) which is not a parameter included in the SWAT auto calibration tool. This parameter which impacts channel degradation processes at the watershed scale had to be adjusted to its upper limit in the SSURGO model and above the limit in the STATSGO model (Table 4.3). Arabi et al. (2007) reported that one of the most sensitive parameters affecting sediment transport is PRF and it is typically determined through calibration. Higher values of PRF may be an indication of channel erosion occurring in the stream reaches.

Two other sensitive parameters affecting sediment transport at the watershed level, SP_CON and SP_EXP, had to be increased to higher values to increase channel sediment transport capacity. These parameters were increased to the upper limit in both models (Table 4.3). These two parameters are the linear and exponential coefficients in equation 3. The parameter values used for sediment calibration clearly indicate that stream channels are an important source of fine sediment loading in the NFBR watershed. Similar results were reported by Radcliffe and Rasmussen (2001) in an effort to calibrate the HSPF model for suspended sediment at the USGS gauging station site further downstream on the Broad River near Bell, Georgia. The annual sediment load prediction by the two models clearly indicated the model's inability to simulate higher sediment loads (Table 4.4 and Fig. 4.5).

The sediment yield from the individual sub-basins (upland sources) was compared with the sediment yield from the main outlet of the watershed to determine the relative source contribution. Stream channel erosion rates for individual main channel reaches and two major tributaries was determined by the taking the difference in sediment load going in and out of a reach. A negative value indicated a net deposition and a positive value indicated channel erosion.

Model output before calibration showed that there was no contribution of sediment from stream channels as the sediment going in and out of each reach was the same. In effect, the default model parameters predict stable stream channels. However, the calibrated models showed that there was significant erosion occurring in the stream channels. The model was able to simulate this for the main channel and the two major tributaries. The average annual channel erosion rate predicted by the two models is presented in Fig. 4.10 and Fig. 4.11. The average rate for all the stream reaches in this watershed during the simulation period were 151 T km^{-1} and 183 T km^{-1} with the STATSGO and the SSURGO models.

The difference in channel erosion prediction by the two models was attributed to the difference in flow prediction by the two models. A 1:1 line showed that the flow values from the SSURGO model was high during high flows and low during low flows compared to the STATSGO model (Fig. 4.12).

The relative proportion of sediment coming from channel erosion compared to the proportion coming from uplands predicted by the two models is presented in Table 4.5. These values corroborate with the results of our sediment fingerprinting study in the same watershed that showed that 60% of the sediment originated from stream banks and 10-

15% sediment from pastures that were found to be the only upland surface source (Chapter III). The study also showed that 23-30% of the sediment originated from upland sub-surface soil sources such as construction sites and unpaved roads. However, this was not simulated in our SWAT models. The STATSGO model predicted more upland erosion compared to SSURGO and this was attributed to the influence of HRUs. Sediment generation decreases with increase in the number of HRUs as a result of non-linear relationship between MUSLE runoff term and HRU area (Fitzhugh and Mackay, 2000; Chen and Mackay, 2004).

Effect of flood control dams

The uncalibrated models showed a 38% reduction in average annual sediment yield due to the effect of dams with the STATSGO model and 36% reduction with the SSURGO model. This was due to the fact that before calibration most of the simulated sediment emanated from uplands and dams were efficient in trapping the sediment. In contrast, the calibrated models predicted that most of the sediment originated from the downstream channels which were below the dams. For this reason the effect of dams was less pronounced in the calibrated STATSGO model, predicting a 10% reduction, and the calibrated SSURGO model, predicting a 3% reduction in the average annual sediment yields. The higher reduction with STATSGO was because of the relatively higher proportion of sediment predicted from uplands compared to SSURGO (Table 4.5).

CONCLUSIONS

Comparison of flow and sediment yield by the two models before calibration showed that the influence of spatial resolution of soil data was relatively insignificant in this Piedmont watershed. This was mostly due to the similarity in the key model parameter values related to flow and sediment in the two soil databases. Though statistically significant differences existed in soil properties such as bulk density and available water content, the differences were small even in the high resolution soil database. Differences in saturated hydraulic conductivity values between the two databases were not statistically significant. The differences in silt content values in the two databases were statistically insignificant. The silt fraction is a governing factor in determining the *USLE_K* value used by the SWAT model and these were not different in the two soil databases. Both models produced similar sediment load predictions. The two models when calibrated for flow and sediment performed with comparable model efficiency. Flow prediction efficiency was the same with both models but the SSURGO model had a slightly better prediction for sediment. The calibrated models indicated that stream channels contributed most of the suspended sediment in this watershed. These results are consistent with our sediment fingerprinting study of the NFBR which indicated bank erosion was the largest source of sediment.

It appears that in the Piedmont physiographic region, parameters related to topography and land use may have more influence on stream flow and sediment yield than the parameters related to soils. The use of a detailed soil data layer did not increase the model performance considerably. The results may however vary with the physiographic region and the water quality parameter modeled. More time, effort, and computational resources are required to set up and calibrate a model with more detailed soil data especially in a larger watershed. The effect of spatial resolution of soil data may be more pronounced in a smaller watershed where the effects of soil variability are not lumped. Hence high resolution soils data may be more appropriate for smaller watershed in formulating and simulating land use management strategies at local scales.

Acknowledgements

The authors would like to thank the U. S. Environmental Protection Agency (USEPA) for providing a Section 319 grant through Georgia Environmental Protection Division (GAEPD) for the monitoring program. The study was also supported by the USDA-CSREES grant # 2007-51130-03869 (A New Approach to Sediment TMDL Watersheds in the Southern Piedmont).

REFERENCES

Arabi, M., R. S. Govindaraju and M. M. Hantush, 2007. A probabilistic approach for analysis of uncertainty in the evaluation of watershed management practices. *Journal of Hydrology* 333:459-471.

Chaplot, V., 2005. Impact of DEM mesh size and soil map scale on SWAT runoff, sediment, and NO3–N loads predictions. *Journal of Hydrology* 312:207-222.

Chen, E. and D. S. Mackay, 2004. Effects of distribution-based parameter aggregation on a spatially distributed agricultural nonpoint source pollution model. *Journal of Hydrology* 295:211-224.

Cotter, A. S., I. Chaubey, T. A. Costello, T. S. Soerens and M. A. Nelson, 2003. Water quality model output uncertainty as affected by spatial resolution of input data. Journal of the American Water Resources Association 39:977-986.

Di Luzio, M., J. G. Arnold and R. Srinivasan, 2005. Effect of GIS data quality on small watershed stream flow and sediment simulations. Hydrological Processes 19: 629-650.

Eckhardt, K., and J.G. Arnold, 2001 Automatic calibration of a distributed catchment model, Journal of Hydrology 251, 103-109.

FitzHugh, T. W. and D. S. Mackay, 2000. Impacts of input parameter spatial aggregation on an agricultural nonpoint source pollution model. *Journal of Hydrology* 236:35-53.

Geza, M., and J. E. McCray, 2008. Effects of soil data resolution on SWAT model stream flow and water quality predictions. Journal of Environmental Management 88:393-406.

Gowda, P.H., and Mulla, D.J., 2005. Scale effects of STATSGO vs. SSURGO soil databases on water quality predictions.Watershed management to meet water quality standards and emerging total maximum daily load (TMDL). In: Proceedings of the Third Conference 5–9 March 2005, Atlanta, Georgia,USA.

Green, C. H. and A. van Griensven, 2008. Autocalibration in hydrologic modeling: Using SWAT2005 in small-scale watersheds. Environmental Modelling and Software 23:422-434.

Guy, H., 1969. Laboratory theory and methods for sediment analysis. *Water-Resource Investigations United States Geological Survey*

Juracek, K. E., and D. M. Wolock, 2002. Spatial and statistical differences between 1: 250, 000- and 1: 24, 000-scale digital soil databases. Journal of Soil and Water Conservation 57:89-94.

NARSAL. 1998. 1998 land cover map of Georgia. Institute of Ecology, Univ. of GA, Athens, GA.

National Research Council, 2001. Assessing the TMDL approach to water quality management. Committee to Assess the Scientific Basis of the Total Maximum Daily Load Approach to Water Pollution Reduction. National Research Council, Washington, DC.

Neitsch, S.L., J. Arnold, J.R. Kiniry, and J.R Williams. 2000. Soil and Water Assessment Tool User's Manual Version 2000, Blackland Research Center, Texas Agricultural Experiment Station, Temple, TX.

Olivera, F., M. Valenzuela, R. Srinivasan, J. Choi, H. Cho, S. Koka and A. Agrawal, 2006. ArcGIS-SWAT: A Geodata Model and GIS Interface for SWAT 1. Journal of the American Water Resources Association 42:295-309.

Peschel, J.M., P.K. Haan, and R.E. Lacey, 2003. A SSURGO Pre-Processing Extension for the ArcView Soil and Water Assessment Tool. ASAE Paper No. 032123, St. Joseph, Michigan.

Peschel, J. M., P. K. Haan and R. E. Lacey, 2006. Influences of soil dataset resolution on hydrologic modeling. *Journal of the American Water Resources Association* 42:1371-1389.

Radcliffe, D. E., and T. C., Rasmussen 2001. Preliminary Sediment Analysis for the Broad River. Proceedings of the 2001 Georgia Water Resources Conference, held March 26-27, 2001, at the University of Georgia. Kathryn J. Hatcher, Editor, Institute of Ecology, the University of Georgia, Athens, Georgia.

Romanowicz, A. A., M. Vanclooster, M. Rounsevell and I. La Junesse, 2005. Sensitivity of the SWAT model to the soil and land use data parametrisation: a case study in the Thyle catchment, Belgium. Ecological Modelling 187:27-39.

Runkel, R. L., C. G. Crawford and T. A. Cohn, 2004. Load estimator (LOADEST): A FORTRAN program for estimating constituent loads in streams and rivers. US Geological Survey Techniques and Methods Book 4:69.

Sierra Club v. Hankinson. 2003. United States Court of Appeals for the 11th Circuit. Available online at http://altlaw.org/v1/cases/1128530 (verified 29 October 2008).

USDA, 2006. Soil Survey Geographic (SSURGO) database for Elbert, Franklin, and Madison Counties, Georgia. USDA NRCS, Fort Worth, TX.

USDA, 2007. Soil Survey Geographic (SSURGO) database for Banks and Stephens, Counties Georgia. USDA NRCS, Fort Worth, TX.

USDA-NRCS, 2004, State soil geographic (SSURGO) data base [online]. Available at http://www.soils.usda.gov/survey/geography/ssurgo/description.html (verified 2 February 2009)

USDA-SCS, 1994, State soil geographic (STATSGO) data base [online]. Available at http://www.soils.usda.gov/survey/geography/statsgo/description.html (verified 2 February 2009.

USEPA, 2000 Total Maximum Daily Load Development for the North Fork Broad River Watershed, EPA region 4, Water Management Division [online] Available at http://www.gaepd.org/Files_PDF/techguide/wpb/TMDL/Savannah/EPA_NorthForkBroadRiver_Sediment_TMDL.pdf (verified 2 February 2009)

van Griensven, A., A. Francos and W. Bauwens, 2002. Sensitivity analysis and auto-calibration of an integral dynamic model for river water quality. Water Science and Technology Vol 45 No 9 pp 325–332

van Griensven, A., T. Meixner, S. Grunwald, T. Bishop, M. Diluzio and R. Srinivasan, 2006. A global sensitivity analysis tool for the parameters of multi-variable catchment models. Journal of Hydrology 324:10-23.

Wang, X., and A. M. Melesse, 2006. Effects of STATSGO and SSURGO as inputs on SWAT model's snowmelt simulation. Journal of the American Water Resources Association 42:1217-1236.

Wischmeier, W. H., and D. D. Smith, 1978. Predicting Rainfall Erosion Losses: A Guide to Conservation Planning, Dept. of Agriculture, Science and Education Administration: for sale by the Supt. of Docs., US Govt. Print. Off.

Table 4.1 Soil parameters used by SWAT for modeling flow and sediment directly affected by soil data resolution: BD- bulk density (g cm^{-3}), AWC- available water capacity (mm mm^{-1}), Sol_K- saturated hydraulic conductivity (mm h^{-1}), and USLE_K- USLE soil erodibility K-factor (T h MJ^{-1} mm^{-1}).

