QUANTIFYING PHOSPHORUS LOADS AND STREAMBANK EROSION IN THE OZARK HIGHLAND ECOREGION USING THE SWAT MODEL

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Phosphorus (P) and sediment are major pollutants of waterbodies in the United States. Recent research has shown that future management practices must consider legacy P, especially from streambanks. Identifying and quantifying legacy P sources are necessary to select the most cost-efficient conservation practices. The overall objectives of this research were to (1) develop and apply a mass balance and uncertainty analysis to determine the quantity and location of legacy P stored in the Eucha-Spavinaw and Illinois River watersheds, (2) apply the SWAT model to determine the current sources of P reaching Lakes Eucha and Tenkiller and the management practices required to meet Oklahoma numeric water-quality standards, (3) test and improve a proposed streambank-erosion routine for the SWAT model and (4) apply the improved streambank-erosion routine to determine the significance of streambank-derived P for the Barren Fork Creek watershed. The results of the P mass balance and uncertainty analysis found that an average of 7.0 kg ha^{-1} yr^{-1} P were added to the watersheds from 1925 to 2015 and approximately 80% was retained, predominantly in the soils and stream systems. SWAT proved to be a capable tool in the water-quality standard evaluation process and illustrated how a watershed-scale model, such as SWAT, can be used to provide critical information when developing watershed-based plans. At current conditions, the water-quality standard is being met for Lake Eucha and the Illinois River subwatershed above the point source discharge at Tahlequah, Oklahoma, but is exceeded below the point source and for the Flint Creek and Barren Fork Creek subwatersheds. To meet the water-quality standard, the proper management of cattle production and elevated soil test P are essential. Modifications to the streambank-erosion routine for the SWAT model improved model predictions. Although the process-based applied shear stress equation was the most influential modification, incorporating the top width, streambank depth and area-adjustment factor more accurately represented the measured irregular cross-sections. The improved routine predicted streambank erosion on the Barren Fork Creek with a relative error of 34% before calibration. During the study period, 47% of the total P entering the Barren Fork Creek derived from streambanks.

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CHAPTER I

INTRODUCTION

1.1 Background

Phosphorus (P) and sediment are major pollutants to streams, rivers and reservoirs worldwide (FAO, 2013). Nutrients, the second leading cause of water quality issues in the United States (US), have resulted in the impairment of 7,765 waterbodies in the US. Over a third of the impaired bodies on the 303(d) list due to nutrients are a direct result of total P (USEPA, 2015a). Though P is a required nutrient for both terrestrial and aquatic plants, the P component of animal manure is often over applied resulting in elevated P in runoff and elevated Soil Test P (STP) (Daniel et al., 1998). This can lead to the eutrophication of the receiving waterbodies causing algal blooms, oxygen depletion and the overall degradation of the water quality (Sims and Sharpley, 2005). Other major P additions to waterbodies include agricultural crop production, livestock, point sources and land use change.

Like P, sediment is a primary pollutant to surface waters and the fifth leading cause of water quality impairment in the US (USEPA, 2015a). Though erosion is a natural process, the rate of erosion has been accelerated due to anthropogenic activities, such as farming and urbanization. Although sediment loss from agricultural fields, deforestation and construction sites is significant, streambank erosion can be the most

significant contributor of sediment to rivers and streams in some watersheds (Simon and Darby, 1999; Walling et al, 1999; Simon et al, 1996). Excess sediment in streams and reservoirs is aesthetically displeasing (Pfluger et al., 2010), reduces water clarity (Neupane et al., 2015), increases water-treatment costs (Dearmont et al., 1998) and has an overall negative impact on the aquatic ecosystem (Lloyd, 1987).

Though P is usually the limiting nutrient in southern waterbodies and contributes to water-quality issues (Daniel et al., 1998), the total quantity and sources of P are often unknown. When addressing water-quality issues, the current sources of P, and the quantity and location of legacy P is vital information. Without this knowledge, the indiscrete implementation of conservation practices across a watershed will not maximize P reductions. While P mass balances have been completed for several watersheds in the US (Bennett et al., 1998; Fluck et al., 1992; Jaworski et al., 1992; Engel et al., 2013), none have quantified and identified the location of legacy P. For example, Bennett et al. (1998) quantified 1995 P inputs and outputs for the Lake Mendota watershed, while Fluck et al. (1992) and Jaworski et al. (1992) used static data to characterize the Lake Okeechobee and Upper Potomac watersheds. Though Engel et al. (2013) completed a mass balance from 1949 to 2002, they did not estimate the total quantity of stored P and its distribution across the watershed.

Once P sources in a watershed are determined, management plans are needed to efficiently address water-quality impairment and/or protection. States must establish priority rankings for their impaired waters on the 303(d) list and develop Total Maximum Daily Loads (TMDLs) (USEPA, 2015a). A TMDL is the maximum quantity of a pollutant that a waterbody can receive and still meet water-quality standards (USEPA,

2015a). Often water-quality models, such as the Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998) are used to develop the TMDLs (Manoj et al., 2010; Borah et al., 2006) and may also be a useful tool to develop strategies to reduce pollutant loads to meet water-quality standards. An increasing number of states are implementing numeric nutrient water-quality standards for streams/rivers, lakes/reservoirs and estuaries (USEPA, 2015b). Several of these management plans have been implemented in the last few years while a large number of plans will be adopted in the next five years. Examples of State nutrient standards include Colorado's 0.035 and 0.027 mg L^{-1} total P for the Cherry Creek and Chatfield Reservoirs, respectively, and Arizona's stream standards (USEPA, 2015b). In the US, the only study to use a hydrological model to develop a management plan to meet numeric nutrient water-quality standards for streams was Storm et al. (2010). No study has been published for a lake or reservoir in the US.

Streambanks can not only be a major source of sediment to streams and reservoirs, but also P (Kronvankg et al., 2012; Laubel et al., 2003; Miller et al., 2014). The failure to account for P from streambanks may deem a watershed-management plan and its implementation less effective. Predicting streambank erosion and the associated particulate P on a watershed scale presents several challenges. Watershed-scale models, such as SWAT, do not account for all of the governing chemical, physical and biological processes and may lack the required spatial and temporal detail. Though several limitations remain, the proposed 2015 streambank-erosion subroutine (Narasimhan et al., 2015) improved the model's ability to predict streambank erosion. The new routine has only been tested on cohesive soils in the Cedar Creek watershed in North-Central Texas. Therefore, more testing is needed before the routine is implemented into the official

SWAT release. If streambank erosion is significant in a watershed, it is imperative that watershed managers consider streambank erosion when developing TMDLs and watershed-based plans. With limited time and money, and the large amount of data required to accurately estimate streambank erosion, watershed managers need guidance in modeling and estimating the most sensitive parameters that affect streambank erosion.

1.2 Objectives and Overview

The overall objectives of this research were to (1) develop a method, using a detailed P mass balance and uncertainty analysis, to determine the quantity and location of legacy P at the watershed scale, (2) apply the SWAT model in the evaluation of numeric water-quality standards, (3) improve and test the proposed streambank-erosion routine for the SWAT model, and (4) apply the improved streambank-erosion routine to determine the significance of P derived from streambank erosion to the total P load at the watershed scale.

First, a method was developed to quantify and identify legacy P using a mass balance and uncertainty analysis (Chapter 2). This method, which can be easily transferred to other watersheds, was applied to two of Oklahoma's most studied and high priority watersheds, the Eucha-Spavinaw and Illinois River. The quantity of P stored in the soil, reservoirs and stream systems were successfully estimated.

By itself, identifying P sources does not provide the required information to quantify P reaching streams and reservoirs. The objectives of Chapter 3 were to use SWAT to (1) identify and quantify the current sources of P reaching Lakes Tenkiller Ferry and Eucha, (2) determine if Oklahoma was meeting the water quality standards and

(3) simulate and provide recommendations on the various management practices required to meet the numeric water-quality standards.

SWAT model results and other recent literature support the hypothesis that P modeling predictions for the Barren Fork Creek would improve with the incorporation of streambank erosion. In Chapter 4, the proposed streambank erosion routine for the SWAT model was improved and tested on composite streambanks at ten study sites on the Barren Fork Creek. Recommendations were given concerning channel-parameter measurement, application and influence on streambank erosion.

The modified streambank-erosion and in-stream P routines were used to simulate the streambank erosion and P for the Barren Fork Creek watershed (Chapter 5). Incorporating the improved streambank-erosion routine and streambank P data into the SWAT model improved the predictions for streambank erosion and the associated P for the Barren Fork Creek watershed. These results support previous research findings that streambanks can be significant source of total P.

CHAPTER II

QUANTIFYING LEGACY PHOSPHORUS USING A MASS BALANCE APPROACH AND UNCERTAINTY ANALYSIS

2.1 Abstract

Classic conservation practices may not address decades of phosphorus (P) accumulation, known as legacy P. Identifying and quantifying legacy P sources are necessary to identify the most cost-efficient conservation practices. A method was developed to quantify and identify legacy P at the watershed scale using a mass-balance approach and uncertainty analysis. The method was applied to two nutrient-rich watersheds in northeast Oklahoma and northwest Arkansas. Each P import and export to and from the two watersheds was identified and quantified using a probability distribution and uncertainty analysis. The P retained in the soils, reservoirs and stream systems were estimated from 1925 to 2015. Over 8.5 and 6.1 kg ha^{-1} yr^{-1} of P were added to the Illinois River and Eucha-Spavinaw watersheds with 53% and 55% from poultry production, respectively. Other major historical P sources were attributed to human population and commercial fertilizer use. Currently, the net addition of P in the watersheds is small due to the export of approximately 90% of the poultry litter. However, historically only 14 to 19% of all P imported to the Illinois River and Eucha-Spavinaw watersheds was removed via outflow from the reservoir spillways, and poultry litter and food exports.

The majority of the retained P is located in the soil, 3.6 to 5.8 kg ha^{-1} yr^{-1}, and stream systems, 0.01 to 3.0 kg ha^{-1} yr^{-1}.

2.2 Introduction

Excessive nutrients are a major pollutant to many waterbodies worldwide. The United States (US) has 7,765 nutrient-impaired waterbodies on the Clean Water Act 303(d) list. Over a third of the nutrient-impaired waterbodies are a direct result of excess total P (USEPA, 2015a). Major P sources typically include crop and livestock production, wastewater treatment plant (WWTP) effluent, animal manure and commercial fertilizer (Sims and Sharpley, 2005). Excess P entering streams, lakes and reservoirs can lead to eutrophication, resulting in algal blooms, oxygen depletion and the overall degradation of the water quality (Sims and Sharpley, 2005). Excessive algae can cause drinking water problems, including taste and odor issues (Blackstock, 2003), and decrease water clarity and overall aesthetics. As the short-lived algae die off, their decay process consumes dissolved oxygen and depletes the oxygen for other aquatic species. In severe cases, this can result in hypoxic conditions.