	BD**		AWC**		Sol_K		Sand (%)**		Silt (%)		Clay (%)**		USLE_K**	
Sub-basin	STA	SUR	STA	SUR	STA	SUR	STA	SUR	STA	SUR	STA	SUR	STA	SUR
1	1.25	1.41	0.11	0.14	73.00	32.40	66.68	55.50	19.32	14.50	14.00	30.00	0.20	0.28
2	1.25	1.50	0.11	0.14	73.00	100.80	66.68	67.30	19.32	20.20	14.00	12.50	0.20	0.24
3	1.25	1.63	0.11	0.13	73.00	100.80	66.68	67.30	19.32	20.20	14.00	12.50	0.20	0.24
4	1.25	1.55	0.11	0.11	73.00	100.80	66.68	67.80	19.32	23.70	14.00	8.50	0.20	0.10
5	1.25	1.41	0.11	0.14	73.00	32.40	66.68	55.50	19.32	14.50	14.00	30.00	0.20	0.28
6	1.48	1.63	0.14	0.13	110.00	100.80	67.85	67.30	19.65	20.20	12.50	12.50	0.24	0.24
7	1.25	1.47	0.11	0.14	73.00	32.40	66.68	55.10	19.32	17.40	14.00	27.50	0.20	0.28
8	1.25	1.63	0.11	0.13	73.00	100.80	66.68	67.30	19.32	20.20	14.00	12.50	0.20	0.24
9	1.25	1.47	0.11	0.14	73.00	32.40	66.68	55.10	19.32	17.40	14.00	27.50	0.20	0.28
10	1.25	1.47	0.11	0.12	73.00	32.40	66.68	55.10	19.32	17.40	14.00	27.50	0.20	0.24
11	1.25	1.41	0.11	0.14	73.00	32.40	66.68	55.50	19.32	14.50	14.00	30.00	0.20	0.28
12	1.25	1.47	0.11	0.14	73.00	32.40	66.68	55.10	19.32	17.40	14.00	27.50	0.20	0.28
13	1.25	1.50	0.11	0.16	73.00	32.40	66.68	34.70	19.32	37.80	14.00	27.50	0.20	0.28
14	1.25	1.47	0.11	0.14	73.00	32.40	66.68	55.10	19.32	17.40	14.00	27.50	0.20	0.28
15	1.25	1.47	0.11	0.14	73.00	32.40	66.68	55.10	19.32	17.40	14.00	27.50	0.20	0.28
16	1.25	1.47	0.11	0.13	73.00	100.80	66.68	67.90	19.32	19.60	14.00	12.50	0.20	0.28
17	1.25	1.52	0.11	0.14	73.00	32.40	66.68	55.50	19.32	14.50	14.00	30.00	0.20	0.28
18	1.25	1.31	0.11	0.10	73.00	100.80	66.68	66.80	19.32	19.20	14.00	14.00	0.20	0.20
19	1.25	1.52	0.11	0.14	73.00	32.40	66.68	55.50	19.32	14.50	14.00	30.00	0.20	0.28
20	1.25	1.47	0.11	0.13	73.00	100.80	66.68	67.90	19.32	19.60	14.00	12.50	0.20	0.28
21	1.25	1.47	0.11	0.14	73.00	32.40	66.68	55.10	19.32	17.40	14.00	27.50	0.20	0.28
22	1.25	1.47	0.11	0.14	73.00	32.40	66.68	55.10	19.32	17.40	14.00	27.50	0.20	0.28
23	1.55	1.36	0.11	0.10	87.00	331.20	67.85	65.40	19.65	19.60	12.50	15.00	0.24	0.24
24	1.55	1.36	0.11	0.10	87.00	331.20	67.85	65.40	19.65	19.60	12.50	15.00	0.24	0.24
25	1.25	1.47	0.11	0.14	73.00	32.40	66.68	55.10	19.32	17.40	14.00	27.50	0.20	0.28
Average	1.28	1.48	0.11	0.13	75.60	78.19	66.82	59.14	19.36	18.76	13.82	22.10	0.20	0.26
CV (%)	7.21	5.35	5.40	11.36	10.77	105.58	0.58	13.11	0.57	24.47	3.60	35.87	6.48	15.47

** Sub-basin averages statistically significant at α=0.01

Table 4.2 Derived parameters used by SWAT for modeling flow and sediment indirectly affected by soil data resolution.

Sub-basin	SLOPE (%) **		SLOPE LENGTH (m) *		CN II $^{+}$		CN II		# of HRUs	
	STA	SUR	STA	SUR	STA	SUR	STA	SUR	STA	SUR
1	10	11	44	42	66	66	61	61	8	12
2	24	23	14	13	66	66	63	63	6	3
3	6	10	63	54	66	66	61	60	9	14
4	26	32	10	10	66	66	63	63	3	5
5	8	10	56	50	66	66	62	63	5	15
6	13	14	33	30	83	66	75	62	6	6
7	16	15	21	23	66	66	61	61	3	10
8	12	11	37	49	66	66	63	65	3	6
9	14	14	28	30	66	66	60	60	4	11
10	7	10	61	47	55	55	59	59	4	12
11	7	12	61	38	66	66	61	61	2	3
12	7	11	61	43	66	66	60	62	4	11
13	7	8	61	54	66	66	61	61	4	7
14	7	8	61	52	66	66	63	63	5	18
15	7	8	61	57	66	66	61	61	4	6
16	7	6	61	61	66	66	61	61	4	7
17	7	9	61	50	66	66	64	66	2	7
18	7	8	61	53	66	66	61	61	4	13
19	7	7	61	66	66	66	61	63	4	11
20	7	8	61	57	66	66	62	62	3	13
21	7	6	61	73	66	66	63	65	6	11
22	7	9	61	54	66	66	62	62	4	10
23	7	5	61	80	66	77	61	66	8	14
24	4	6	89	78	66	77	64	69	6	10
25	6	5	72	74	66	66	62	64	8	13
Average	9	11	53	49	66	66	62	63		
CV (%)	60	55	35	37	6	6	5	4		

** Sub-basin averages statistically significant at α=0.01 * Sub-basin averages statistically significant at α=0.05 $^{+}$ with single HRU

Table 4.3 Parameter values used in the calibrated model.

Parameter	Description	Location	Model range	Value used	
				STATSGO	SSURGO
ALPHA_BF	Base flow recession factor, days	*.gw	0-1	1.0	1.0
CH_COV	Channel cover factor	*.rte	0-0.6	0.2	0.3
CH_EROD	Channel erodibility factor	*.rte	0-0.6	0.1	0.3
CH_K2	Hydraulic conductivity in the main channel, (mm h^{-1})	*.rte	0-150	10	10
CH_N2	Manning's "n" value for the main channel	*.rte	0.01-0.3	0.3	0.3
CNII	Change in SCS runoff curve number	*.mgt	+/- 25%	-25%	-25%
ESCO	Soil evaporation compensation factor	*.hru	0-1	0.80	0.80
PRF	Peak rate adjustment factor for sediment routing in main channel	*.bsn	0-2	2.5	2.0
SP_CON	Linear coefficient for sediment routing in the channel	*.bsn	0.01-0.0001	.01	.01
SP_EXP	Exponential coefficient for sediment routing in the channel	*.bsn	1.0-2.0	2.0	2.0
USLE_K	Soil erodibility factor	*.sol	0.01-0.65	0.35	0.35

Table 4.4 Observed and model predicted sediment yields (T ha^{-1} yr^{-1}).

Year	Observed	Model predicted STATSGO	Model predicted SSURGO
2005	2.3	0.53	0.58
2006	0.26	0.22	0.25
2007	0.37	0.37	0.42

Table 4.5 Predicted relative source contribution (T ha^{-1} yr^{-1}).

Source	STATSGO	SSURGO
Bank	0.30 (80%)	0.37 (88%)
Upland	0.076 (20%)	0.049 (12%)

*Values in parenthesis indicate the percent source contribution of total load.

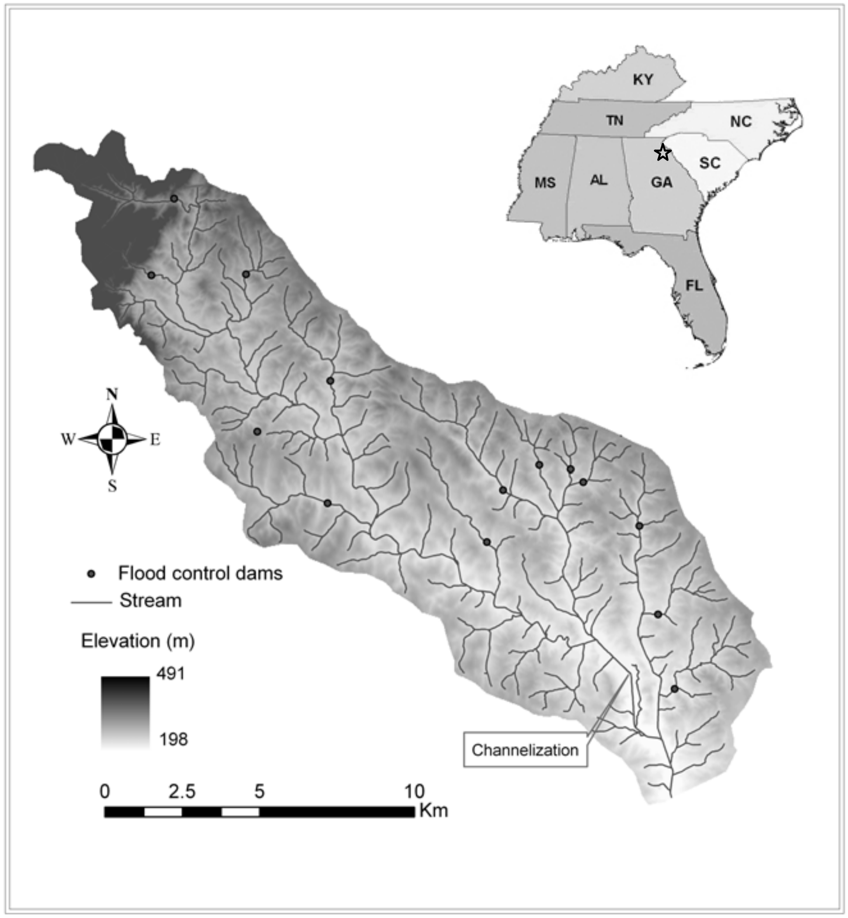

Figure 4.1 Location of the study site.

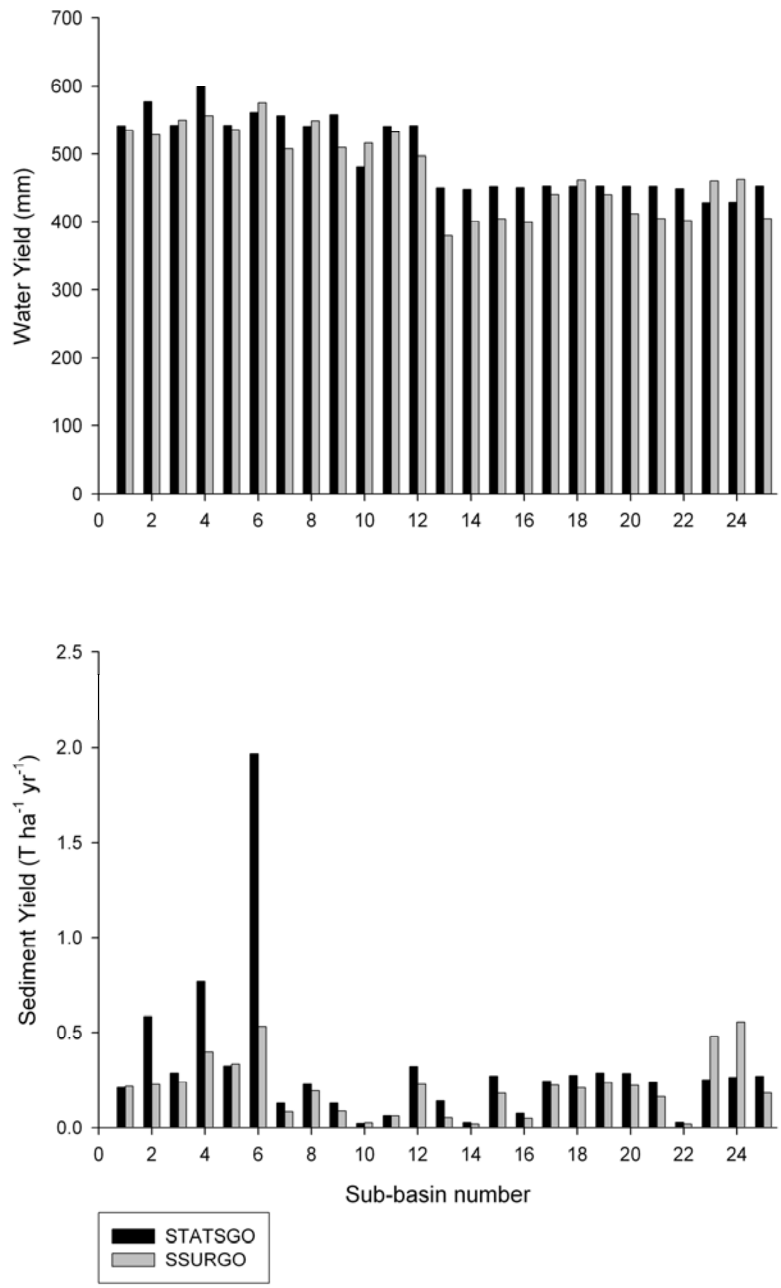

Figure 4.2 Water and sediment yield prediction by uncalibrated STATSGO and SSURGO models.