Billions of dollars have been spent on conservation measures in the US (Monke and Johnson, 2010), yet for several watersheds the outcome has not justified the cost (Sharpley et al., 2013). Years of P application has contributed to the accumulation of P, known as legacy P (Kleinman et al., 2011). This legacy P has built up over several decades in the soils, floodplains, streambanks, ditches and biomass (Sharpley et al., 2013). While billions of dollars have been spent on conservation practices, the allocation of those practices may not be in the correct location and at a sufficient scale and intensity (Sharpley et al., 2009). Though P in runoff is controlled by a number of processes,

distinguishing between P from sources, sinks and the effect of conservation practices is challenging. In order to better identify the location and type of conservation practices needed, identification and quantification of legacy P is necessary.

A mass balance can be used to both quantify and identify legacy P. P mass balances have been completed for several watersheds in the US (Bennett et al., 1998; Fluck et al., 1992; Jaworski et al., 1992; Engel et al., 2013), yet most use relatively short time periods. For example, Bennett et al. (1999) quantified 1995 P inputs and outputs for the Lake Mendota watershed, while Fluck et al. (1992) and Jaworski et al. (1992) used static data to characterize the Lake Okeechobee and Upper Potomac watersheds, respectfully. While a few studies have considered long-term trends in P budgets (MacDonald and Bennett, 2009; Engel et al., 2013), none have quantified and identified the location of legacy P. Engel et al. (2013) completed a mass balance from 1949 to 2002, but did not estimate the total quantity of stored P and its distribution across the watershed. MacDonald and Bennett (2009) calculated the P accumulation in soils from 1901 to 2001, but only accounted for agricultural P and assumed all P was stored in the soil. While there is often considerable uncertainty in the P inputs and outputs, few studies have accounted for this uncertainty (Bennett et al., 1999) and none have applied a Monte Carlo analysis.

The objectives of this research were to (1) develop a method to quantify and identify legacy P at the watershed scale and (2) apply a Monte Carlo analysis to quantify the uncertainty in the P inputs, outputs and sinks. This method was applied to two P-rich watersheds in northeast Oklahoma and northwest Arkansas, the Eucha-Spavinaw and Illinois River watersheds.

2.2 Methodology

Three steps are needed to quantify and identify legacy P for a watershed: (1) identify and quantify P imports and exports to and from the watershed, (2) estimate P stored in soils, reservoirs and stream systems and (3) characterize all P imports and exports using probability distributions and an uncertainty analysis.

2.2.1 Phosphorus Imports and Exports

Each P import and export must first be identified (Figure 2.1). Typical P imports include humans, livestock, pets, industrial facilities, commercial fertilizer and atmospheric deposition. P exports from a watershed normally include harvested crops, cattle, manure export and discharge via the spillway. To avoid double-accounting, it is assumed that all food for humans and livestock are imported into the watershed, while all harvested crops within the watershed are exported outside the watershed.

Typical US sources of P imports and exports can be found in Table 2.1. Some sources, such as commercial fertilizer and manure export, vary by state. Beef cattle are the only livestock considered for P export since they graze on pasture, and thus remove P stored in the soil. Grazing livestock do not add P to the watersheds, but recycle it (Slaton et al., 2004; Engel et al., 2013); however, beef cattle are usually given imported supplements during the winter months. Eggs and milk are not considered exports because P added from poultry and dairy cattle is calculated from manure, not imported feed.

Often data is not available for the imports and exports for the entire period of the mass balance. For example, Agriculture Census data is reported every four to five years. In this case, each available data point can be graphed and a regression equation derived.

This can then be used to estimate the livestock populations between census years. Flow and P data from US Geological Survey gage stations are often not available for the full extent of the study. When data is not available, it can be assumed that the recorded data is representative of the entire study period or a hydrological model can be used to estimate the flow and P for the missing dates.

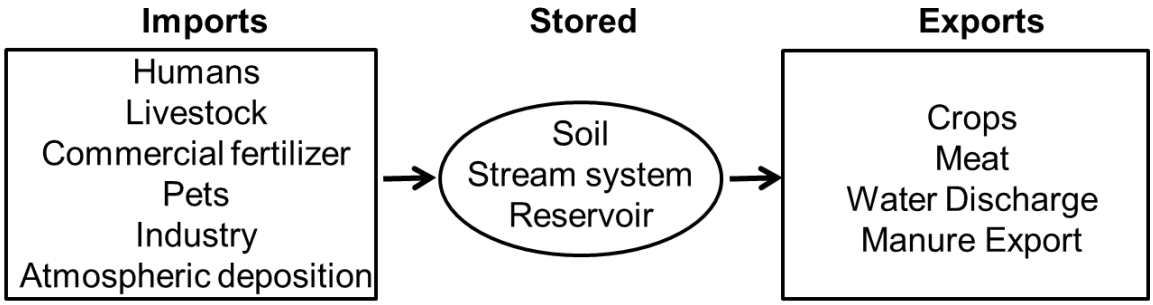

Figure 2.1. Typical phosphorus imports, exports and stored legacy phosphorus in a watershed.

Table 2.1. Typical phosphorus (P) imports, exports and their sources to and from a watershed.

Data	Source
Phosphorus Imports	
Human population	United States Census Bureau http://www.census.gov/
Human P content	Sarac et al. (2001); Crites and Tchobanoglous (1998)
Pet population and P content	Baker and Czarnecki-Maulden (1991)
Livestock population	Agricultural Census: http://www.agcensus.usda.gov/
Livestock P content	ASABE (2005) D384.1 standard
Industrial sources	http:/www.epa.gov/enviro/html/pcs/pcs_query_java.html
Commercial fertilizer	Varies by state
Atmospheric deposition	Newman (1995)
Phosphorus Exports	
Crops harvested	Agricultural Census: http://www.agcensus.usda.gov/
Cattle supplement	Gill and Lusby (2003)
Manure export	Varies by watershed
Reservoir spillway or river discharge	www.usgs.gov/water/

2.2.2 Phosphorus Stored in Soil, Reservoirs and Stream Systems

P stored in a watershed is defined as the difference between P imports and exports. All stored P is assumed to be located in the reservoirs, soil or stream system (Figure 2.1). Although P stored in the stream system is difficult to quantify due to limited data, P in the soil and reservoirs can be estimated using Mehlich III soil-test P

(STP) and US Geological Survey (USGS) flow and stream P data. After subtracting P stored in the soils and reservoirs from the total P retained in the watershed, all remaining P is assumed to be retained in the floodplains, streambanks and stream benthos.

The over-application of livestock manure and commercial fertilizer results in P accumulation in soils. MacDonald and Bennett (2009) found a positive trend with surplus P and STP. Sharpley et al. (2010) found that 7 to 15 mg P_2O_5 per ha soil was required to increase the STP by one mg P per kg of soil, depending on the soil properties. Two major assumptions of this method are the following: available STP data are representative of the land cover and watershed and the assumed quantity of P_2O_5 required to increase STP is correct.

Estimates of P retained in reservoirs can be attained by using USGS gage station data upstream and downstream of the reservoir. This is a good assumption if the two gages are immediately upstream and downstream of the reservoir. If this data is not available, watershed or reservoir models can be used.

2.2.3 Uncertainty Analysis

Monte Carlo simulations are conducted to account for the uncertainty and assumptions in P imports and exports, and stored P. A probability distribution for each variable is assigned using software, such as Minitab (Golden Software, 2002) and @RISK (Palisade Corporation, 2015). A uniform distribution is selected when only estimates of data range are known, such as atmospheric deposition. For example, the world wide P contribution by atmospheric deposition ranged from 0.07 to 1.7 kg ha^{-1} yr^{-1} (Newman, 1995). A normal distribution should be selected when the sample mean, standard deviation or coefficient of variation are reported. For example, the average P

content for broilers is 0.30 kg per 1,000 kg of broilers with a standard deviation of 0.05 kg per 1,000 kg of broilers (ASABE, 2005). A triangular distribution is selected when only the range and the mean or the mode is available. Many times the minimum and maximum distribution values are estimated based on professional judgment. Finally, a confidence interval must be chosen to conduct the Monte Carlo analyses.

2.3 Case Study

2.3.1 Study Area

The State of Oklahoma has 657 waterbodies on the 303(d) list (USEPA, 2015a), with several of these waterbodies located in the Ozark Highlands Ecoregion, e.g. the Illinois River and Eucha-Spavinaw watersheds (Figure 2.2). The Illinois River watershed encompasses an area of approximately 4,440 km^2 in northeast Oklahoma and northwest Arkansas with 53% in Oklahoma and 47% in Arkansas. The largest rivers and tributaries are the Illinois River, the Barren Fork Creek and Flint Creek. The Illinois River flows into Tenkiller Ferry Lake in Cherokee County, Oklahoma and is arguably one of Oklahoma's most valued rivers. The National Land Cover Database (NLCD) (NLCD, 2006) characterized the watershed into 15 land cover classes, with pasture (45%) and forest (42%) being the dominant land covers. The Eucha-Spavinaw watershed, adjacent to the Illinois River watershed to the north, encompasses an area of approximately 1,100 km^2 with 63% in Oklahoma and 37% in Arkansas. The watershed spans two counties before emptying into Lake Eucha, which then flows to Spavinaw Lake. The largest streams in the watershed are Spavinaw and Beaty Creeks. The watershed was also predominantly pasture (45%) and forest (48%) (NLCD, 2006). The two largest agricultural producers in the watersheds were cattle and poultry. Though these two

watersheds were predominantly forest and pasture, urbanization has increased substantially in northwest Arkansas. Seventy percent of the developed land in the watersheds is in Benton and Washington Counties, Arkansas, while 66% of the forested area is within the four Oklahoma counties.

Historically both watersheds were valued by Oklahomans for their high water quality (Brill, 1957). The Illinois River and its receiving water body, Tenkiller Ferry Lake, provide recreational benefits for hundreds of thousands of people each year, and float trips on the river provide about $9 million per year in direct economic impact (Soerens et al., 2003). The Illinois River along with Flint and Barren Fork Creeks were designated as "Scenic River Areas" in 1970 by the Oklahoma Legislature. Lakes Eucha and Spavinaw provide drinking water for the Cities of Tulsa and Jay, Oklahoma and Rural Water District 1. Several thousand people visit Eucha State Park each year to camp, fish and boat (Wagner and Woodruff, 1997).

Fifty years ago, the streams and reservoirs in the Illinois River and Eucha-Spavinaw watersheds were clear with exceptional water quality (Brill, 1957; Cooke et al., 2011). The reservoirs were oligotrophic and the phytoplankton was dominated by diatoms (Cooke et al., 2011). Currently the Illinois River, along with its tributaries Barren Fork Creek and Flint Creek, are on the Oklahoma 303(d) list of impaired waters due to elevated P (DEQ, 2012), the streams and reservoirs are eutrophic and the dominant algae is cyanobacteria (Cooke et al., 2011). The average annual total P concentrations entering Lake Tenkiller has increased from 20 $\mu g\ L^{-1}$ after the dam closure in 1952 to 163 $\mu g\ L^{-1}$ from 1976-1985, and to 210 $\mu g\ L^{-1}$ from 1997 to 2006 (Cooke et al., 2011). The Oklahoma Conservation Commission (OCC, 1997) reported the total P in Lake Eucha

increased three-fold from 1975 to 1995. The increased P load led to excessive algae growth in the two watersheds causing Lakes Tenkiller Ferry, Eucha and Spavinaw to be added to the 303(d) list due to excessive chlorophyll-a and low dissolved oxygen, and Flint Creek due to low dissolved oxygen (USEPA, 2015a).

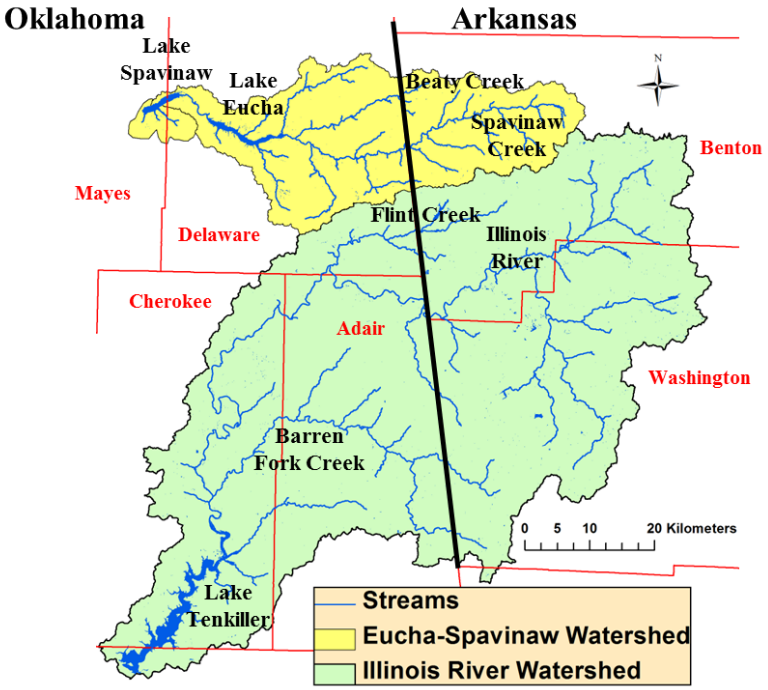

Figure 2.2. Illinois River and Eucha-Spavinaw watersheds in northeast Oklahoma and northwest Arkansas showing county boundaries (red), the Oklahoma/Arkansas state line (black) and the major streams and reservoirs (blue).

2.3.2 Case Study Methods

2.3.2.1 Phosphorus Imports and Exports

There were six major P imports and four major P exports to and from the Eucha-Spavinaw and Illinois River watersheds (Table 2.2). The US Census (U.S. Census, 2012) was used to estimate the human population for each county represented in the watersheds for the years 1960 to 2010. Regression equations (R^2=0.99, 0.98) were used to estimate the human population from 1925 to 1960 and to interpolate between census years for the Illinois River (Equation 2.1) and Eucha-Spavinaw (Equation 2.2) watersheds:

$$HP = 6.82 * 10^{23} * e^{0.028 * x} \quad (2.1)$$

$$HP = 309 * x - 5.99 * 10^{5} \quad (2.2)$$

where *HP* is the population and *x* is the year, e.g. 1980. The urban population has grown exponentially in the Illinois River watershed compared to the slow steady rural population growth in the Eucha-Spavinaw watershed (Figure 2.3). The number of recreational visitors and population of cats and dogs were assumed to be a proportional to the watershed population. Though the number of visitors to the Eucha-Spavinaw watershed was relatively small, the Illinois River watershed had 2.6 million annual visitors to Tenkiller Ferry Lake and another 160,000 that floated the Illinois River (Caneday, 2008; Smith et al., 2008). Based on Wagner and Woodruff (1997), 75% of these visitors were assumed to be from outside the watershed and stayed for an average of 24 hours.

Table 2.2. Phosphorus (P) imports, exports and their sources to and from the Eucha-Spavinaw and Illinois River watersheds.

Data	Source
Phosphorus Addition	
Human population	US Census Bureau http://www.census.gov/
Human P content	Sarac et al. (2001); Crites and Tchobanoglous (1998)
Pet population and P content	Baker and Czarnecki-Maulden (1991)
Livestock population	Agricultural Census: http://www.agcensus.usda.gov/
Livestock P content	ASABE (2005) D384.1 standard
Industrial sources	http:/www.epa.gov/enviro/html/pcs/pcs_query_java.html
Commercial fertilizer	Oklahoma Department of Agriculture and Arkansas State Board
Atmospheric deposition	Newman (1995)
Phosphorus Removal	
Crops harvested	Agricultural Census: http://www.agcensus.usda.gov/
Cattle supplement	Gill and Lusby, 2003
Manure export	ODAFF (2014); ANRC (2014)
Spillway Discharge	www.usgs.gov/water/

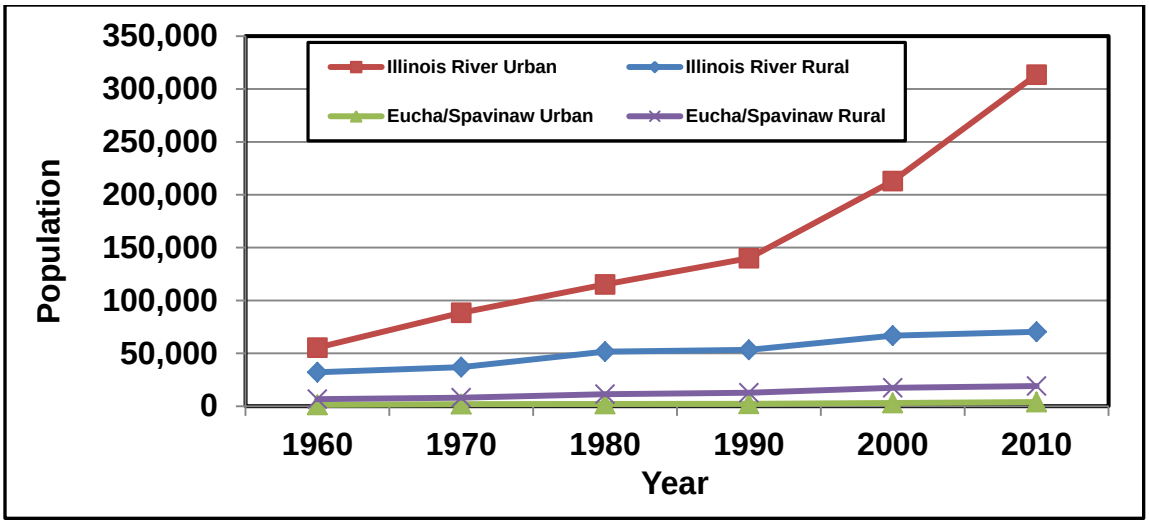

Figure 2.3. Urban and rural populations from 1960 to 2010 for the Illinois River and Eucha-Spavinaw watersheds (U.S. Census, 2012).

Historical data from the US Census of Agriculture (Ag Census) (USDA Census of Agriculture, 2012) were analyzed from 1925 to 2012 to estimate livestock populations (Table 2.3). Until recently, the population of beef cattle had increased steadily over time. The recent decline in cattle was likely due to the drought from the fall of 2010 through the summer of 2011 (Brock et al., 1995; McPherson et al., 2007) and the reduction of freely available nitrogen due to the export of poultry litter (ODAFF, 2014; ANRC, 2014). Populations of dairy cattle and swine declined during the latter part of the period in both watersheds. Until recently, broiler populations in both watersheds had increased exponentially with time, while the other poultry populations have been on the decline. The decline from 2007 to 2012 in the broiler population in the Illinois River watershed was attributed to reduction in broiler farms in Washington County, Arkansas. Based on the American Society of Biological Engineers D384.1 standard (ASABE, 2005), 1,000 kg of layers and broilers produce 0.30 kg P d^{-1}, while turkeys produce 0.23 kg P d^{-1}, dairy cattle 0.094 kg P d^{-1} and swine 0.18 kg P d^{-1}.

Table 2.3. Livestock populations in the Illinois River and Eucha-Spavinaw watersheds from 1925-2012 (USDA Census, 2012). NR: not reported; *Estimated based on population of poultry layers

Year	Beef Cows and Heifers that Calved	Dairy Cattle	Swine	Broilers	Layers	Pullets	Turkeys
Illinois River Watershed							
1925	24,128	14,202	31,289	570,712	447,742	146,815*	NR
1930	21,287	18,518	NR	NR	384,395	126,043*	NR
1935	30,058	23,669	NR	756,515	379,124	124,315*	1,390
1940	29,190	22,203	58,399	2,270,728	509,226	166,975*	6,727
1945	41,090	27,003	61,112	4,330,060	720,652	236,302*	13,735
1949	39,080	29,181	77,408	12,014,012	352,355	115,537*	39,087
1954	48,643	29,765	37,875	18,744,529	170,250	55,825*	303,369
1959	49,745	21,223	50,356	35,926,405	377,997	123,945*	491,716
1964	59,155	14,897	28,325	60,976,942	3,331,453	1,092,384*	916,334
1969	70,931	11,727	44,239	76,042,425	6,677,357	2,189,507*	1,403,003
1974	94,556	9,331	57,164	81,055,773	3,882,739	1,273,151*	1,478,477
1978	89,507	11,798	213,493	87,352,067	6,371,409	2,294,074	1,174,098
1982	97,654	15,466	284,131	91,942,053	7,826,957	2,300,576	2,918,742
1987	93,303	13,338	486,229	100,554,578	9,405,063	2,430,760	5,502,684
1992	96,333	12,179	325,748	125,626,732	5,114,961	2,356,175	4,041,947
1997	105,215	9,820	317,042	127,688,339	6,442,041	1,948,465	4,814,157
2002	110,242	10,258	207,246	140,655,072	5,653,783	962,075	4,055,066
2007	103,388	5,978	141,850	148,150,905	3,634,244	1,678,596	3,304,178
2012	83,707	3,164	1,347	137,389,647	3,102859	1,253,798	2,360,687
Eucha-Spavinaw Watershed							
1925	6,179	3,394	9,452	143,604	111,127	37,042*	NR
1930	5,526	4,846	NR	NR	101,781	33,927*	NR
1935	7,583	6,267	NR	233,375	99,191	33,064*	237
1940	7,043	5,392	20,226	643,860	137,557	45,852*	549
1945	10,500	6,374	18,818	1,119,422	179,515	59,838*	4,669
1949	9,405	7,247	21,847	2,784,422	88,850	29,617*	12,657
1954	11,983	7,257	11,108	4,093,201	42,840	14,280*	41,728
1959	11,600	5,705	16,377	7,542,528	73,480	24,493*	114,195
1964	14,239	3,986	10,559	10,911,140	413,460	137,820*	136,621
1969	16,537	3,101	13,365	7,542,528	648,738	216,246*	336,291
1974	21,449	2,273	13,568	11,687,173	637,088	212,363*	466,086
1978	20,833	3,008	44,316	11,718,528	788,383	366,687	235,137
1982	21,578	3,401	71,676	14,878,831	968,486	181,840	808,347
1987	20,962	3,034	119,510	19,705,455	1,153,192	320,723	1,701,966
1992	21,550	2,518	79,107	27,807,167	892,975	241,379	939,224
1997	24,383	2,029	115,053	32,770,800	995,557	222,369	1,196,586
2002	26,388	1,662	74,504	38,420,654	868,096	195,152	928,144
2007	24,459	1,221	451	38,720,630	1,182,788	272,952	506,391
2012	21,261	951	104	39,933,390	1,041,236	207,856	377,751

Annual P commercial fertilizer sales for the six counties in the watersheds peaked at approximately 4,500 tons of P_2O_5 in 1966 and declined to 290 tons in 2012. The decline was believed to be attributed to increased poultry litter availability, decreased row

crop production, increased fertilizer prices, and education on proper fertilizer application rates.