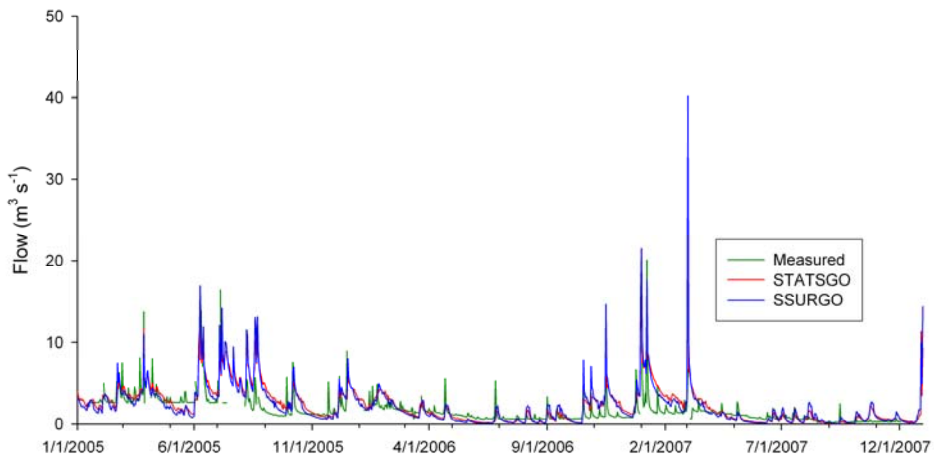

Figure 4.3 Measured and simulated daily flow (calibrated).

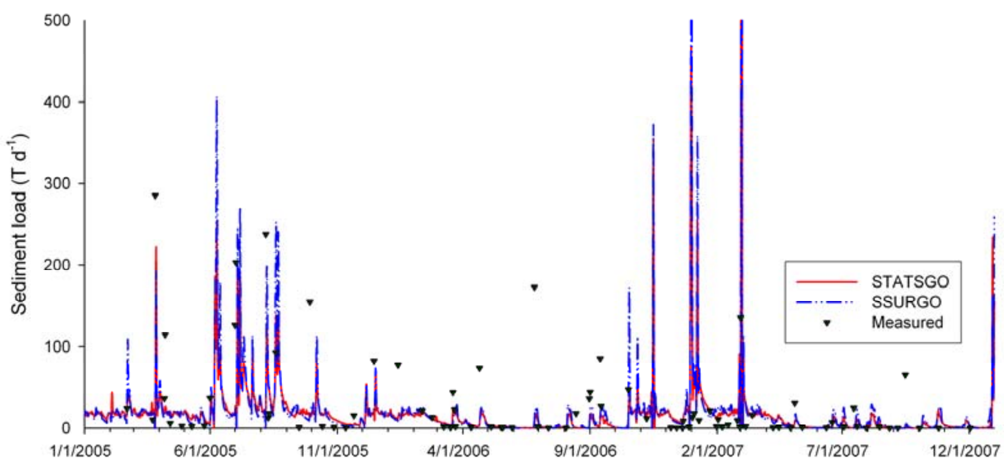

Figure 4.4 Measured and simulated daily sediment loads (calibrated).

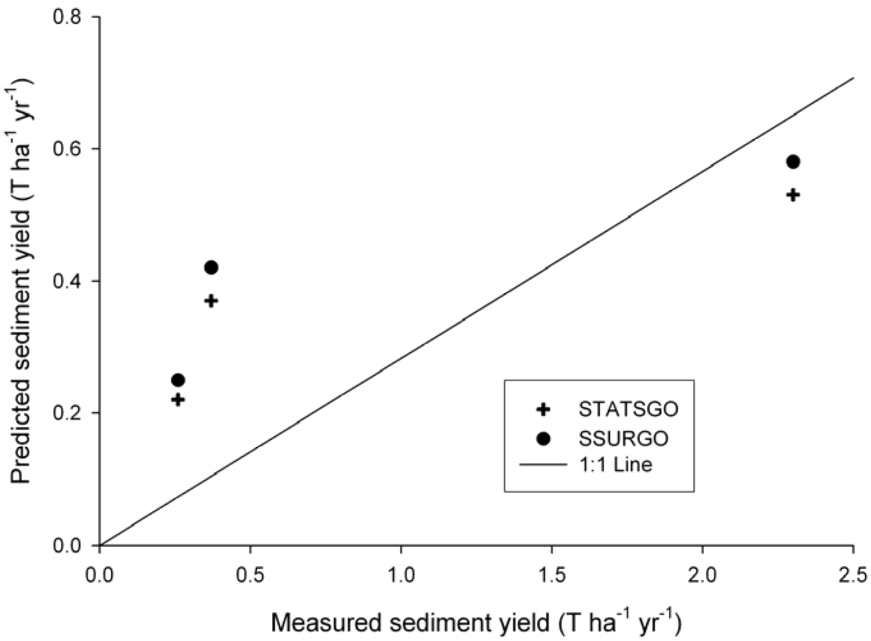

Figure 4.5 Measured and predicted annual sediment yields (T ha^{-1} yr^{-1})