Since USGS data was not available immediately upstream of the reservoirs, calibrated SWAT models for the Illinois River and Eucha-Spavinaw watersheds (Storm and Mittelstet, 2015) were used to estimate P loads entering Lakes Tenkiller Ferry and Eucha. USGS gage stations and City of Tulsa water-quality stations were then used to estimate P leaving the watersheds and stored in the reservoirs. The P load entering Spavinaw Lake was calculated using the sum of P leaving Lake Eucha and the SWAT-predicted P entering from the upland areas downstream of the Lake Eucha spillway. Approximately 50% of the P entering Spavinaw Lake was assumed to be retained in the reservoir (OWRB, 2002). Using these data, coupled with water withdrawn data by the City of Tulsa, Oklahoma and other small towns, the quantity of P stored in the reservoirs was estimated. Due to lack of data prior to the installation of the USGS gage stations, the quantity of crops and cattle from the 1925 Ag Census were used to develop a revised land cover data set for the two watersheds. The revised land cover and average precipitation for the watersheds and baseline STP were used to estimate P entering and leaving the reservoirs in 1925. The P concentrations were then interpolated from 1925 to the beginning of the recorded USGS data.

Following litigation in Federal Court (Case No. 01-CV-0900) and the resulting 2003 settlement agreement between the City of Tulsa (plaintiff) and the poultry industry and the City of Decatur, Arkansas (defendants), there were extensive changes in agricultural management, reduction in point source P discharge (White et al., 2011) and the export of poultry litter from the Eucha-Spavinaw watershed (Sharpley et al., 2012).

BMPs, Inc., a nonprofit entity created as part of the settlement agreement, began facilitating litter export from the Eucha-Spavinaw watershed in 2003 and then expanded their program to the Illinois River watershed in 2004 (Fisher et al., 2009). Current litter exports were obtained from the Oklahoma Poultry Waste Report (ODAFF, 2014) and the Arkansas 2013 Poultry Waste Report (ANRC, 2014). Approximately 89% of the litter was exported from the Oklahoma portion of the Illinois River watershed in 2011 compared to 86% in the Eucha-Spavinaw watershed in 2012 (ODAFF, 2014). Note that the Arkansas Poultry Report presented data by county and not by watershed. In 2013, Benton and Washington Counties in Arkansas generated over 200,000 and 155,000 tons of poultry litter and applied only six and ten percent, respectfully (ANRC, 2014). Litter export data were not available from 2007 to 2009 and 2011 for the Illinois River watershed and 2010 to 2012 for Eucha-Spavinaw watershed; therefore, regression equations were used to estimate the percent of litter export for those years.

The Agricultural Census was used to identify the crops and yields from 1925 to 2007 for each county within the watersheds. Crop production declined over the time period (Figure 2.4). The percentage of each crop harvested within the watershed was assumed to be the same as the percentage of crops within the watershed from the 2006 NLCD land cover dataset. Although hay is a major crop in the watershed, all hay was assumed to remain in the watershed for foraging livestock. A Crop Nutrient Tool (USDA, 2012) was used to estimate the quantity of P removed from the watersheds using default parameters (Table 2.4).

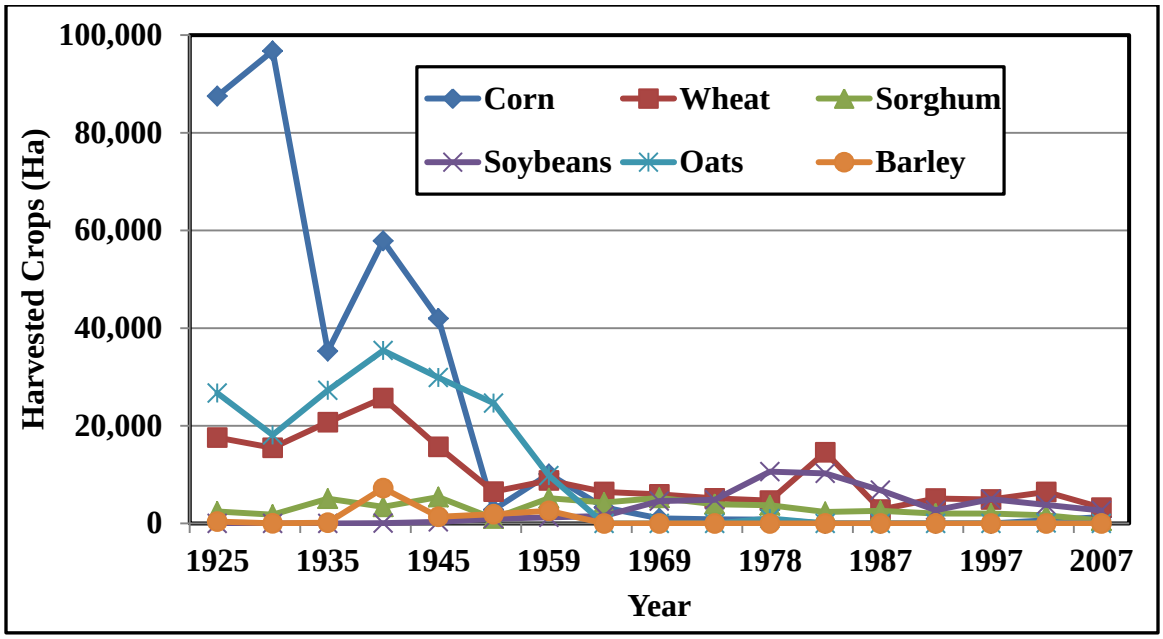

Figure 2.4. Harvested crops for the Oklahoma and Arkansas counties of Adair, Benton, Cherokee, Delaware and Washington from 1925 to 2007 (Ag Census, 1925 to 2007).

Table 2.4. Default phosphorus and moisture contents used in the Crop Nutrient Tool (USDA, 2012) for six historically major crops in the Illinois River and Eucha-Spavinaw watersheds.

Crop	Corn	Wheat	Sorghum	Soy Beans	Oats	Barley
Phosphorus Content, Wet Basis (kg/m^3)	2.0	2.0	2.1	4.7	1.4	1.9
Moisture Content (%)	13.5	10.2	11.2	10.1	10.7	13.3

2.3.2.2 Phosphorus Stored in Soil, Reservoirs and Stream System

Soil P was estimated using over 50,000 STP samples across the watersheds. The average Mehlich III STP for pasture, crops and urban Bermuda grass were obtained from 1994 to 2008 and 2000 to 2011 from the Oklahoma and the Arkansas soil testing labs, respectively (Table 2.5). Based on soil properties of the Illinois River and Eucha-Spavinaw watersheds, 13 kg P per ha was assumed to be required to increase the STP by one mg P per kg of soil (Sharpley, A., personal communication, June 13, 2012).

A baseline STP of 15 mg P per kg of soil was assumed in 1925, which corresponded to the measured background level in the John T. Nickel Family Nature and Wildlife Preserve in the Illinois River watershed (Engel et al., 2013). For each county

and each land use, the following equations were applied to calculate the P added to the soils from 1925 to 2015:

$$\Delta STP\left(\frac{mg\ P}{kg\ soil}\right) = STP\left(\frac{mg\ P}{kg\ soil}\right) - Baseline\ STP\left(\frac{mg\ P}{kg\ soil}\right) \tag{2.5}$$

$$Added\ P\ to\ Soil\left(\frac{kg\ P}{ha}\right) = \Delta STP\left(\frac{mg\ P}{kg\ soil}\right) * 13\left(\frac{kg\ P/ha}{mg\ P/kg\ soil}\right) \tag{2.6}$$

where ΔSTP is the historic increase in STP. For example, Adair County had 45,530 ha of pasture with an average STP of 67 mg P per kg of soil. Based on equation 2.5, the change in STP was 67-15 or 52 mg P per kg of soil. The total P added to the soil was 52*13 or 680 kg per ha, which equates to 30,800 Mg of excess P stored in the pasture soil in Adair County.

Table 2.5. Average Mehlich III Soil Test Phosphorus (mg/kg) for pasture, crops and urban lawns for six counties in the Illinois River and Eucha-Spavinaw watersheds based on data from 1994 to 2008 for Oklahoma (Oklahoma State University Soil, Water, and Forage Analytical Laboratory) and 1999 to 2011 for Arkansas (University of Arkansas Soil Testing Laboratory).

Land Cover	**County**					
	Oklahoma				Arkansas	
	Adair	Cherokee	Delaware	Sequoyah	Benton	Washington
Pasture	67	38	52	22	210	150
Crops	94	27	62	33	143	125
Urban Lawns	111	180	228	111	191	153

2.3.2.3 Uncertainty Analysis

Triangular, normal or uniform probability distributions were assigned to each variable (Tables 2.6 and 2.7). The @RISK software (Palisade Corporation, 2015) was used to perform Monte Carlo simulations for each variable annually for each watershed. For each variable, 10,000 realizations were performed to characterize the uncertainty and estimate the 90% confidence intervals for the P stored in the soils, reservoirs and streams systems.