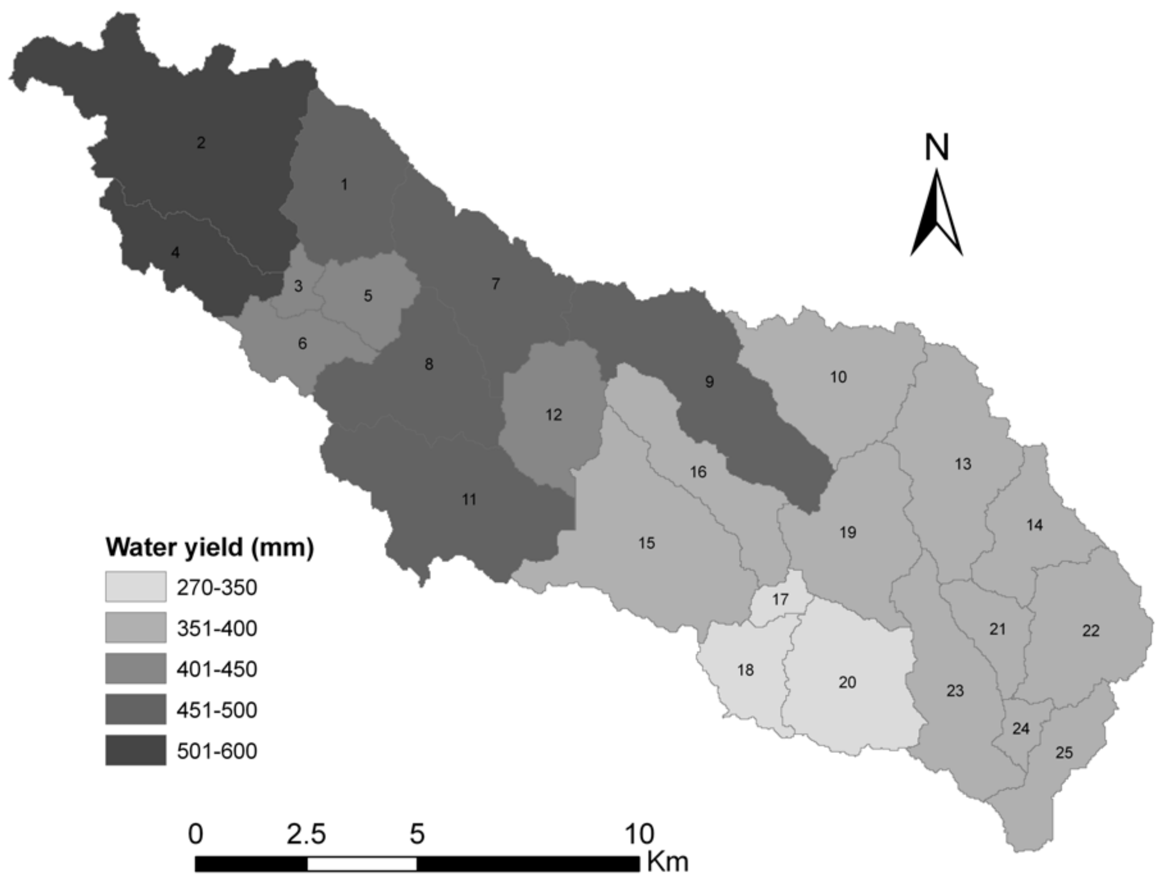

Figure 4.6 Calibrated STATSGO model prediction of average annual water yield

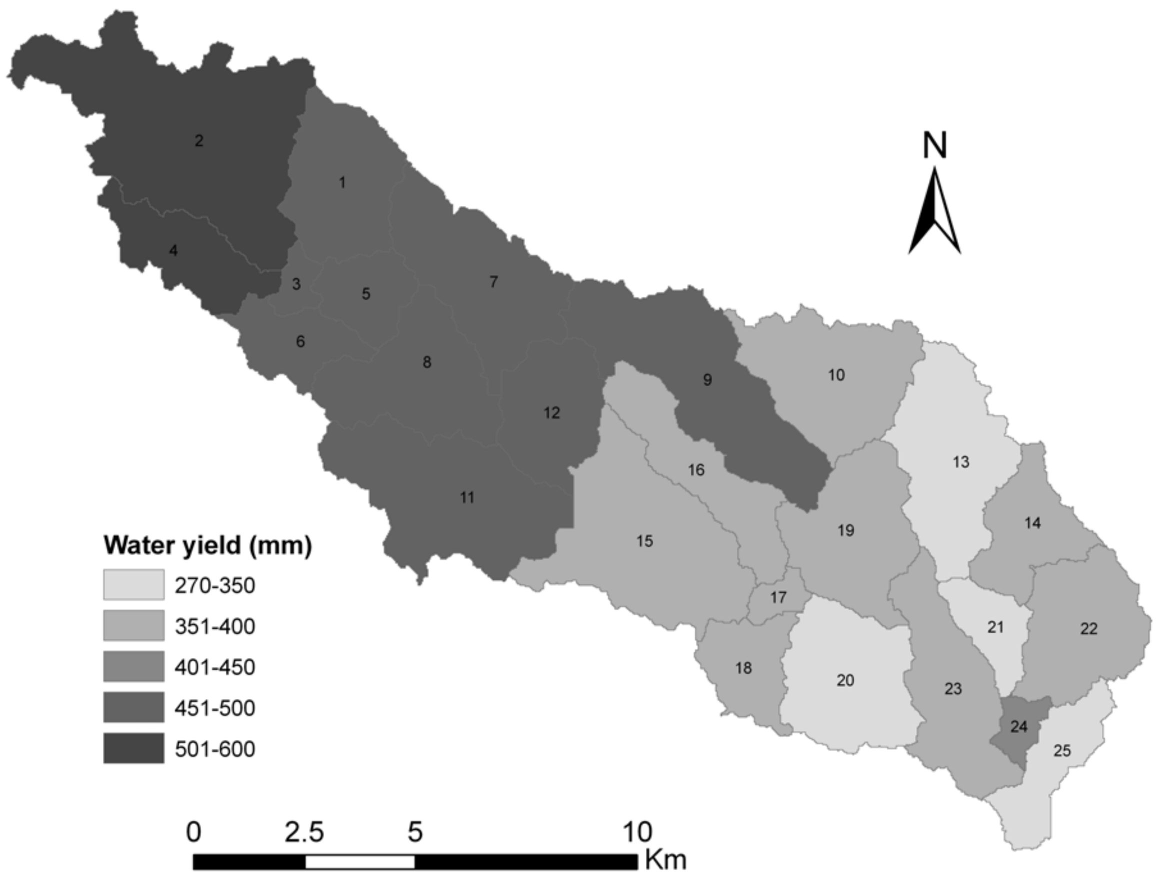

Figure 4.7 Calibrated SSURGO model prediction of average annual water yield

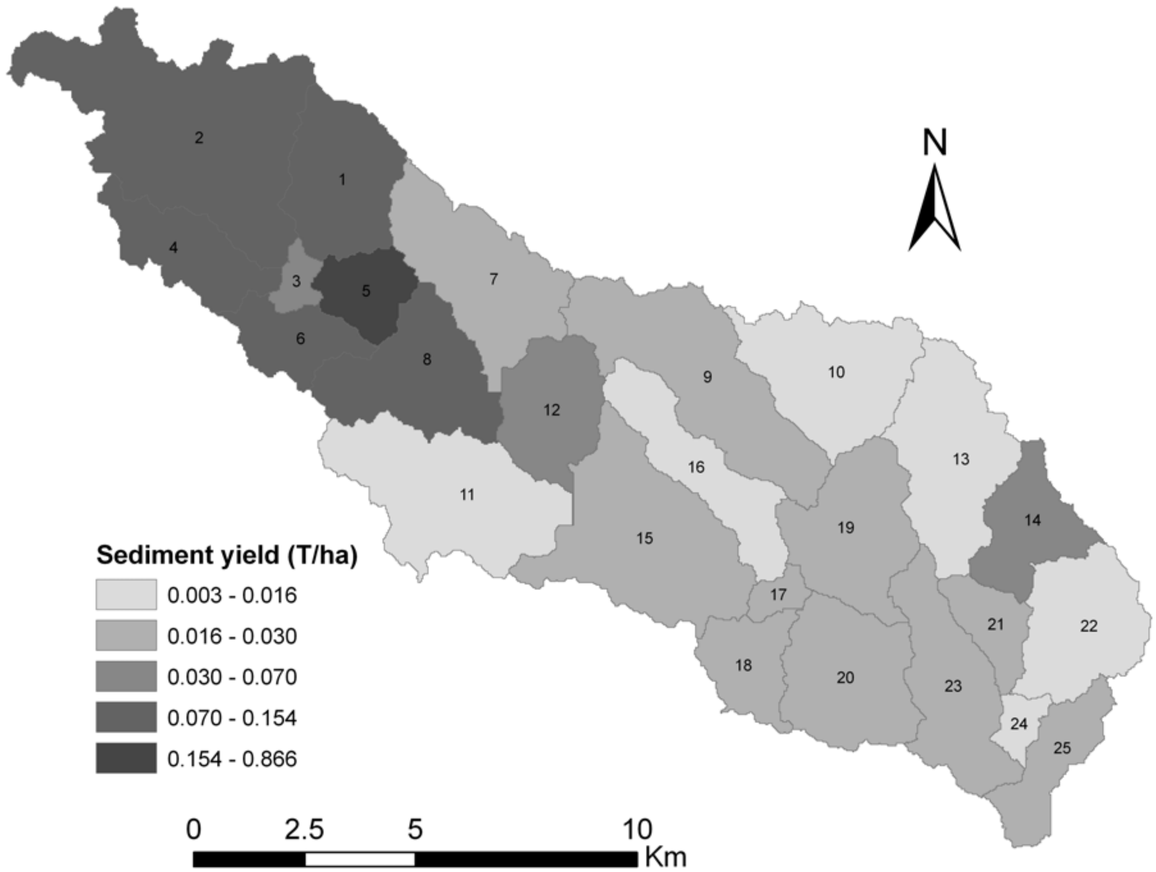

Figure 4.8 Calibrated STATSGO model prediction of average annual sediment yield

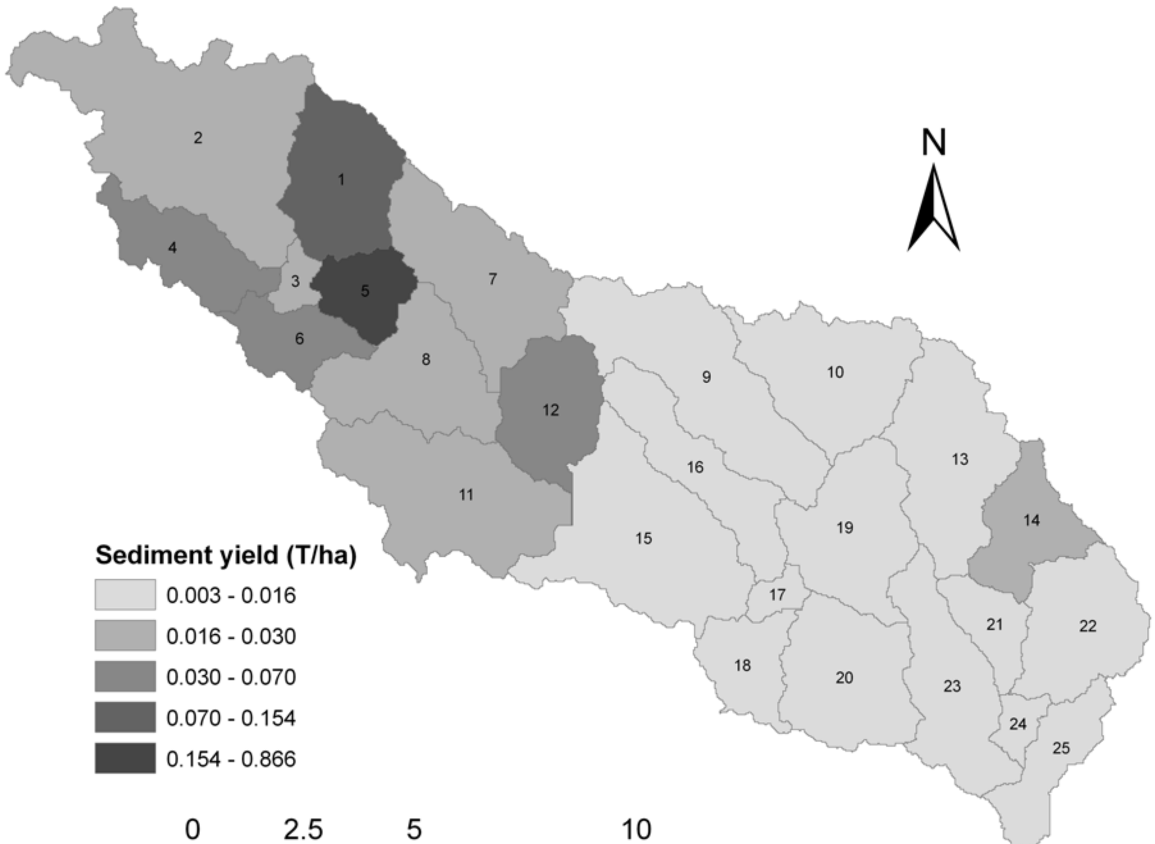

Figure 4.9 Calibrated SSURGO model prediction of average annual sediment yield

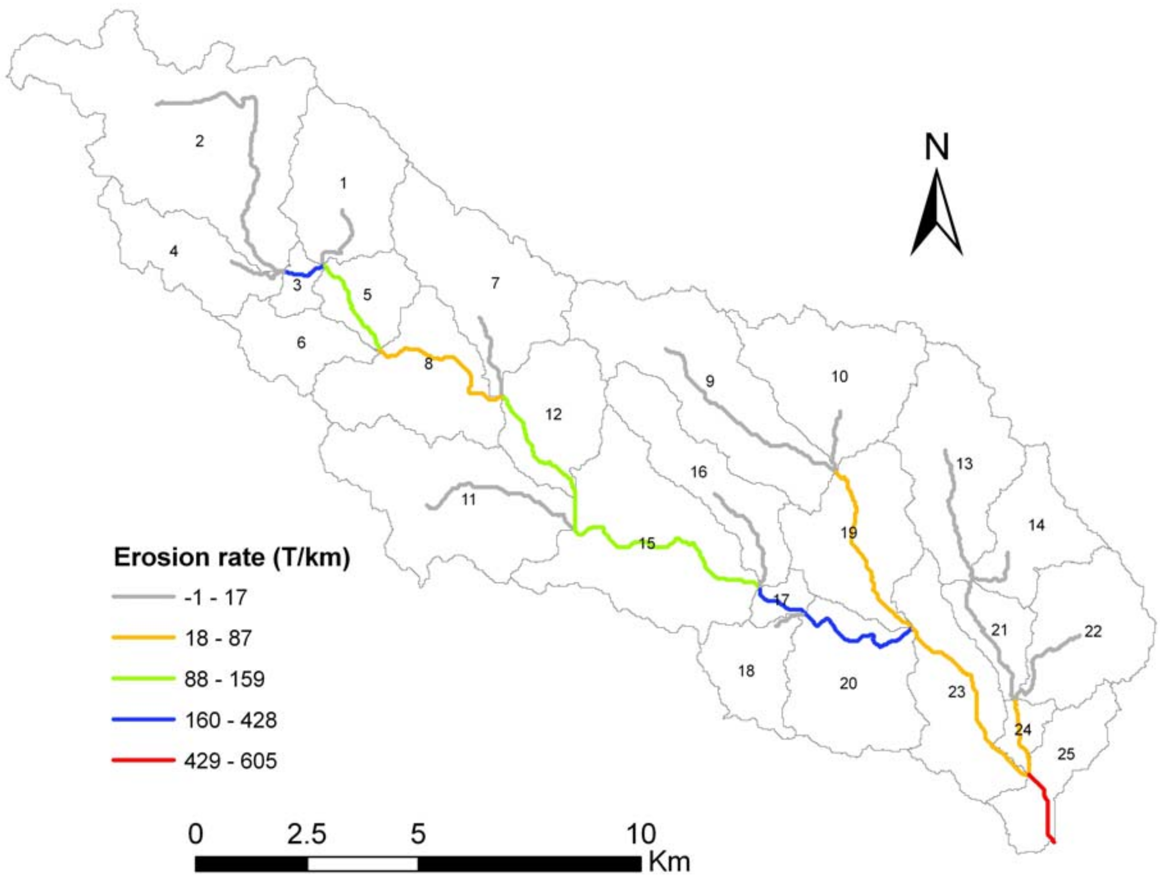

Figure 4.10 Calibrated STATSGO model prediction of stream channel erosion

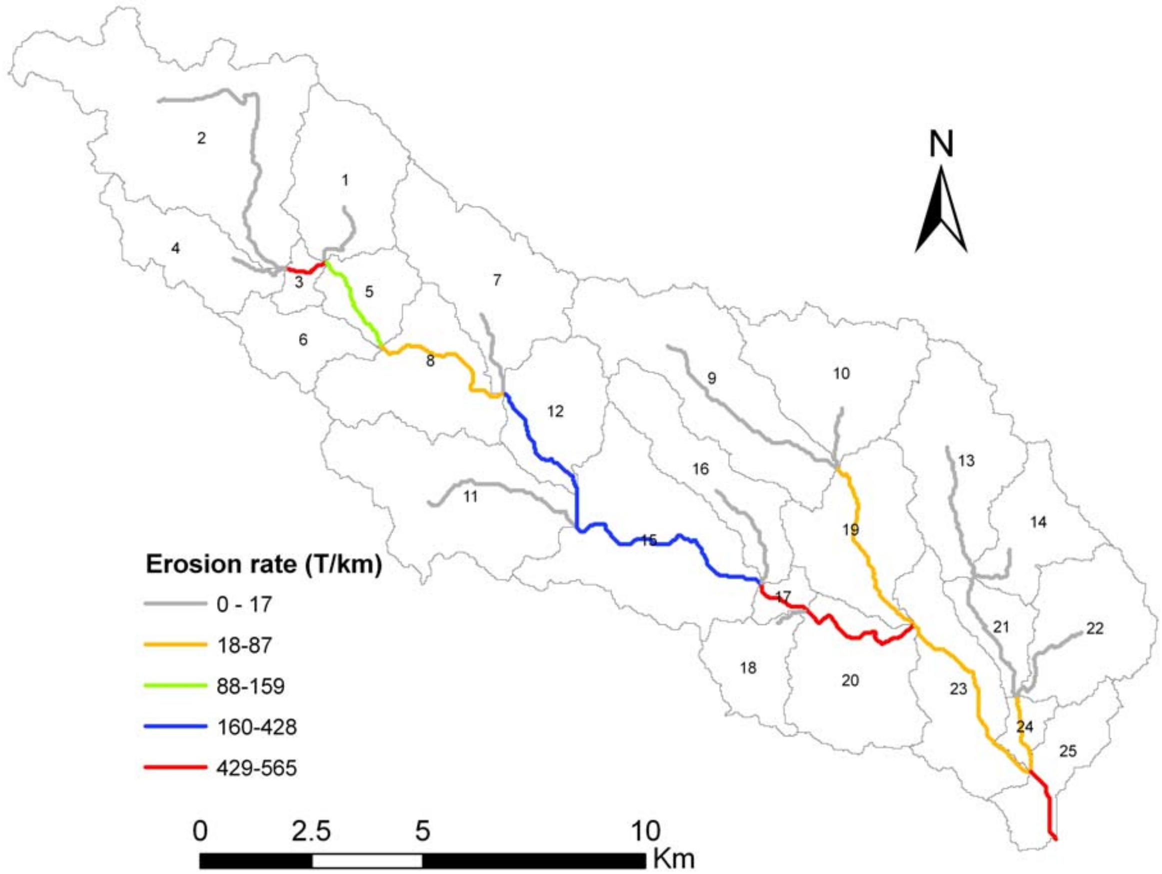

Figure 4.11 Calibrated SSURGO model prediction of stream channel erosion

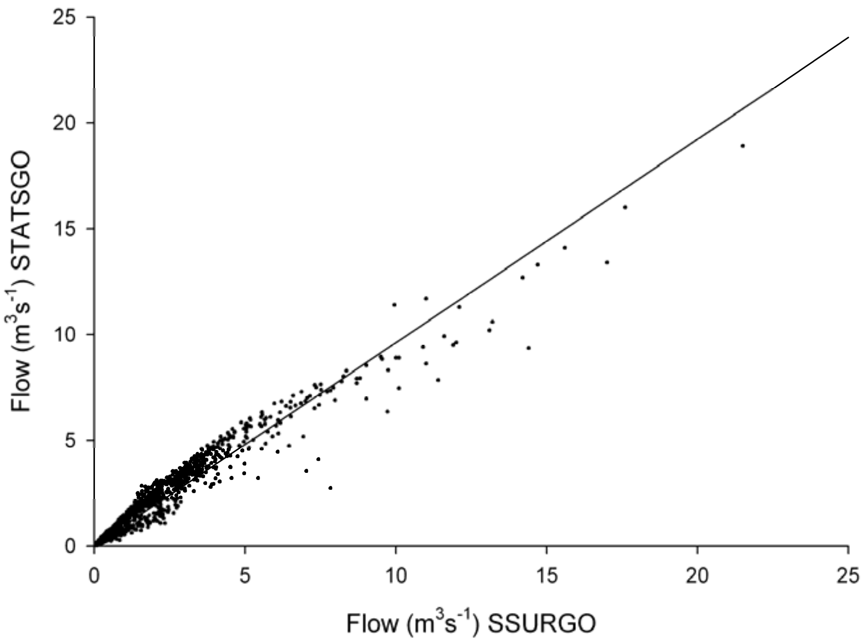

Figure 4.12 Comparison of flow prediction by calibrated models

CHAPTER V

OVERALL CONCLUSIONS

The primary objective of this research was to assess the water quality of a Piedmont stream with respect to suspended sediment from a TMDL perspective. A new approach to implementing sediment TMDL in watersheds in the Piedmont that have high sediment loads was developed that includes assessing the current condition of stream channels and the rate of sediment transported, applying principles from fluvial geomorphology and hydrology.

Stream channels in the NFBR watershed are relatively unstable as evidenced by the geomorphic assessment. Channel erosion processes such as mass wasting and fluvial erosion are dominant in the channels. The sediment yield estimates are comparable with streams in the Piedmont that are in an unstable stage of channel evolution. Excess sediment is being delivered into the stream channels compared to the streams ability to transport, as is evident from the large number of zones of accretions within the channel.

The vestiges of human activity in the past may still be having an effect on the Piedmont stream channels. Over a period of time one can expect a complete transition of the channel towards stable conditions where most of the suspended sediment will be found emanating from the tributary streams and field gullies. However, this process may take a long time. Export of *legacy* sediment from southern river basins has been estimated to occur at the rate of 5% every 100 years (Trimble, 1975). If this rate is constant then it will take approximately 2000 years for the stream channels in the

Piedmont to become stable and regain their transport ability. Information on the current sediment loads along with long term regional sediment loads for a physiographic region can help in determining target loads and developing sediment load reduction plans for a watershed.

The sediment fingerprinting study conducted as part of this research indicated that the majority of the suspended sediment in this Piedmont stream emanated from eroding stream banks (60% of the total). This consisted mostly of sediment eroded from uplands during the cotton era and deposited in the flood plains. Legacy sediment will be difficult to control as a source compared to other sources. However, the rapid geomorphic assessment of stream channels used in this research can identify hotspots within a channel and help prioritize stream bank restoration efforts at a watershed scale. The second most dominant source of suspended sediment in this stream was upland subsurface sources such as construction sites, unpaved roads, field gullies, and ditches that accounted about 23-30% of the total suspended load. This is an important source to control and can produce significant load reduction within a short period of time through best management practice (BMPs). The third important source of suspended sediment identified is pastures that occupy about 15% of the land use area in the southern Piedmont. Pastures contributed only about 10-15% of the total sediment in this watershed. This may be another important source that needs more control efforts not only from a sediment management perspective but also from a P management perspective.

This study was able to overcome two major limitations in sediment fingerprinting studies. One key area that required refinement was developing improved methods for suspended sediment sampling during storms so that sufficient mass of suspended sediment can be collected for all the different analyses. The use of a truck-mounted continuous flow centrifuge linked to a water pump ensured that sufficient mass of suspended sediment could be collected at will. A larger sample size also ensured that the collected suspended sediment samples were a truly representative mixture from multiple sources. To the best of my knowledge the method of suspended sediment sampling adopted in this study has not been tried elsewhere. Another area that required attention was finding tracers that could identify multiple sediment sources. The use of ^{137}Cs is limited when there are more than one sub-surface source such as banks, construction sites, and unpaved roads due to the fact that ^{137}Cs is concentrated at the soil surface. This problem was overcome by using ^{15}N. The $\delta^{15}N$ values showed distinct signatures in all the potential sediment sources and was found to be a unique tracer to differentiate bank soil from upland sub-surface soils. Moreover, $\delta^{15}N$ values are not affected by particle size distribution of the sources and the sediment and hence do not require particle size correction which is another key issue in sediment fingerprinting studies. This is also important because the end member mixing analysis used in this study is insensitive to using tracer ratios for determining relative sediment source contributions when multiple tracers are used. More research is required in identifying unique tracers especially to fingerprint suspended sediment in urbanizing watersheds. A preliminary result on the use of background levels of rare earth elements seems promising. Also results clearly indicate that total C may be used as a viable and cost effective alternative to the more expensive ^{137}Cs for sediment fingerprinting.

The third objective of this research was to calibrate the SWAT model for stream flow and suspended sediment transport in the NFBR watershed and to test the effect of spatial resolution of soil data using two models: one based on STATSGO soil data and the other based on SSURGO soil data. Comparison of flow and sediment yield by the two models before calibration showed that the influence of soil data resolution was relatively insignificant in this Piedmont watershed. This was mostly due to the similarity in the key model parameter values related to flow and sediment in the two databases. The two models when calibrated for flow and sediment performed with comparable model efficiency. Both models had similar flow predictions and the SSURGO model had a slightly better sediment predictions. The calibrated model parameter values for sediment clearly indicated that stream channels are an important source of fine sediment loading in the NFBR watershed. The relative source proportion of sediment from upland erosion *vs*. bank erosion predicted by the model corroborated the results of the fingerprinting study.

It appears that in the Piedmont physiographic region, parameters related to topography and land use may have more influence on stream flow and sediment yield than the parameters related to soils. The use of a high resolution soil data layer did not improve the model performance considerably. The results may, however, vary with the physiographic region and the water quality parameter modeled. More time, effort and computational resources are required to set up and calibrate a model with more detailed soil data, especially in a larger watershed. The effect of soil data resolution may be more pronounced in a smaller watershed where the effects of soil variability are less likely to get lumped.

There is every reason to believe that the NFBR watershed is a typical watershed in the southern Piedmont and hence the results may be applicable to other watersheds in the southern Piedmont. Geomorphic analysis of fluvial systems can aid in stream bank restoration and sediment source assessment. Sediment source identification has practical implications on soil erosion control strategies because the methods used for surface erosion control are different from that of bank erosion control. The holistic approach adopted in this research can be easily adapted to other watersheds or regions of varying scales depending on the availability of resources.

REFERENCES

Trimble, S. W., 1975. Denudation studies: can we assume stream steady state? *Science* 188:1207-1208.

APPENDIX A

PHOTOGRAPHS OF CHANNEL EVOLUTION STAGES

Incised Stage III channel

Stage IV channel showing signs of undercutting and fluvial erosion

Aggrading Stage V channel

Relatively stable Stage VI channel

APPENDIX B

TRACER CONCENTRATIONS IN SOIL AND SEDIMENT SAMPLES

Sample ID	Source	^{137}Cs	$\delta^{15}N$	C (%)	N (%)	S (%)	Be (ppm)	Mg (ppm)	Al (ppm)	P (ppm)	K (ppm)
1	Bank	0.00	6.88	0.80	0.05	0.01	0.65	1817	15873	78	775
2	Bank	3.29	3.52	1.54	0.10	0.04	0.58	1882	16320	183	1737
3	Bank	0.00	1.29	0.53	0.05	0.02	0.29	1207	8123	90	1166
4	Bank	0.00	6.49	0.42	0.03	0.01	0.89	1197	33973	244	784
5	Bank	0.00	4.27	0.74	0.04	0.02	0.33	1207	7653	89	1166
6	Bank	5.50	6.91	0.84	0.06	0.02	0.45	1667	9923	117	1576
7	Bank	2.65	6.65	1.23	0.08	0.03	0.34	1317	8093	104	1336
8	Bank	1.24	5.12	0.84	0.06	0.02	0.38	1607	9793	138	1398
9	Bank	6.37	2.98	0.64	0.04	0.02	0.45	1837	11373	135	1741
10	Bank	0.00	5.42	1.17	0.07	0.02	0.49	1647	10473	131	1670
11	Bank	10.02	6.41	1.48	0.09	0.02	0.51	507	14473	172	213
12	Bank	1.24	7.59	0.64	0.03	0.01	0.29	1227	8243	114	1231
13	Bank	0.00	6.69	1.80	0.09	0.02	1.31	2497	42473	352	1427
14	Bank	0.00	5.77	1.32	0.08	0.02	0.59	2127	21373	194	2103
15	Bank	0.00	5.38	1.40	0.09	0.02	1.12	1847	40373	262	1094
16	Bank	3.14	9.16	1.15	0.08	0.03	0.51	1457	14973	156	1192
17	Bank	0.00	3.09	0.57	0.03	0.01	0.56	1377	14873	156	1121
18	Bank	0.00	2.21	0.11	0.01	0.01	0.48	1597	18273	140	1574
19	Bank	2.24	5.31	1.33	0.07	0.02	1.75	2527	34973	312	554
20	Bank	0.00	7.63	1.45	0.07	0.02	1.32	1107	35673	369	601
21	Bank	2.36	2.87	1.59	0.08	0.02	1.08	2607	35373	261	2309
22	Bank	0.00	4.27	0.16	0.01	0.01	0.58	1607	20373	126	1247
23	Bank	2.64	6.88	0.83	0.05	0.02	1.04	2257	33873	211	1490
24	Bank	2.29	5.64	0.83	0.05	0.01	0.88	1837	25073	169	1238
25	Bank	3.09	6.57	0.74	0.04	0.01	1.07	1917	45573	299	1121
26	Bank	0.00	4.85	0.96	0.05	0.01	1.09	2057	31673	274	1274
27	Bank	2.33	6.66	0.49	0.03	0.01	0.49	1457	17173	103	1416
28	Bank	4.52	5.56	1.18	0.08	0.02	0.85	1167	24873	214	806
29	Bank	2.41	5.61	0.24	0.01	0.00	0.34	1397	7343	91	1319
30	Bank	2.51	3.98	0.91	0.06	0.01	1.25	2217	42873	271	1238
31	Bank	2.52	2.04	0.53	0.03	0.01	0.65	1867	24973	151	1487

Sample ID	Source	^{137}Cs	δ15N	C (%)	N (%)	S (%)	Be (ppm)	Mg (ppm)	Al (ppm)	P (ppm)	K (ppm)
33	Bank	2.08	7.50	0.82	0.06	0.01	0.59	1615	30773	124	1537
34	Bank	1.98	5.01	0.55	0.03	0.01	0.72	2177	16673	170	1877
36	Bank	0.69	7.54	0.80	0.05	0.01	0.71	2027	20473	162	1895
37	Bank	8.25	5.79	0.86	0.05	0.02	0.81	2337	29073	151	2498
38	Bank	0.00	5.90	0.96	0.05	0.03	0.80	1447	25273	206	1067
39	Bank	9.96	6.87	1.02	0.07	0.02	0.72	1867	27473	195	1886
40	Bank	0.00	7.93	0.83	0.05	0.03	0.63	1637	27973	162	1805
41	Bank	0.00	4.05	1.49	0.08	0.03	0.97	1577	35073	281	905
42	Bank	3.60	6.24	0.85	0.05	0.02	0.72	2017	19873	152	1985
43	Bank	0.00	-	1.50	0.09	0.02	1.03	1827	25573	205	1202
67	Bank	0.00	5.20	0.67	0.05	0.02	0.50	1313	19138	85	1156
68	Bank	0.00	5.02	0.58	0.04	0.02	0.32	1414	19508	73	1240
72	Bank	0.00	4.37	0.54	0.05	0.00	0.40	797	15788	79	436
74	Bank	0.00	4.64	0.81	0.06	0.02	0.55	1007	18268	116	692
139	Bank	2.76	3.96	1.63	0.12	0.02	0.51	1731	15108	192	1884
154	Bank	1.00	-	0.55	0.03	0.01	0.41	831	12868	30	532
155	Bank	2.47	-	1.93	0.13	0.02	0.59	1747	22468	176	1816
156	Bank	3.03	-	0.99	0.07	0.01	0.37	839	12908	61	732
157	Bank	0.00	-	1.31	0.09	0.02	1.52	1231	30428	120	329
158	Bank	0.96	-	0.97	0.05	0.01	0.34	855	11988	53	672
159	Bank	1.35	-	1.15	0.08	0.02	0.23	419	24588	105	476
160	Bank	0.00	-	1.07	0.07	0.01	0.40	967	12948	83	708
161	Bank	1.95	-	2.01	0.14	0.02	0.47	635	35948	270	724
162	Bank	0.00	-	0.54	0.04	0.01	1.24	915	25708	71	318
163	Bank	0.00	-	0.82	0.04	0.02	0.42	1259	30428	92	1456
164	Bank	0.00	-	0.46	0.04	0.02	0.95	1079	22828	78	748
165	Bank	1.91	-	1.29	0.09	0.01	0.47	887	15508	109	752
57	Construction	5.65	4.98	1.58	0.12	0.03	0.41	400	20848	294	528
58	Construction	0.00	6.70	0.62	0.08	0.03	0.58	1646	23128	268	960
59	Construction	4.13	4.46	1.70	0.13	0.05	0.73	400	21988	262	478
89	Construction	2.10	BDL	0.11	0.01	0.02	1.02	1335	21228	388	1020