Table 2.6. Assumed underlying probability distribution, and minimum, maximum and mode for variables used in the uncertainty analysis of the phosphorus (P) mass balance for the Illinois River and Eucha-Spavinaw watersheds. The minimum and maximum percentages represent the minimum and maximum population or P content. var. = variable

Parameter	Range			Distribution
	Minimum (%)	Maximum (%)	Mode	
Human Population	-10	10	var.	Triangular
Visitors to Watersheds	-25	25	var.	Triangular
Pets	-50	50	var.	Triangular
Commercial Fertilizer	-20	20	var.	Triangular
P Entering Reservoirs	-25	25	var.	Triangular
P Leaving Lake Tenkiller	1	100	1%	Triangular
P Leaving Lake Eucha	5	100	5%	Triangular
P Leaving Lake Spavinaw	40	60	50%	Triangular
Poultry Litter Export	-20	20	var.	Triangular
P Content in Crops	-10	10	var.	Triangular
Deer Population and P Content	-20	20	var.	Triangular
Industrial Facilities	-20	20	var.	Triangular
Human P (kg yr^{-1})	0.5	1.2	N/A	Uniform
Atmospheric Deposition (kg ha^{-1} yr^{-1})	0.07	1.7	N/A	Uniform
Beef Cattle Supplement (kg yr^{-1})	2.5	2.95	N/A	Uniform
Soil Test P (mg kg^{-1})	3.0	4.8	N/A	Uniform

Table 2.7. Coefficient of variation for the livestock population and crop yields and standard deviation for phosphorus (P) content used in the uncertainty analysis of the P mass balance for the Illinois River and Eucha-Spavinaw watersheds.

Parameter	Coefficient of Variation (%)		Distribution
	Illinois River Watershed	Eucha-Spavinaw Watershed	
Broiler population	16.0	8.5	Normal
Layer population	20.3	14.4	Normal
Turkey population	9.8	10.5	Normal
Beef cattle population	4.9	4.4	Normal
Swine population	27.0	12.9	Normal
Dairy cattle population	6.8	1.9	Normal
Corn yield	1.8	1.6	Normal
Wheat yield	8.3	8.0	Normal
Sorghum yield	24.3	18.3	Normal
Soybeans yield	11.0	10.9	Normal
Oats yield	13.5	13.5	Normal
Barley yield	44.1	44.1	Normal
Parameter	**Standard Deviation (kg P per 1000 kg live animal mass d^{-1})**		Distribution
Broiler P content	0.053		Normal
Layer P content	0.081		Normal
Turkey P content	0.093		Normal
Swine P content	0.100		Normal
Dairy cattle P content	0.024		Normal

2.3.3 Results and Discussion

2.3.3.1 Phosphorus Imports

Table 2.8 provides a summary of the total P imports, removals and sinks from 1925 to 1990 and 1991 to 2015. The 90 years of data were split into two periods to illustrate the historical versus current P imports and exports. Based on a 90% confidence interval, 6.4 to 8.7 kg ha^{-1} yr^{-1} P were imported into the Illinois River watershed from 1925 to 2015 (Figure 2.5), with approximately 40% added in the last 25 years. From 1925 to 1990, the largest imports of P were poultry feed, commercial fertilizer and dairy cattle feed (Figure 2.6). In the last 25 years, poultry feed was the most dominant P import as the number of dairy cattle and application of commercial fertilizer declined. In 2015, poultry feed added 68% of all P to the watershed with 54% from broilers. For the Eucha-Spavinaw watershed, 5.6 to 7.5 kg ha^{-1} yr^{-1} P was added from 1925 to 2015, with 43% added since 1991. Poultry feed, dairy cattle feed and atmospheric deposition were the largest sources of P from 1925 to 1990 (Figure 2.7). In the last 25 years, poultry feed accounted for 75% of all P imports with two-thirds from broilers. P inputs to eleven watersheds in Minnesota ranged from 0.32 to 6.0 kg ha^{-1} yr^{-1} (Schussler et al., 2007). Though only 45% of the Illinois River and Eucha-Spavinaw watersheds are agricultural compared to up to 90% for the Minnesota watersheds, the large number of poultry drastically increases the P input to the watersheds. The P input to the Lake Mendota watershed in Wisconsin was over twice the amount as the Illinois River and Eucha- Spavinaw watersheds with 19 kg ha^{-1} added in 1995, primarily from commercial fertilizer (Bennett et al., 1999).

Table 2.8. Phosphorus imports, exports and sinks from 1925 to 1990 and 1991 to 2015 for the Illinois River (IRW) and Eucha-Spavinaw (ESW) watersheds.

Watershed	Time Period	P Import (kg ha^{-1} yr^{-1})	P Export (kg ha^{-1} yr^{-1})	Storage Sink	P Storage (kg ha^{-1} yr^{-1})
IRW	1925-1990	5.2-7.1	0.49-0.50	Tenkiller Ferry Lake	0.14-0.17
				Soil and stream system	4.6-6.6
				Total	4.7-6.6
	1991-2015	9.3-13.1	2.0-3.1	Tenkiller Ferry Lake	0.23-0.30
				Soil and stream system	7.0-10.0
				Total	7.2-10.3
ESW	1925-1990	4.4-5.9	0.28-30	Lake Eucha	0.020-0.024
				Lake Spavinaw	0.010-0.014
				Soil and stream system	4.0-5.5
				Total	4.1-5.6
	1991-2015	8.4-12.0	2.9-4.4	Lake Eucha	0.10-0.12
				Lake Spavinaw	0.03
				Soil and stream system	5.3-7.5
				Total	5.5-7.7

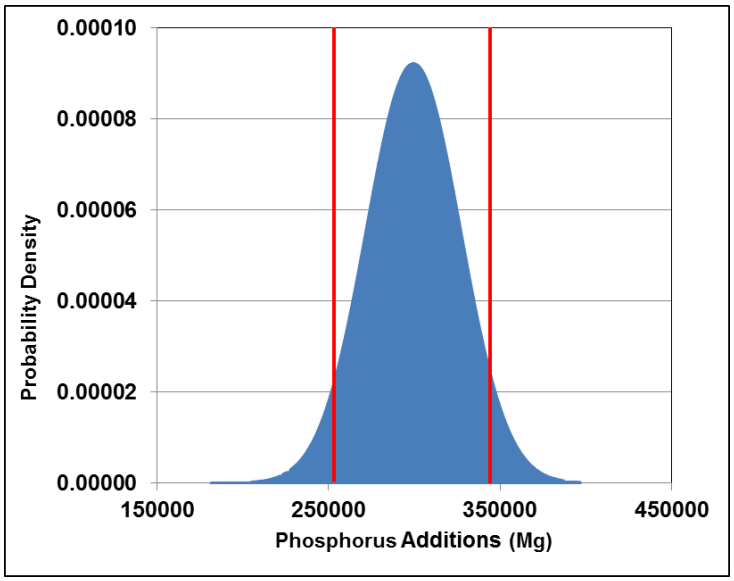

Figure 2.5. Probability density for total phosphorus added to the Illinois River watershed from 1925 to 2015, assuming a normal distribution. Red lines indicate 5th (left) and 95th (right) percentiles.

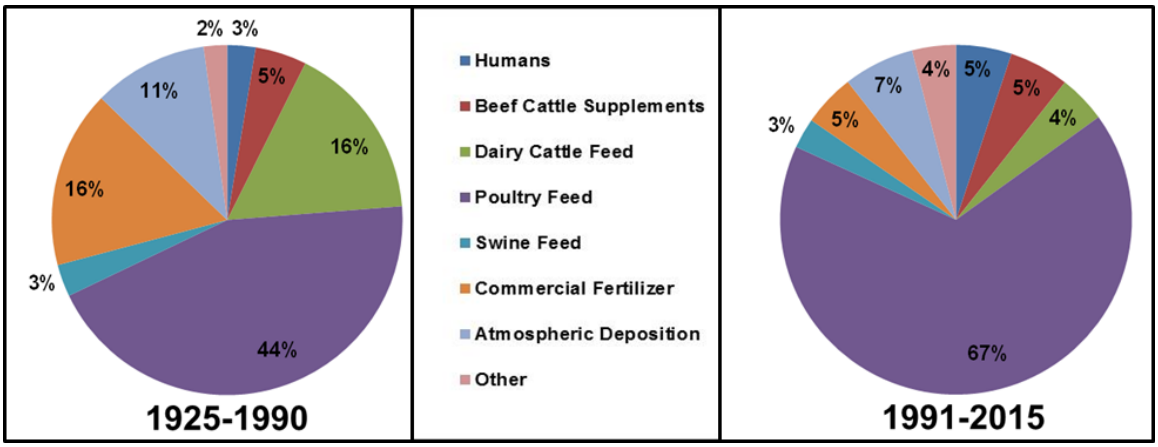

Figure 2.6. Imported phosphorus sources to the Illinois River watershed from 1925 to 1990 (left) and 1991 to 2015 (right).

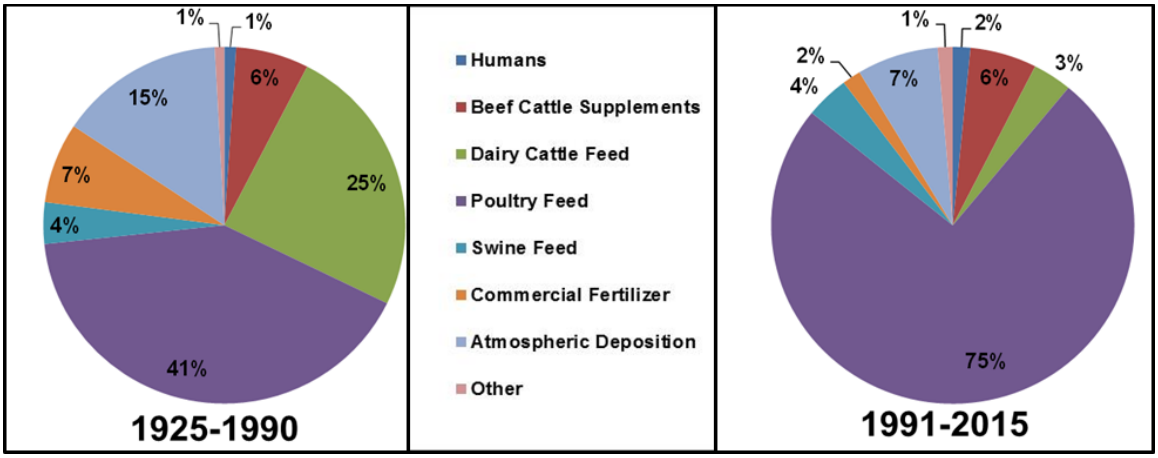

Figure 2.7. Imported phosphorus sources to the Eucha-Spavinaw watershed from 1925 to 1990 (left) and 1991 to 2015 (right).