Sample ID	Source	^{137}Cs	δ15N	C (%)	N (%)	S (%)	Be (ppm)	Mg (ppm)	Al (ppm)	P (ppm)	K (ppm)
92	Construction	1.52	1.42	0.58	0.04	0.01	0.49	2563	37268	47	2384
97	Construction	4.74	-0.04	0.84	0.06	0.02	0.47	2675	31388	27	3244
99	Construction	0.11	0.17	0.34	0.02	0.04	0.09	137	39868	47	110
101	Construction	1.99	3.74	0.24	0.03	0.04	0.17	165	20388	26	171
110	Construction	7.29	0.31	0.48	0.03	0.02	0.38	1365	30908	39	844
121	Construction	3.58	-0.08	0.63	0.04	0.03	0.25	175	33588	80	153
129	Construction	0.00	3.01	0.38	0.02	0.03	0.29	1835	22668	43	2304
130	Construction	0.00	BDL	0.11	0.01	0.01	0.19	1207	14348	BDL	1804
137	Construction	4.08	-0.18	0.45	0.04	0.01	0.74	3439	35908	108	3684
141	Construction	0.23	-2.64	0.45	0.04	0.04	0.17	307	25788	40	428
145	Construction	1.16	0.31	0.42	0.03	0.03	0.24	1323	21508	23	1812
150	Construction	0.00	BDL	0.08	0.01	0.02	0.31	1015	15388	BDL	1592
152	Construction	0.00	0.10	0.28	0.02	0.01	0.28	2479	19428	26	3724
44	Forest	13.80	-1.69	4.12	0.28	0.03	0.16	319	8047	154	208
46	Forest	5.26	2.16	2.55	0.19	0.04	0.85	1496	15691	2080	1694
47	Forest	9.24	-0.59	3.22	0.21	0.03	0.58	400	11505	181	163
50	Forest	13.25	-0.29	7.75	0.39	0.08	1.65	335	18422	418	192
52	Forest	24.28	2.46	8.09	0.50	0.10	0.65	1288	28348	654	1432
56	Forest	18.08	5.72	5.29	0.42	0.09	1.17	1129	25598	510	740
61	Forest	14.70	0.40	4.50	0.31	0.03	1.20	529	11386	152	226
64	Forest	22.60	-0.39	11.70	0.58	0.12	0.09	438	9391	158	306
69	Forest	18.87	-0.12	3.61	0.23	0.04	0.45	876	25548	232	592
70	Forest	0.00	5.13	1.43	0.10	0.04	0.65	566	23836	212	301
75	Forest	19.02	0.08	3.96	0.22	0.03	0.43	477	10988	76	349
78	Forest	23.83	0.69	6.43	0.37	0.04	0.88	337	18848	213	259
82	Forest	18.43	-1.57	6.99	0.39	0.06	0.49	391	22843	279	266
87	Forest	33.62	-0.09	11.64	0.66	0.12	0.09	411	18748	338	472
96	Forest	11.99	-0.26	2.01	0.14	0.02	0.04	181	7308	34	207
105	Forest	14.35	-2.41	5.23	0.28	0.04	0.25	469	14308	126	568
112	Forest	7.37	-0.87	4.93	0.28	0.04	0.28	1873	21988	147	1232
113	Forest	11.00	-0.32	4.85	0.24	0.04	0.14	379	17308	116	432

Sample ID	Source	^{137}Cs	δ15N	C (%)	N (%)	S (%)	Be (ppm)	Mg (ppm)	Al (ppm)	P (ppm)	K (ppm)
118	Forest	3.08	-0.49	2.80	0.14	0.03	0.67	235	28748	324	241
119	Forest	8.40	-2.90	3.68	0.18	0.04	0.44	743	29388	240	568
126	Forest	20.75	-0.47	2.95	0.19	0.02	0.50	717	20228	106	620
132	Forest	12.58	-0.42	7.85	0.45	0.06	0.25	631	23308	326	552
135	Forest	25.80	0.20	5.57	0.30	0.04	0.57	2139	17748	151	2120
140	Forest	42.04	-1.93	7.56	0.39	0.06	0.11	186	6988	168	263
143	Forest	16.25	-1.25	10.63	0.68	0.12	0.44	377	23908	486	333
146	Forest	24.36	-0.64	5.68	0.32	0.05	0.09	327	9228	156	317
147	Forest	25.48	-1.26	7.31	0.31	0.06	0.06	169	11988	145	246
148	Forest	1.34	-2.66	7.19	0.24	0.05	0.19	1255	12868	111	1820
48	Pasture	9.34	9.41	10.62	0.66	0.16	0.29	510	7501	660	774
49	Pasture	4.83	9.06	4.42	0.41	0.09	0.25	576	4607	496	754
53	Pasture	7.76	10.20	8.27	0.79	0.20	0.84	2334	27708	2854	2400
54	Pasture	1.91	10.16	1.44	0.17	0.04	1.14	5375	21908	2094	6028
55	Pasture	5.45	10.76	6.17	0.63	0.13	0.48	1044	16528	1678	1392
60	Pasture	6.60	6.40	6.36	0.61	0.12	0.41	586	18093	950	882
62	Pasture	10.68	9.01	7.14	0.70	0.13	0.84	1405	29398	1938	1390
63	Pasture	5.84	6.52	3.92	0.33	0.04	0.21	493	11424	362	522
71	Pasture	4.19	8.91	4.40	0.40	0.04	0.20	885	15108	1162	388
73	Pasture	14.36	1.70	2.67	0.17	0.03	0.21	266	7708	198	412
76	Pasture	3.04	10.85	8.80	0.79	0.12	0.22	1889	18748	2176	1100
77	Pasture	3.43	10.70	2.95	0.28	0.01	0.36	908	18648	1146	932
83	Pasture	8.55	7.23	4.42	0.41	0.06	0.81	2389	24363	1076	1522
84	Pasture	4.45	8.87	4.65	0.42	0.08	0.95	1999	34808	1376	2096
86	Pasture	5.45	9.88	2.04	0.20	0.01	0.05	643	16308	370	371
93	Pasture	5.22	0.16	3.01	0.24	0.03	0.37	711	22468	313	744
95	Pasture	5.08	4.71	1.39	0.09	0.03	0.10	239	20068	248	374
98	Pasture	0.30	5.99	2.05	0.19	0.03	0.26	1183	17548	338	1916
100	Pasture	1.68	3.87	0.87	0.05	0.01	0.21	809	37548	100	416
106	Pasture	5.54	2.93	5.12	0.44	0.07	0.36	605	19028	610	584
109	Pasture	28.04	9.15	3.00	0.28	0.05	0.07	318	9948	586	311

Sample ID	Source	^{137}Cs	δ15N	C (%)	N (%)	S (%)	Be (ppm)	Mg (ppm)	Al (ppm)	P (ppm)	K (ppm)
120	Pasture	5.26	5.55	-	-	-	0.04	425	12428	1050	832
123	Pasture	3.79	5.39	1.50	0.14	0.03	0.14	413	63148	417	488
124	Pasture	3.88	8.36	5.68	0.49	0.06	0.26	1029	20348	2970	728
131	Pasture	5.97	3.55	2.45	0.20	0.03	0.10	579	4468	70	476
138	Pasture	-	7.65	2.87	0.25	-	-	-	-	-	-
144	Pasture	7.33	11.26	12.44	1.01	0.20	0.34	839	20948	3318	980
149	Pasture	4.97	9.27	5.76	0.52	0.07	0.14	919	10108	1190	384
153	Pasture	5.55	2.90	1.68	0.14	0.03	0.27	507	21148	218	552
51	Row crop	8.39	8.83	2.55	0.30	0.06	0.44	960	16428	1094	764
65	Row crop	9.86	5.48	3.49	0.29	0.05	0.91	2678	27308	486	2294
66	Row crop	8.37	6.36	2.07	0.19	0.04	0.54	1509	16174	322	1362
81	Row crop	7.63	4.25	1.68	0.15	0.02	0.25	256	19423	389	301
88	Row crop	3.32	8.02	1.41	0.14	0.02	0.03	171	9228	258	318
91	Row crop	3.02	6.57	1.39	0.11	0.02	BDL	89	6508	59	73
103	Row crop	4.05	1.94	1.73	0.15	0.02	0.05	360	8748	115	460
122	Row crop	4.80	1.77	2.09	0.16	0.03	0.03	160	16268	256	259
125	Row crop	5.63	6.29	1.96	0.15	0.03	0.17	225	18188	1026	340
127	Row crop	7.03	9.59	1.48	0.15	0.03	0.70	673	31468	858	856
133	Row crop	7.60	7.35	2.38	0.20	0.04	0.28	1443	17588	470	1880
134	Row crop	6.01	8.48	1.76	0.16	0.03	0.75	1971	17788	534	1980
136	Row crop	7.27	4.14	1.61	0.13	0.02	0.48	811	7708	85	512
151	Row crop	0.00	3.64	0.74	0.05	0.03	0.23	321	20268	102	388
190	Sediment	3.74	5.08	3.77	0.31	-	-	2200	30112	-	-
191	Sediment	4.00	5.44	4.08	0.34	0.08	1.26	2240	31120	728	1192
192	Sediment	2.27	5.55	4.16	0.35	0.07	1.48	2496	35120	772	1256
193	Sediment	2.96	4.02	3.21	0.22	0.08	1.53	3472	38160	480	1644
194	Sediment	3.97	4.00	3.13	0.21	0.08	1.68	3604	40000	484	1708
200	Sediment	2.44	5.57	3.19	0.24	-	-	2400	33007	-	-
201	Sediment	2.83	5.62	3.19	0.25	-	-	2412	32763	-	-
203	Sediment	1.99	4.90	2.38	0.15	-	-	2313	20900	-	-
204	Sediment	2.88	4.64	2.19	0.14	-	-	2470	22700	-	-

Sample ID	Source	^{137}Cs	δ15N	C (%)	N (%)	S (%)	Be (ppm)	Mg (ppm)	Al (ppm)	P (ppm)	K (ppm)
206	Sediment	2.62	4.51	2.11	0.13	-	-	2617	23000	-	-
207	Sediment	2.48	4.55	2.17	0.14	-	-	2391	20500	-	-
208	Sediment	2.66	5.44	2.78	0.20	-	-	1580	21250	-	-
209	Sediment	3.48	5.71	2.62	0.21	-	-	1712	22750	-	-
210	Sediment	4.21	3.83	2.49	0.16	-	-	2393	24350	-	-
211	Sediment	3.48	3.95	2.57	0.17	-	-	2454	27650	-	-
212	Sediment	3.80	4.42	2.52	0.17	-	-	2219	24750	-	-
213	Sediment	2.77	4.34	2.38	0.16	-	-	1109	13150	-	-
214	Sediment	3.55	3.87	2.46	0.16	-	-	2242	26850	-	-
215	Sediment	5.25	4.29	2.64	0.18	-	-	2333	29850	-	-
80	Unpaved road	0.00	-0.26	0.61	0.04	0.03	0.40	490	11248	220	458
85	Unpaved road	2.09	2.91	0.16	0.05	0.05	0.76	349	39868	37	277
94	Unpaved road	2.25	2.05	0.46	0.04	0.01	0.21	304	14908	7	203
104	Unpaved road	0.00	1.35	0.20	0.02	0.06	0.70	263	41548	210	259
107	Unpaved road	6.19	-0.05	2.75	0.14	0.07	0.35	581	66748	260	429
108	Unpaved road	3.41	0.92	1.05	0.06	0.04	0.68	1853	40348	78	1516
115	Unpaved road	1.76	-0.08	0.48	0.03	0.03	0.53	1983	35628	113	2884
116	Unpaved road	1.08	2.32	0.59	0.04	0.05	0.55	175	47548	286	205
117	Unpaved road	1.80	1.93	0.39	0.03	0.04	0.62	294	52348	300	322
128	Unpaved road	0.00	BDL	0.08	0.02	0.02	0.40	647	23028	28	832

Sample ID	Source	Ca (ppm)	Cr (ppm)	Mn (ppm)	Fe (ppm)	Co (ppm)	Ni (ppm)	Cu (ppm)	Zn (ppm)	Pb (ppm)	U (ppm)
1	Bank	206	13.20	271	10793	6.41	4.10	7.55	31.55	8.71	1.51
2	Bank	309	17.40	529	11917	6.06	4.30	9.66	33.03	11.45	1.40
3	Bank	395	5.92	273	5083	3.11	2.39	4.37	21.25	5.18	0.93
4	Bank	430	22.10	1035	19793	7.91	5.44	12.13	44.70	16.49	2.35
5	Bank	396	6.48	298	5555	3.21	2.41	4.59	23.23	5.61	1.15
6	Bank	525	10.20	322	6913	4.48	4.00	7.10	31.05	7.20	1.25
7	Bank	374	6.87	278	5683	3.25	2.75	4.81	23.85	6.21	0.97
8	Bank	435	8.09	349	6804	4.10	2.79	6.19	27.15	6.90	1.08
9	Bank	516	10.00	313	7530	4.34	4.12	6.62	33.25	8.35	1.27
10	Bank	494	10.20	325	7625	4.27	4.21	6.83	33.87	8.28	1.36
11	Bank	130	12.70	193	6618	5.96	4.00	8.16	22.95	12.61	1.44
12	Bank	462	6.62	253	5070	3.05	2.68	4.58	23.75	4.97	0.83
13	Bank	615	44.20	1054	27629	16.49	11.87	23.25	72.25	25.51	3.23
14	Bank	327	21.00	649	17045	7.33	5.06	14.35	49.25	15.86	1.87
15	Bank	238	46.20	1083	28021	14.89	14.47	24.35	63.45	21.81	3.21
16	Bank	228	20.30	528	12537	6.22	5.14	10.35	38.45	10.81	1.85
17	Bank	221	20.40	534	12233	5.75	4.91	10.21	37.67	10.50	1.96
18	Bank	162	19.20	611	15673	6.81	4.20	10.05	31.25	12.91	1.74
19	Bank	348	29.20	1719	18353	12.77	11.47	17.95	73.45	18.71	3.25
20	Bank	303	35.90	731	26003	12.50	9.47	21.35	49.05	29.21	4.16
21	Bank	495	32.60	1449	26723	11.21	9.87	19.55	57.05	21.21	4.02
22	Bank	224	17.10	805	12773	6.71	5.35	9.67	29.55	10.86	1.65
23	Bank	555	26.00	540	20153	9.74	7.25	15.25	50.75	20.86	3.28
24	Bank	177	20.20	722	14483	7.82	5.52	12.55	38.55	16.51	2.59
25	Bank	385	35.70	694	31853	11.12	10.37	20.65	62.85	25.51	3.44
26	Bank	424	30.50	1287	24743	11.21	6.79	17.25	49.15	23.91	3.19
27	Bank	119	11.20	377	14893	5.89	0.82	9.70	27.75	12.01	1.48
28	Bank	223	20.20	363	21493	8.92	2.32	13.65	34.25	17.16	2.39
29	Bank	318	6.28	254	5555	3.14	2.38	4.84	22.28	6.32	0.99
30	Bank	784	35.00	1422	29063	13.97	11.27	19.81	67.21	23.47	3.62
31	Bank	230	17.30	461	16283	6.73	5.83	12.13	36.91	15.35	2.21