2.3.3.2 Phosphorus Removals

From 1925 to 1990, an average of only 0.37 kg ha^{-1} yr^{-1} P was removed from the Illinois River watershed or 7% of all P imports. Forty percent was removed via beef cattle supplements, 41% from the Tenkiller Ferry Lake spillway and 18% from crops. Since 1991, 22% of all P was removed with over 75% via the export of poultry litter. The remaining removals were from beef cattle (16%) and via the spillway (7%). The net P imports peaked in the early 1990's and steadily declined until 2004 when litter began to be exported at an increasing rate (Figure 2.8). Currently 64% of all P added to the

watershed is removed with the majority due to the export of poultry litter. Of the 1,700 Mg of net P added in 2015, 27% was from poultry feed and 22% from humans.

The results were similar for the Eucha-Spavinaw watershed. From 1925 to 1990, only 6% of all P imports were removed. Most of the P removals were from crops (64%) and the export of beef cattle (32%). Only 4% of the P removed was via the Lake Spavinaw spillway. This was comparable to the 4.6% hydrologic P export in the Lake Mendota watershed (Bennett et al., 1999). Since 1991, 36% of all P added to the watershed was removed with 90% from poultry litter export. Currently 76% of P added to the watershed was removed. Due to the relatively small human population, the net P added to the Eucha-Spavinaw watershed is approaching the 1925 levels (Figure 2.8). Of the 300 Mg of net P added to the watershed in 2015, 39% was from poultry feed, 28% from atmospheric deposition and 18% from beef cattle.

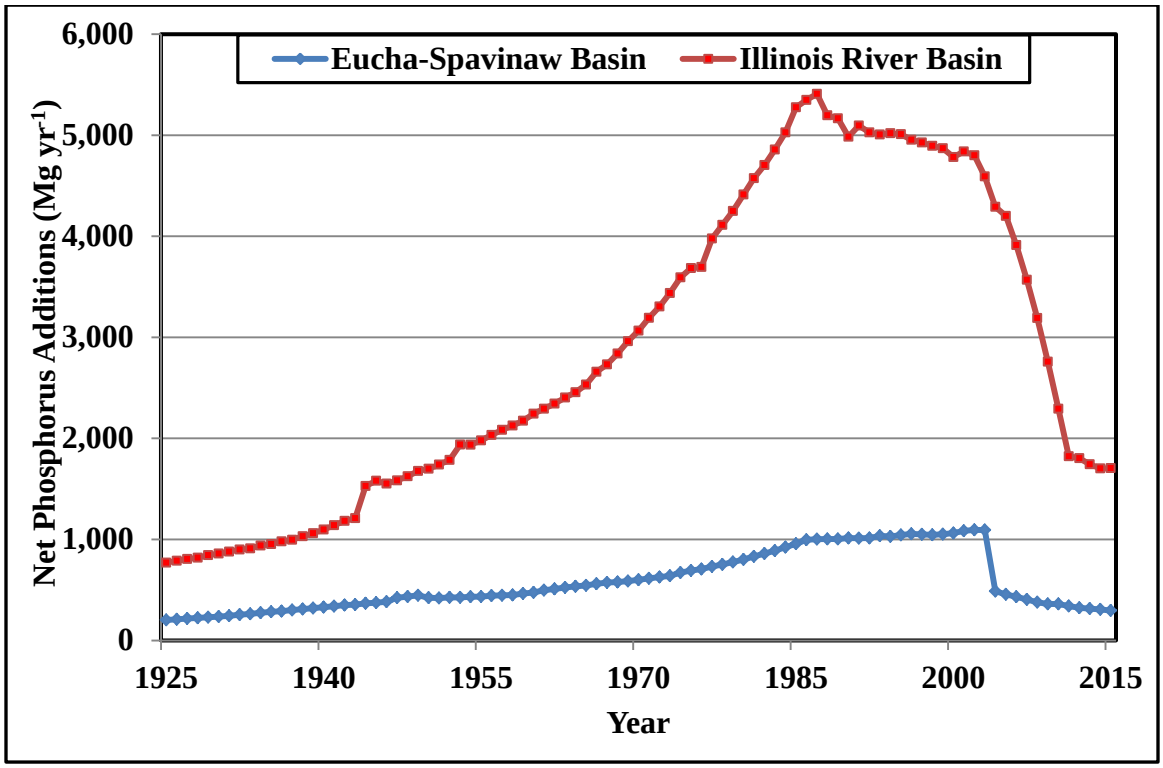

Figure 2.8. Net phosphorus additions to the Illinois River and Eucha-Spavinaw watersheds from 1925 to 2015.

P surplus in the Illinois River watershed was the highest in 1990 with 5,500 Mg yr^{-1} or 13 kg ha^{-1} yr^{-1}. In 2002, the P surplus peaked in the Eucha-Spavinaw watershed with 1,100 Mg yr^{-1} of excess P or 11 kg ha^{-1} yr^{-1}. For comparison, MacDonald and Bennett (2009) conducted a P accumulation study for the Saint Lawrence River watershed from 1901 to 2001. The maximum P surplus in the 574,000 km^2 watershed was in 1981 with 42,000 Mg yr^{-1} or 0.7 kg ha^{-1} yr^{-1}. The maximum P surplus in the Illinois River and Eucha-Spavinaw watersheds were 16 to 18 times higher compared to the Saint Lawrence River watershed. Due to improved management and conservation practices, P surplus declined after 1981 in the Saint Lawrence River watershed, similar to the decline in the Illinois River and Eucha-Spavinaw watersheds (Figure 2.8). In the Lake Mendota watershed in Wisconsin, the results were more comparable to the Illinois River and Eucha-Spavinaw watersheds with 8.4 kg ha^{-1} yr^{-1} retained in 1995 (Bennett et al., 1999).

2.3.3.3 Phosphorus Mass Balance

The cumulative P retained from 1925 to 2015 in the Illinois River and Eucha-Spavinaw watersheds was 5.5 to 7.5 kg ha^{-1} yr^{-1} and 4.5 to 6.1 kg ha^{-1} yr^{-1}, respectively. This equates to an average P retention of 86% and 81% for the two watersheds, respectively, which is comparable to the 80% P retention and net P import of 10 kg ha^{-1} yr^{-1} in the Lake Okeechobee watershed in Florida (Fluck et al., 1992). The P retention for eleven watersheds in Minnesota ranged from 10 to 89% (Schussler et al., 2007) and 44% for the Lake Mendota watershed in Wisconsin (Bennett et al., 1999). In the Gjern River watershed in Denmark, the P retention was only 17% with a net P import of 4.6 kg

ha^{-1} yr^{-1} (Kronvang et al., 1999). The difference in retention rates between the watersheds was attributed to the import of poultry feed and crop export.

While poultry production in the Illinois River and Eucha-Spavinaw watersheds was high relative to the Lake Mendota, Gjern River and Minnesota watersheds, crop production and export was minimal. All of this stored P was assumed to be in the soil, reservoirs and stream system. Based on current STP levels, an estimated 150,000 to 228,000 Mg of P is stored in the soil in the Illinois River watershed. A larger percent of the surplus P is stored in the soil in the Eucha-Spavinaw watershed, 36,100 to 49,800 Mg. This equates to a soil P accumulation of 3.8 to 5.8 kg ha^{-1} yr^{-1} in the Illinois River watersheds and 3.6 to 5.0 kg ha^{-1} yr^{-1} in the Eucha-Spavinaw watershed. For comparison, the cumulative P accumulation from 1901 to 2001 was 0.5 kg ha^{-1} yr^{-1} in the Saint Lawrence River watershed (MacDonald and Bennett, 2009). MacDonald and Bennett (2009) found that there was a positive trend between cumulative surplus P and STP. Therefore, some of the differences in surplus P in the watersheds can be accounted for in the soils. The average STP in the Saint Lawrence River watershed was 23 to 155 mg kg^{-1} soil compared to 27 to 210 mg kg^{-1} soil in the Illinois River and Eucha-Spavinaw watersheds. Other sites sampled in the US from poultry production areas ranged from 42 to 568 mg kg^{-1} soil (Hood et al., 2001). Both STP and net P accumulation rates were much higher at ten field sites in the United Kingdom. Hood et al. (2001) reported STP levels that ranged from 447 to 2320 mg kg^{-1} and net accumulation rates from 16 to 233 kg P ha^{-1} yr^{-1}. These extremely high rates were attributed to the long-term application of swine, cattle and poultry manure.

The construction of Tenkiller Ferry Lake was completed in 1952. Therefore, P was only retained in the reservoir from 1953 to 2015. Based on the SWAT model predictions (Storm and Mittelstet, 2015) and data from the USGS gage station below the reservoir, the average P retention in the reservoir was 61%. Therefore, the total quantity of P retained in the reservoir ranged from 6,800 to 8,000 Mg, or 2 to 4% of the total P imports. The annual average P concentration in Lake Tenkiller from 1974 to 2007 ranged from 0.049-0.055 mg L^{-1} (Cooke and Welch, 2008). Assuming this concentration for the average volume of the reservoir (0.80 km^3 from 1994 to 2007) (Boyer et al., 2008), 41 to 46 Mg of total P was stored in the water column or less than two percent of the total P stored in the reservoir. This was not surprising considering the elevated average P concentrations of soil cores taken from Lake Tenkiller: 1,100 mg kg^{-1} at a depth of 0-2 cm, and 900 mg kg^{-1} at a depth of 32 cm (Fisher et al., 2009).

The construction of Spavinaw Lake was completed in 1924 and Lake Eucha in 1952. Therefore from 1925 to 1952, P was only retained in Spavinaw Lake (63%). After 1953, most of the P was retained in Lake Eucha (64%), while 18% was retained in Spavinaw Lake. Between 410 and 490 Mg is currently stored in Lake Eucha and 150 to 190 Mg in Spavinaw Lake. Therefore, only 1 to 2% of the stored P is retained in the two reservoirs. The City of Tulsa collected monthly water samples from 2000 to 2012 at four locations on Lake Eucha, with total P concentrations ranging from 0.03 to 0.05 mg L^{-1} and 0.03 to 0.12 mg L^{-1} 0.5 m below the surface and 0.5 m from the bottom of the reservoir, respectively. Based on normal capacity of 0.09 km^3 (USGS, 2014), the quantity of P stored in the Lake Eucha water column ranged from 3 to 12 Mg or approximately two percent of the total P stored in the reservoir. Therefore most of the

stored P is in the benthic sediments. This corroborates research by Haggard et al. (2005) that found the internal P load from the bottom sediment was significant. The P retention rates in Lakes Tenkiller, Spavinaw and Eucha were comparable to several studies summarized by Michael and Benjamin (2008), where the % P retention ranged from 18% to 104% with an average of 54%.

Using Equation 2.4 and completing the mass balance, 3,100 to 118,000 Mg (0.08 to 3.0 kg ha^{-1} yr^{-1}) of P is stored in the stream system in the Illinois River watershed and 0 to 19,200 Mg (0 to 1.9 kg ha^{-1} yr^{-1}) in the Eucha-Spavinaw watershed stream system during 1925 to 2015. Though there was significant uncertainty in the percent of P stored in the soil and stream system, 74 to 89% of P added to the two watersheds is currently retained in the watersheds (Figure 2.9). P retention in 11 watersheds in Minnesota ranged from 10 to 89% with the primary P export being crops (Schussler et al., 2007). Overall, a larger percent of P is stored in the stream system in the Illinois River watershed compared to the Eucha-Spavinaw watershed. This stream system P is stored in the bottom sediment of the streams, on the floodplains and in the streambanks. Recent work by Purvis et al. (2015) supported these findings, where they documented the P content of streambanks was much higher in the Barren Fork Creek, a tributary to the Illinois River, compared to Spavinaw Creek.

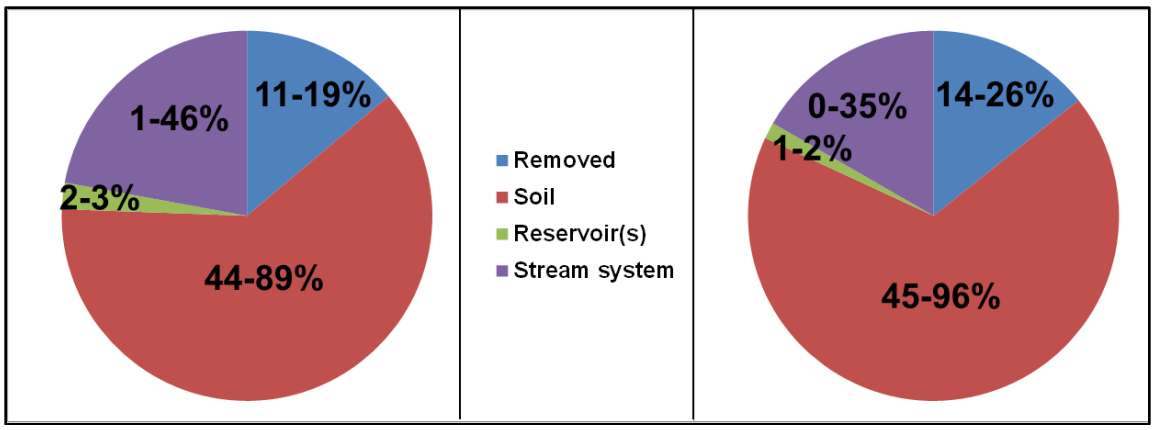

Figure 2.9. Average and range of percent phosphorus removed and stored as legacy phosphorus from the Illinois River (left) and Eucha-Spavinaw (right) watershed during the period 1925 to 2015.

2.4 Conclusions

This research provided a method to quantify legacy P in a watershed using a P mass balance with an uncertainty analysis. Though other watersheds around the world will have different P imports and exports, the methodology and uncertainty analysis applied in this study is easily transferred to other watersheds. Although there is uncertainty with P imports, exports and P retained within the watershed, this method gives watershed managers reasonable estimates to the quantity and location of the legacy P within their watershed.

The method was successfully applied to two P nutrient-rich watersheds, Illinois River and Eucha-Spavinaw. In the two watersheds, the P stored in the soil, stream system and reservoirs from 1925 to 2015 was estimated at 74 to 89% of the total amount of P imported into the watersheds. A small fraction of the P stored in the watersheds was retained in the reservoirs' sediment, and thus most of the P is stored in the soil and stream systems. This is not only an issue in these watersheds, but worldwide. P in both soils and oceans is increasing worldwide (Carpenter et al., 1998; Howarth et al., 1995). It takes decades, if not centuries, for elevated STP to decline to manageable levels. With

this knowledge, policy makers are equipped to make more informed decisions on future management plans for the two watersheds. Although the inclusion of the uncertainty analysis demonstrated the great deal of uncertainty in the various P imports, removals and sinks in the watersheds, it give policy makers and watershed managers more confidence in the estimates. To decrease the uncertainty in P estimates, yet maintain the desired confidence, more time should be spent attaining more detailed data for the most dominant P imports and exports. Providing more certainty in the large P imports and exports will reduce the overall uncertainty of the estimates.

Future research is needed to more accurately quantify where P is stored in the stream sediment, floodplains and streambanks. This information is needed to better refine recommended management practices. With high uncertainty in the P storage in the stream system, soil and field tests need to be conducted to better understand the movement of P through the stream system and long-term sinks. In addition, a study to quantify P from atmospheric deposition is needed for the Ozark Highland Ecoregion since there is significant uncertainty (Newman, 1995) and the contributions vary spatially (Tipping et al., 2014).

CHAPTER III

USING SWAT TO ENHANCE WATERSHED-BASED PLANS TO MEET NUMERIC NUTRIENT WATER-QUALITY STANDARDS

3.1 Abstract

The number of states that have adopted numeric nutrient water-quality standards has increased to 23, up from ten in 1998. To date, watershed-scale hydrological models have not yet been used to evaluate numeric water-quality standards. The objective of this study was to use the Soil and Water Assessment Tool (SWAT) to evaluate both stream and reservoir water-quality standards. Stream and reservoir numeric water-quality standards exist in the Illinois River and Eucha-Spavinaw watersheds, respectively. The two watersheds in northeast Oklahoma and northwest Arkansas have been an area of controversy in recent years. The streams and reservoirs have elevated phosphorus (P) due to a history of intense poultry production, cattle operations, point source discharges and increased urbanization. A SWAT model was created for each watershed, and then calibrated and validated for streamflow and total and dissolved P. An average of 30 and 190 Mg yr^{-1} entered Lakes Eucha and Tenkiller from 2004 to 2013, respectively. The two largest P sources were soil test P and cattle. Due to recent land-management changes in the Eucha-Spavinaw watershed, Oklahoma is meeting the established water-quality standard, 0.0168 mg L^{-1} total P, in Lake Eucha. Although extensive efforts to reduce P

loads have been conducted in the last decade in the Illinois River watershed, a large quantity of P is still reaching the streams and Tenkiller Ferry Lake. The model was used to identify a combination of potential land-management practices required to meet the water-quality standard, 0.037 mg L^{-1} total P, in three of Oklahoma's designated Scenic Rivers: the Illinois River, Barren Fork Creek and Flint Creek. With recent reductions in poultry litter application and improvements in municipal waste water treatment plants, future conservation practices need to focus on cattle production and legacy P, including floodplains and streambanks. This research illustrated how a watershed model can provide critical information for watershed-based plans to address numeric water-quality standards and legacy P.

3.2 Introduction

Excessive nutrients are a major pollutant to many waterbodies worldwide. The United States (US) alone has 7,765 waterbodies impaired due to nutrients with over a third on the States' Clean Water Act 303(d) list as a direct result of total phosphorus (P) (USEPA, 2015a). Major P sources include crop and livestock production, wastewater treatment plants (WWTPs), animal manure and commercial fertilizer (Sims and Sharpley, 2005). Excess P entering streams, lakes and reservoirs can lead to eutrophication, resulting in algal blooms, oxygen depletion and the overall degradation of the water quality (Sims and Sharpley, 2005). Water-quality degradation in streams and reservoirs has led to an increasing number of states to implement numeric water-quality standards, which can provide a remediation goal for state agencies to develop and implement effective watershed-based plans (USEPA, 2015b). The number of states adopting

numeric water-quality standards continues to increase. There are currently 23 states with numeric water-quality standards, compared to only ten in 1998 (USEPA, 2015b).

The Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998), a watershed-scale hydrological model, has been used worldwide to meet several different types of project needs such as the development of Total Maximum Daily Loads (TMDL) (Borah et al., 2006), modelling climate and land use change (Li et al., 2015) and the effects of conservation practices on water quality (Liu et al., 2014). However, to date the model has only been used by Storm et al. (2010) to aid in the development of a watershed-management plan to meet stream numeric water-quality standards. SWAT predictions have been used as input to reservoir models to evaluate if the water quality standard for a reservoir is exceeded (OWRB, 2008), but to date has not been used without a reservoir model.

Oklahoma has stream, Illinois River watershed, and reservoir, Eucha-Spavinaw watershed, numeric water-quality standards. Both watersheds have a long history of poultry and cattle production leading to elevated soil P, and elevated P in WWTP discharges. Major changes have occurred in the Illinois River and Eucha-Spavinaw watersheds during the last decade. The State of Oklahoma sued the City of Fayetteville, Arkansas in 1986 concerning the city's wastewater discharge into the Illinois River. Reaching the US Supreme Court in 1992, the court ruled that a downstream State's water quality standards must be met (Soerens et al., 2003). This led, in part, to improved WWTP facilities at the Arkansas Cities of Fayetteville, Springdale, Rogers, Bentonville and Siloam Springs and the implementation of a total P standard for Oklahoma scenic rivers. The City of Tulsa joined the confrontation in the mid 1990's, provoked by taste

and odor issues in their drinking water, to sue several poultry companies and the City of Decatur, Arkansas in 2001 for damages and injunctive relief in Federal Court (White et al., 2011). The lawsuit was settled in 2003 and lead to improvements in the Decatur, Arkansas WWTP, poultry litter management and export requirements, and Federal Court oversight of the agreement. As part of the settlement agreement, regulations on poultry litter application were implemented and overseen by the Court. Next, the Oklahoma Attorney General filed suit in Federal Court in 2005 against 11 poultry companies operating in the Illinois River watershed. The court ruling is still pending. In the decade since these lawsuits were filed, the quantity of poultry litter exported from the two watersheds has gradually increased to approximately 90% in 2014 (State of Oklahoma, 2014; ANRC, 2014).

Litigation induced changes, along with numerous technical assistance and agricultural cost share programs conducted by state and federal agencies, contributed to the decline in flow-adjusted total P concentrations in the Illinois River near the Oklahoma/Arkansas state line, at Tahlequah, Oklahoma and on the Barren Fork Creek near Eldon, Oklahoma (Scott et al., 2011; Haggard et al., 2010). In spite of the water-quality improvements in the watersheds, it is unknown if these are sufficient to meet Oklahoma water-quality standards. The objective of this research is to use SWAT to estimate P loads originating from the Oklahoma portion of these watersheds and to determine if those loads exceed existing water-quality standards. For this research, loads originating from Arkansas will not be considered. If standards are not being met, new management practices will be evaluated to reduce P loads. A variety of agricultural

management practices and land use changes were simulated to determine the necessary changes required to meet the Oklahoma water quality standards.

3.3 Methods and Materials

3.3.1 Watershed Descriptions

The Illinois River and Eucha-Spavinaw watersheds occupy over 4,100 km^2 and 1,100 km^2, respectively (Figure 3.1). Dominated by forest and pasture, the watersheds are located in the Ozark Highland ecoregion. The area receives on average 1,100 mm of precipitation annually. The headwaters of the Illinois River are in Benton County, Arkansas, and flow 230 km through Washington and Adair Counties before emptying into Tenkiller Ferry Lake (Lake Tenkiller) in Oklahoma. The two main tributaries are Barren Fork Creek and Flint Creek. Spavinaw Creek, also originating in Benton County, flows into eastern Oklahoma to Lake Eucha. Its main tributary is Beaty Creek. The watersheds are characterized by cherty soils and gravel-bed streams. The main agricultural activities are pastured cow/calf operations and poultry production. The Illinois River and, to a lesser degree, Spavinaw Creek, are popular destinations for float trips, fishing and swimming. Lakes Tenkiller and Eucha are also popular recreational areas for fishing, boating and camping.