Sample ID	Source	Ca (ppm)	Cr (ppm)	Mn (ppm)	Fe (ppm)	Co (ppm)	Ni (ppm)	Cu (ppm)	Zn (ppm)	Pb (ppm)	U (ppm)
33	Bank	393	22.80	300	11873	5.76	10.97	8.54	34.91	10.17	1.99
34	Bank	439	14.90	529	13223	6.49	4.74	9.93	39.28	14.40	2.16
36	Bank	217	14.60	468	15563	6.09	3.82	9.25	35.96	15.97	2.40
37	Bank	164	15.60	1035	22763	8.91	4.85	11.17	47.64	18.29	2.89
38	Bank	338	23.10	1044	21773	11.39	5.88	12.61	35.48	15.25	2.47
39	Bank	253	16.50	558	17363	6.78	5.79	12.33	42.04	17.63	2.60
40	Bank	162	15.10	547	18173	6.32	4.55	11.65	37.29	15.16	2.30
41	Bank	283	24.50	1098	26723	9.65	5.93	17.32	42.70	22.38	3.01
42	Bank	154	14.70	523	15023	6.42	4.40	10.50	40.23	16.30	2.43
43	Bank	481	21.20	443	19703	8.34	4.95	13.19	41.09	20.57	3.10
67	Bank	164	29.70	664	16425	8.97	7.35	11.18	32.44	8.45	0.83
68	Bank	129	31.20	422	17325	7.07	7.15	12.00	46.44	9.49	0.88
72	Bank	154	19.64	253	11245	5.47	5.55	8.20	33.24	7.95	0.84
74	Bank	175	41.20	382	20245	8.63	8.87	16.48	50.84	11.39	1.65
139	Bank	337	4.84	458	13961	5.95	1.79	8.40	35.04	8.67	1.16
154	Bank	117	7.76	362	8361	3.61	1.59	4.04	18.52	5.75	0.78
155	Bank	390	25.36	458	15761	6.51	6.67	11.24	39.44	12.85	1.09
156	Bank	81	7.76	238	9121	3.63	1.56	5.48	23.52	7.49	0.81
157	Bank	406	27.96	142	8281	5.59	5.99	8.48	49.84	14.07	1.56
158	Bank	109	6.80	329	8841	3.75	1.46	4.96	24.88	6.13	0.74
159	Bank	105	7.92	255	12001	2.22	1.15	5.00	30.88	7.29	0.59
160	Bank	179	15.52	402	8841	4.55	2.96	5.64	20.48	8.29	0.71
161	Bank	395	11.40	610	15401	3.69	2.25	8.36	49.84	11.01	0.91
162	Bank	306	25.00	97	14881	4.63	5.03	8.64	39.44	12.61	1.36
163	Bank	68	13.36	434	16281	5.07	4.63	9.56	37.04	10.69	0.81
164	Bank	184	15.24	402	20801	10.95	2.90	7.28	33.84	12.23	1.93
165	Bank	286	14.12	426	12881	5.43	2.70	7.48	29.20	7.77	0.92
57	Construction	314	56.20	404	25125	5.11	4.99	19.88	27.04	11.85	1.36
58	Construction	135	45.10	193	29725	7.09	11.01	25.88	43.64	23.75	0.53
59	Construction	324	73.50	320	37125	8.95	8.73	24.08	37.84	9.47	1.64
89	Construction	32	45.60	240	38485	4.23	6.95	30.68	65.64	15.05	3.47

Sample ID	Source	Ca (ppm)	Cr (ppm)	Mn (ppm)	Fe (ppm)	Co (ppm)	Ni (ppm)	Cu (ppm)	Zn (ppm)	Pb (ppm)	U (ppm)
92	Construction	235	22.00	229	16205	4.59	24.79	7.64	63.64	10.27	0.99
97	Construction	102	10.92	248	23445	3.41	6.07	13.56	45.24	20.15	4.07
99	Construction	35	11.60	104	20845	1.24	2.51	8.12	28.68	5.15	0.67
101	Construction	105	59.40	184	17845	3.33	20.15	8.92	44.04	7.51	0.65
110	Construction	167	19.56	220	26285	8.67	7.75	24.88	39.64	7.81	0.64
121	Construction	83	66.20	316	30845	3.17	31.07	12.32	48.44	11.93	1.26
129	Construction	81	6.20	253	15441	3.66	2.22	6.56	24.20	8.59	1.29
130	Construction	25	1.02	223	7481	2.47	0.48	1.27	17.60	5.55	0.73
137	Construction	204	9.12	276	19201	6.87	4.07	10.72	47.44	11.67	2.11
141	Construction	75	12.96	47	19161	0.91	1.20	5.72	19.08	11.27	0.98
145	Construction	41	6.52	172	15681	2.04	0.74	2.47	21.00	9.93	0.97
150	Construction	47	2.79	111	13281	1.82	0.06	5.88	20.96	12.05	0.80
152	Construction	73	3.88	288	13761	3.65	1.41	2.57	33.84	11.01	0.97
44	Forest	678	6.84	277	7885	1.78	1.59	4.88	15.26	9.01	0.31
46	Forest	284	16.82	269	10125	3.73	5.35	10.26	32.46	8.77	1.03
47	Forest	374	11.94	603	17445	4.83	4.61	8.20	24.86	11.25	0.74
50	Forest	306	21.60	713	24945	5.51	7.99	18.80	21.86	14.32	1.88
52	Forest	486	4.44	394	14325	3.18	1.95	6.92	54.04	12.39	2.00
56	Forest	716	17.68	1916	18065	10.07	4.47	12.88	48.44	18.45	1.78
61	Forest	1740	54.10	1722	14825	25.73	12.23	10.36	53.44	12.49	0.71
64	Forest	378	2.32	177	5165	1.25	0.41	4.60	21.24	7.10	0.30
69	Forest	189	46.40	602	24925	9.31	7.47	20.40	48.04	18.85	1.03
70	Forest	80	40.00	622	27365	10.07	7.39	17.08	43.24	10.15	0.95
75	Forest	108	11.52	678	8165	5.47	2.55	6.56	29.94	17.53	1.32
78	Forest	499	15.18	2079	9210	8.51	5.08	7.33	27.04	17.15	0.69
82	Forest	444	49.98	1091	25620	14.96	7.87	25.71	60.24	20.20	0.77
87	Forest	472	20.24	383	12365	1.99	12.83	8.68	58.84	23.91	0.64
96	Forest	85	3.32	235	3449	1.79	2.99	1.80	12.48	8.59	0.52
105	Forest	218	7.44	304	6565	1.78	1.79	3.00	20.04	12.01	1.18
112	Forest	480	12.20	450	13165	5.79	3.19	9.08	54.04	11.51	0.49
113	Forest	227	4.32	105	7605	2.77	1.83	6.68	16.04	10.41	1.11

Sample ID	Source	Ca (ppm)	Cr (ppm)	Mn (ppm)	Fe (ppm)	Co (ppm)	Ni (ppm)	Cu (ppm)	Zn (ppm)	Pb (ppm)	U (ppm)
118	Forest	178	25.36	320	24845	4.15	3.04	10.40	14.16	6.65	0.72
119	Forest	344	19.64	305	28485	7.71	3.55	13.92	22.36	8.03	0.47
126	Forest	264	1.24	434	5645	2.15	1.11	3.80	32.96	12.47	1.66
132	Forest	1396	6.72	498	12121	2.13	3.55	7.52	29.40	13.77	1.09
135	Forest	436	2.69	355	9881	4.27	1.22	3.84	37.84	15.73	0.85
140	Forest	220	1.14	131	3229	0.62	0.61	2.22	14.28	12.09	0.39
142	Forest	287	7.00	1430	10841	6.19	3.23	7.80	28.72	17.29	1.18
143	Forest	559	32.84	562	24441	6.19	4.51	26.00	36.64	22.75	1.46
146	Forest	182	2.26	143	5721	1.05	1.16	2.83	15.44	12.67	0.42
147	Forest	109	2.91	49	7361	0.51	0.31	2.92	13.20	15.09	0.80
148	Forest	118	2.33	164	10841	1.97	0.57	2.96	22.36	11.59	0.55
48	Pasture	704	13.04	299	7605	3.57	3.15	11.70	36.26	7.85	0.45
49	Pasture	438	5.95	150	5005	4.51	3.70	6.96	23.86	2.67	0.52
53	Pasture	3054	25.60	966	22845	8.23	8.15	64.48	237.64	18.05	5.15
54	Pasture	2486	5.56	698	30525	10.03	1.53	15.52	108.04	6.51	1.70
55	Pasture	2356	10.56	522	12345	3.89	2.87	96.48	156.84	9.36	2.37
60	Pasture	1332	56.10	666	23925	5.53	5.11	53.28	75.84	12.56	1.44
62	Pasture	3436	65.90	766	42125	12.89	11.79	172.28	204.04	14.92	2.03
63	Pasture	642	12.16	516	7025	4.87	3.05	8.74	34.84	8.81	0.73
71	Pasture	3126	36.76	306	15565	3.88	3.51	30.92	88.84	9.07	1.12
73	Pasture	179	8.80	253	6445	3.80	2.19	6.08	24.86	11.81	1.19
76	Pasture	4436	12.18	365	11120	1.85	3.18	47.98	162.14	11.50	1.99
77	Pasture	1646	7.53	425	8080	4.00	3.42	25.38	64.24	8.41	1.23
83	Pasture	1616	23.98	468	14520	8.14	10.15	46.80	68.64	10.80	1.62
84	Pasture	1686	31.48	353	25120	7.62	14.35	53.54	81.84	17.65	1.58
86	Pasture	796	12.32	219	8805	3.65	3.83	15.20	35.24	9.53	0.67
93	Pasture	456	16.20	698	11845	5.79	5.35	10.28	36.04	11.37	0.84
95	Pasture	285	24.80	254	17605	3.99	6.11	10.28	23.00	9.49	0.75
98	Pasture	500	7.68	201	10365	1.70	20.31	10.52	79.64	12.91	1.57
100	Pasture	168	5.56	286	21005	5.47	2.19	5.24	32.64	4.38	0.49
106	Pasture	808	90.60	359	24685	5.39	26.23	28.16	87.64	12.71	0.84

Sample ID	Source	Ca (ppm)	Cr (ppm)	Mn (ppm)	Fe (ppm)	Co (ppm)	Ni (ppm)	Cu (ppm)	Zn (ppm)	Pb (ppm)	U (ppm)
111	Pasture	1856	38.00	610	12845	3.29	14.91	83.00	195.64	9.23	1.06
120	Pasture	1068	39.48	135	8685	1.41	14.87	23.20	170.44	9.67	0.83
123	Pasture	394	23.92	100	32405	1.98	4.63	9.96	87.64	9.39	1.01
124	Pasture	7000	8.92	950	12085	6.87	5.47	19.76	164.04	14.49	0.99
131	Pasture	179	2.93	125	2869	1.47	1.10	2.65	13.28	2.94	0.33
138	Pasture	-	-	-	-	-	-	-	-	-	-
144	Pasture	3239	46.00	502	28081	9.03	8.15	109.28	205.04	10.09	1.52
149	Pasture	2038	4.08	237	8721	0.89	1.47	17.84	99.84	7.57	1.30
45	Row crop	129	6.50	569	8225	3.27	2.20	5.60	15.66	4.89	0.44
51	Row crop	982	15.64	714	13925	4.91	4.11	27.72	75.24	11.65	1.90
65	Row crop	908	42.90	824	25925	12.35	9.91	23.08	76.64	15.81	1.59
66	Row crop	476	28.10	426	15145	6.69	5.67	14.64	46.04	8.90	0.91
81	Row crop	240	28.18	958	11220	7.56	5.25	11.74	29.94	31.90	0.96
88	Row crop	163	5.56	107	5565	2.05	1.63	5.88	24.48	8.04	0.45
91	Row crop	107	5.92	144	3669	1.63	0.95	3.76	7.92	4.27	0.31
103	Row crop	221	4.04	87	5685	0.77	0.95	4.76	17.68	6.06	0.80
122	Row crop	230	13.48	140	17125	2.03	1.39	8.08	25.28	7.12	0.70
125	Row crop	386	18.28	258	18805	2.63	3.23	16.48	152.84	11.83	1.09
127	Row crop	532	3.84	650	9485	3.45	2.71	33.52	72.44	18.29	2.50
133	Row crop	756	4.16	267	11721	3.03	1.33	6.20	48.64	13.01	1.17
134	Row crop	684	13.48	526	11641	7.11	5.91	14.96	58.64	10.63	1.55
136	Row crop	220	4.32	326	4401	2.77	1.96	4.48	22.96	5.33	0.71
151	Row crop	367	7.28	95	19601	1.45	1.02	9.60	35.04	15.73	1.07
190	Sediment	-	34.34	-	21206	-	-	-	-	34.04	2.49
191	Sediment	1885	32.20	5441	25718	15.60	11.32	118.40	105.40	21.56	2.16
192	Sediment	1989	35.50	6041	28118	17.32	12.04	63.20	107.40	21.48	2.39
193	Sediment	1653	36.90	1765	21960	13.52	13.68	57.60	113.80	24.58	3.04
194	Sediment	1681	39.00	1821	22760	14.28	14.60	58.00	121.80	24.70	3.12
200	Sediment	-	36.90	-	20226	-	-	-	-	35.95	2.78
201	Sediment	-	37.88	-	20716	-	-	-	-	22.31	2.85
203	Sediment	-	28.80	-	11880	-	-	-	-	16.63	1.89

Sample ID	Source	Ca (ppm)	Cr (ppm)	Mn (ppm)	Fe (ppm)	Co (ppm)	Ni (ppm)	Cu (ppm)	Zn (ppm)	Pb (ppm)	U (ppm)
205	Sediment	-	28.20	-	10187	-	-	-	-	14.23	1.71
206	Sediment	-	35.30	-	12167	-	-	-	-	16.28	2.04
207	Sediment	-	31.50	-	11078	-	-	-	-	14.88	1.93
208	Sediment	-	23.82	-	15500	-	-	-	-	21.56	2.25
209	Sediment	-	22.65	-	15200	-	-	-	-	21.60	2.06
210	Sediment	-	23.36	-	14900	-	-	-	-	19.78	2.43
211	Sediment	-	23.84	-	15700	-	-	-	-	19.22	2.46
212	Sediment	-	22.60	-	14300	-	-	-	-	18.24	2.21
213	Sediment	-	14.18	-	8180	-	-	-	-	10.11	1.18
214	Sediment	-	24.29	-	15500	-	-	-	-	21.45	2.25
80	Unpaved road	102	47.88	146	22120	8.19	11.65	17.82	19.04	6.09	1.08
85	Unpaved road	53	58.40	157	45525	5.75	11.31	20.00	46.04	18.81	2.24
94	Unpaved road	75	27.96	327	14565	7.99	6.51	9.56	16.52	7.49	0.79
104	Unpaved road	150	107.00	203	45925	4.39	27.87	25.32	53.24	15.77	6.35
107	Unpaved road	195	19.80	151	48325	2.76	4.87	17.44	50.04	17.27	1.65
108	Unpaved road	201	7.64	302	41925	18.59	6.67	54.96	56.84	14.43	1.03
115	Unpaved road	71	25.48	318	24205	6.87	7.43	25.00	43.64	10.85	0.71
116	Unpaved road	94	37.92	283	36365	2.56	4.31	21.08	24.00	8.55	0.92
117	Unpaved road	125	85.20	196	35245	2.72	22.23	15.56	51.64	8.19	0.97
128	Unpaved road	36	3.92	143	13321	0.99	0.95	3.05	22.04	13.23	3.11

APPENDIX C

PARTICLE SIZE DISTRIBUTION IN SOIL AND SEDIMENT SAMPLES

Sample ID	Source	Sand (%)	Silt (%)	Clay (%)
1	Bank	44.4	32.0	23.6
2	Bank	17.5	38.5	44.0
3	Bank	36.4	28.5	35.1
4	Bank	59.7	16.0	24.3
5	Bank	60.7	21.5	17.8
6	Bank	67.4	22.8	9.9
7	Bank	21.4	40.5	38.1
8	Bank	59.2	29.0	11.8
9	Bank	47.9	30.5	21.6
10	Bank	41.7	33.5	24.8
11	Bank	59.8	27.0	13.2
12	Bank	59.7	21.5	18.8
13	Bank	32.2	42.3	25.5
14	Bank	25.3	40.0	34.7
15	Bank	25.0	37.0	38.0
16	Bank	24.6	37.0	38.4
17	Bank	65.7	17.5	16.8
18	Bank	59.1	9.8	31.2
19	Bank	25.0	44.5	30.5
20	Bank	49.5	32.5	18.0
21	Bank	29.7	40.0	30.3
22	Bank	20.2	41.0	38.8
23	Bank	45.9	33.5	20.6
24	Bank	47.9	32.3	19.9
25	Bank	61.2	25.4	13.5
26	Bank	25.2	47.0	27.8
27	Bank	80.3	14.8	5.0
28	Bank	51.9	33.5	14.6
29	Bank	62.6	29.8	7.7
30	Bank	62.6	30.5	6.9
31	Bank	62.6	32.5	4.9
32	Bank	82.5	12.3	5.3
33	Bank	80.7	13.8	5.6
34	Bank	86.7	9.8	3.6
35	Bank	80.7	13.5	5.8
36	Bank	81.6	14.5	4.0
37	Bank	47.1	33.0	19.9
38	Bank	49.0	29.0	22.1
39	Bank	46.8	30.0	23.2
40	Bank	46.3	29.5	24.2
41	Bank	60.4	18.3	21.4
42	Bank	50.1	34.0	15.9
43	Bank	32.8	41.0	26.2
67	Bank	65.5	19.8	14.8
68	Bank	69.8	13.5	16.7
72	Bank	67.8	15.5	16.7
74	Bank	57.3	24.5	18.2
139	Bank	69.7	20.8	9.6

Sample ID	Source	Sand (%)	Silt (%)	Clay (%)
154	Bank	71.9	14.7	13.3
155	Bank	58.9	27.0	14.1
156	Bank	71.4	21.5	7.1
157	Bank	48.2	29.5	22.3
158	Bank	74.2	15.0	10.8
159	Bank	64.9	19.5	15.6
160	Bank	71.0	18.0	11.0
161	Bank	47.9	34.0	18.1
162	Bank	45.4	27.5	27.1
163	Bank	53.8	23.5	22.7
164	Bank	42.8	28.5	28.7
165	Bank	69.9	22.0	8.1
92	Construction	32.0	22.5	45.5
57	Construction	64.2	16.0	19.8
58	Construction	41.0	17.3	41.7
59	Construction	40.0	24.0	36.0
89	Construction	51.8	22.5	25.7
90	Construction	42.2	28.0	29.8
97	Construction	51.2	24.5	24.3
99	Construction	54.2	10.0	35.8
101	Construction	46.7	13.1	40.3
102	Construction	45.4	18.0	36.6
110	Construction	57.0	12.0	31.0
121	Construction	54.8	15.0	30.2
129	Construction	54.4	20.5	25.1
130	Construction	62.9	21.5	15.6
137	Construction	67.9	9.5	22.6
141	Construction	45.5	17.0	37.5
145	Construction	49.9	16.5	33.6
150	Construction	63.4	16.0	20.6
152	Construction	68.9	19.0	12.1
44	Forest	81.3	12.0	6.7
46	Forest	61.7	30.0	8.3
47	Forest	74.6	18.0	7.4
50	Forest	76.3	18.0	5.7
52	Forest	70.8	20.0	9.2
56	Forest	69.9	23.0	7.1
61	Forest	82.9	12.0	5.1
64	Forest	74.9	18.5	6.6
69	Forest	70.3	21.0	8.7
70	Forest	56.1	22.0	21.9
75	Forest	75.1	16.8	8.2
78	Forest	77.4	11.0	11.6
82	Forest	75.4	16.0	8.6
87	Forest	71.5	19.5	9.0
96	Forest	73.1	19.4	7.5
105	Forest	71.8	19.0	9.2

Sample ID	Source	Sand (%)	Silt (%)	Clay (%)
112	Forest	77.9	15.0	7.1
113	Forest	72.1	16.0	11.9
114	Forest	79.7	14.5	5.8
118	Forest	67.8	14.5	17.7
119	Forest	65.8	15.0	19.2
126	Forest	78.4	15.0	6.6
132	Forest	70.0	20.0	10.0
135	Forest	72.8	19.5	7.7
140	Forest	70.6	23.0	6.4
142	Forest	63.7	23.2	13.1
143	Forest	67.9	25.5	6.6
146	Forest	73.8	18.0	8.2
147	Forest	69.4	17.0	13.6
148	Forest	67.4	20.0	12.6
48	Pasture	86.6	9.0	4.4
49	Pasture	67.3	24.5	8.2
53	Pasture	61.8	29.0	9.2
54	Pasture	66.8	20.0	13.2
55	Pasture	74.8	18.5	6.7
60	Pasture	72.1	21.2	6.6
62	Pasture	61.6	24.0	14.4
63	Pasture	74.4	17.3	8.4
71	Pasture	69.3	20.0	10.7
73	Pasture	71.3	19.0	9.7
76	Pasture	73.8	17.2	9.0
77	Pasture	68.8	21.0	10.2
83	Pasture	70.3	21.5	8.2
84	Pasture	38.2	41.0	20.8
86	Pasture	67.2	20.2	12.6
93	Pasture	73.4	15.0	11.6
95	Pasture	51.3	14.5	34.2
98	Pasture	60.7	22.5	16.8
100	Pasture	71.8	16.5	11.7
106	Pasture	68.1	18.5	13.4
109	Pasture	80.5	12.0	7.5
111	Pasture	71.5	22.0	6.5
120	Pasture	77.2	16.0	6.8
123	Pasture	40.8	16.5	42.7
124	Pasture	70.0	20.0	10.0
131	Pasture	75.3	20.5	4.2
138	Pasture	56.0	29.0	15.0
144	Pasture	62.7	26.7	10.6
149	Pasture	70.8	23.0	6.2
153	Pasture	59.4	18.5	22.1
45	Row crop	63.5	19.0	17.5
51	Row crop	72.4	20.5	7.1
65	Row crop	54.4	36.0	9.6

Sample ID	Source	Sand (%)	Silt (%)	Clay (%)
66	Row crop	65.3	25.5	9.2
81	Row crop	66.7	22.3	11.1
88	Row crop	65.4	19.0	15.6
91	Row crop	68.6	18.5	12.9
103	Row crop	69.0	22.0	9.0
122	Row crop	69.8	16.0	14.2
125	Row crop	64.9	22.5	12.6
127	Row crop	65.4	23.0	11.6
133	Row crop	59.5	29.3	11.3
134	Row crop	55.2	34.0	10.8
136	Row crop	63.5	26.5	10.0
151	Row crop	48.9	15.0	36.1
190	Sediment	33.2	37.8	29.1
191	Sediment	30.8	46.8	22.4
192	Sediment	30.1	46.2	23.7
193	Sediment	8.2	61.9	29.9
194	Sediment	8.5	61.6	29.8
200	Sediment	15.4	53.8	30.7
201	Sediment	20.2	51.5	28.3
203	Sediment	33.1	47.7	19.2
204	Sediment	37.8	42.8	19.4
205	Sediment	40.2	50.0	9.8
206	Sediment	36.8	40.9	22.3
207	Sediment	33.3	45.5	21.2
208	Sediment	27.0	63.2	9.8
209	Sediment	29.1	64.6	6.3
210	Sediment	25.1	70.6	4.3
211	Sediment	23.7	65.4	10.9
212	Sediment	28.5	59.8	11.6
213	Sediment	32.2	55.0	12.8
214	Sediment	21.4	64.5	14.1
215	Sediment	19.6	63.7	16.7
79	Unpaved road	70.8	12.5	16.7
80	Unpaved road	63.1	17.8	19.2
85	Unpaved road	31.2	19.0	49.8
94	Unpaved road	61.7	12.2	26.1
104	Unpaved road	36.5	19.5	44.0
107	Unpaved road	47.6	14.0	38.4
108	Unpaved road	60.8	8.0	31.1
115	Unpaved road	50.9	17.0	32.1
116	Unpaved road	40.9	18.0	41.1
117	Unpaved road	49.0	14.0	37.0
128	Unpaved road	45.3	20.5	34.2

APPENDIX D

DETAILS OF THE MOBILE CENTRIFUGE SEDIMENT SAMPLER

Mobile centrifuge unit

Centrifuge bowl with collected suspended sediment

Mobile centrifuge in operation

Mobile centrifuge suspended sediment sampler

The mobile centrifuge used to collect suspended sediment in this study consists of three components; the centrifuge, a water pump, and a gasoline motor that powered the other two components. Storm water having high sediment concentrations was pumped into the centrifuge by the water pump. Suspended sediment was collected in the centrifuge bowl by the centrifugal force created by the rotation of the centrifuge designed to spin at 3000 RPM. Cleaner water was discharged through the outlet of the centrifuge. The flow rate into the centrifuge was approximately 20 L per minute with a recovery of approximately 70%. Lower flow rates showed an increase in sediment recovery. The average time for collecting 100 g suspended sediment sample was about an hour. However, this varied depending on the stream sediment concentration and flow rate

Care to be taken while sampling

1. Before starting the equipment make sure that the throttle for the motor is in the high range
2. Make sure that the hoses are connected air tight
3. A strainer attached to the inlet hose can prevent debris being pumped into the centrifuge
4. Pumping water from very close to the stream bed can result in sand particles being sucked up by the hose that can cause damage to the cam and the impeller of the pump
5. After each sample collection, the centrifuge bowl has to be cleaned and wiped before reusing to prevent cracking of the plastic bowl

6. After use make sure that the system is switched to off position to prevent battery from getting discharged
7. Do not operate the pump dry for more than 30 seconds