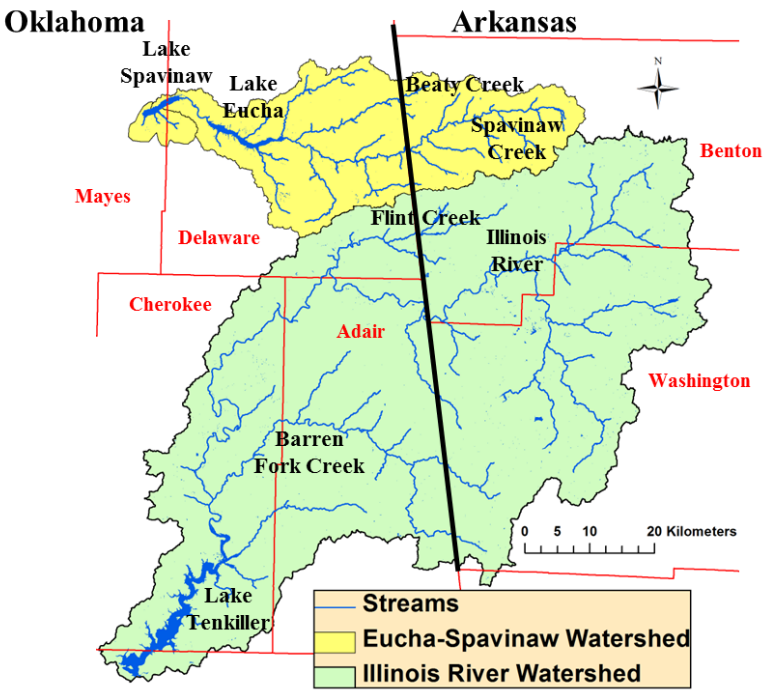

Figure 3.1. Illinois River and Eucha-Spavinaw watersheds in northeast Oklahoma and northwest Arkansas showing counties (red), the Oklahoma/Arkansas state line (black) and the major streams and reservoirs.

3.3.2 SWAT Model Description

SWAT is a basin-scale hydrological/water quality model used to predict streamflow and pollutant losses from watersheds with mixed land covers, soils and slopes. The model was developed to assist water resource-managers assess water quantity and/or quality in large watersheds and as a tool to evaluate the impact on agricultural conservation practices implementation. The SWAT model, a product of over 30 years of model development by the US Department of Agriculture Agricultural Research Service, has been extensively used worldwide (Gassman et al., 2007, 2014). The model is process-based and can simulate several processes such as the hydrological cycle, soil erosion and nutrient transport.

An ArcGIS interface can be used for model input of land cover, soils, elevation, weather, and point sources, and defining the flow network. The interface divides the

watershed into subbasins, which are further split into hydrological response units (HRUs). Each HRU has one soil type, one land use and one slope. SWAT uses the Modified Universal Soil Loss Equation (MUSLE) to calculate sediment yield for each HRU, and then combined with nutrient losses and runoff for each subbasin are routed from reach to reach until arriving at the watershed outlet. Many field-scale activities, such as planting dates, irrigation, fertilization, grazing, harvesting and tillage, are utilized by SWAT as management options scheduled by date. Further details on the theoretical aspects of hydrology, nutrient cycling, crop growth and their linkages are provided in Neitsch et al. (2009).

This research used SWAT 2012 rev. 583 and the recently incorporated simplified in-stream P routine (White et al., 2014), which consists of two components. The first component represents the transformation of soluble P to particulate P (i.e. the uptake of soluble P by algae and P precipitation) and its interactions with sediment, which is based on an equilibrium P concentration (EPC). EPC is the concentration at which there is no net sorption or desorption from benthic sediments into the water column. If the EPC is greater than the concentration of soluble P in the water column, P moves from the streambed to the water column; the reverse occurs if the EPC is less than the soluble P. The second component represents the deposition and scour of particulate P (sediment-bound P and algal P) to/from the benthos. Based on the ratio of flow to bankfull discharge, P is either scoured or deposited.

3.3.3 SWAT Model Setup

Two SWAT models, one for each watershed, were created using the most detailed and accurate data available. Since land cover is one of the most important SWAT inputs

and the latest available data was the 2006 National Land Cover Dataset (NLCD), a more recent land cover using Landsat 4-5 Thematic Mapper imagery was developed using ERDAS IMAGINE 9.3 and ArcGIS 10.0. Four Landsat 4-5 Thematic Mapper images from 2010 and 2011 were analyzed and the Normalized Difference Vegetation Index (NDVI) calculated in ERDAS IMAGINE using the equation:

$$NDVI = \frac{(NIR - VIS)}{(NIR + VIS)} \tag{3.1}$$

where VIS and NIR are the spectral reflectance measurements acquired in the visible (red) and near-infrared regions, respectively. The NDVI aided in the differentiation between vegetative land covers. An unsupervised land cover classification was conducted, and September 2011 ground truth data and 2010 NAIP (National Agricultural Imagery Program) aerial photographs were utilized to validate the 14 assigned land covers: water, wetlands, urban impermeable, urban grass, row crops, bare soil, shrub land, mixed hay, mixed well-managed, mixed overgrazed, warm season hay, warm season well-managed and warm season overgrazed. The August 2011 Landsat 4-5 near infrared image and the following criteria were used to classify the warm and mixed season pastures into well-managed (high biomass) and overgrazed (low biomass):

- Well-maintained pasture: high vegetative biomass and relatively homogeneous in spectral response and apparent color

- Overgrazed pasture: fields with “mottled” appearance due to selective grazing by cattle or high cattle density; varying vegetative biomass with a large soil component to the spectral signature

To simplify the SWAT model, the final land cover data layer was consolidated into nine categories with forest and pasture being the dominant land covers (Table 3.1; Figure 3.2).

Table 3.1. Land cover categories and area percentages used in the Illinois River and Eucha-Spavinaw watershed SWAT models based on the 2010 and 2011 Landsat 4 and 5 Thematic Mapper images.

Land Cover	Watershed Coverage (%)	
	Illinois River	Eucha-Spavinaw
Forest	47.2	48.6
Well-Managed Pasture	19.0	27.0
Overgrazed Pasture	8.3	3.4
Hay	11.9	8.7
Rangeland	3.6	2.6
Row Crops	0.2	1.1
Bare Soil	0.2	0.1
Urban	8.5	2.4
Water	1.3	1.9

Topography was defined by a United States Geological Survey (USGS) 10 m resolution Digital Elevation Model (DEM), which was used to calculate subbasin parameters, e.g. slope and slope length. The SWAT model for the Illinois River and Eucha-Spavinaw watersheds were delineated into 147 subbasins with 4,930 HRUs and 129 subbasins with 3,629 HRUs, respectively. Additional outlets were added at the USGS gage stations and at the Arkansas/Oklahoma state line for all major streams. Soil characteristics were defined using SSURGO (Soil Survey Geographic Database) data. Significant time and effort was invested gathering and creating the weather files, one of the most important parameters for a hydrological model. Nine National Oceanic and Atmospheric Administration (NOAA) weather stations and three Oklahoma Mesonet stations were used in the models (Figure 3.3). Monthly data for 12 WWTPs in the Illinois River watershed and one at Decatur, Arkansas in the Eucha-Spavinaw watershed were added to the SWAT models as point sources (USEPA, 2012) (Figure 3.3). WWTP average annual flow and organic and mineral P from 1990 to 2010 used in the calibration and validation periods are in Table 3.2. The final data layer added to the models was

ponds, which affected the hydrology by impounding water and trapping sediment and nutrients. The pond data layer was obtained from the National Hydrography Dataset (USDA NRCS, 2011), and the drainage area fraction of ponds for each subbasin was calculated and input into SWAT.

Table 3.2. Average annual flow and organic and mineral phosphorus (P) load (1990 to 2010) from the wastewater treatment plants in the Illinois River and Eucha-Spavinaw watersheds used in the SWAT model for the calibration and validation periods (USEPA, 2012).

Point Source	Flow (m^3 d^{-1})	Organic P (kg d^{-1})	Mineral P (kg d^{-1})
Town of Prairie Grove	1,100	0.9	3.7
City of Fayetteville	16,000	1.4	5.7
Town of Lincoln	1,800	0.7	2.8
City of Springdale	41,500	23	98
City of Rogers	20,200	7.7	33
Town of Gentry	1,800	1.4	6.1
City of Siloam Springs	10,300	6.4	27
City of Tahlequah	10,200	1.5	6.5
Town of Westville	700	0.2	0.8
Stilwell Area Development	2,500	0.8	3.5
Town of Decatur	4,900	2.5	10.8

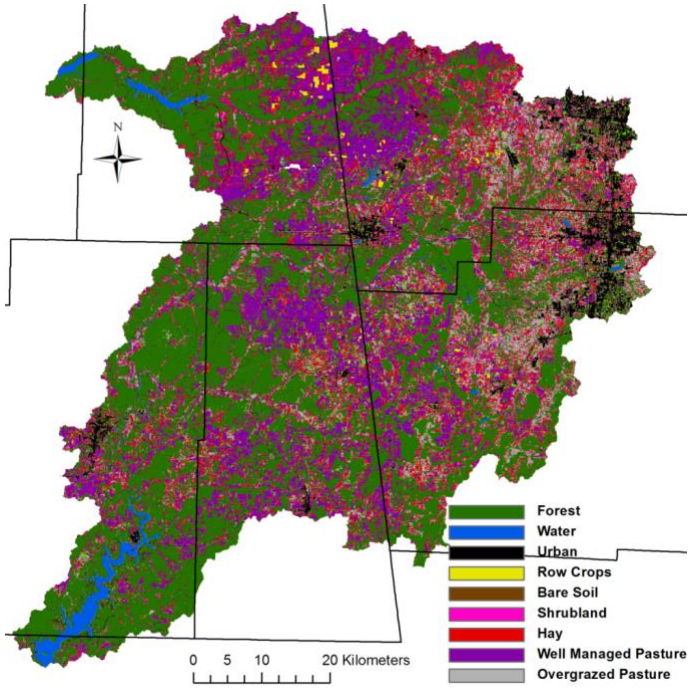

Figure 3.2. Land cover for the Illinois River and Eucha-Spavinaw watershed SWAT models using 2010 and 2011 Landsat 4 and 5 Thematic Mapper images.

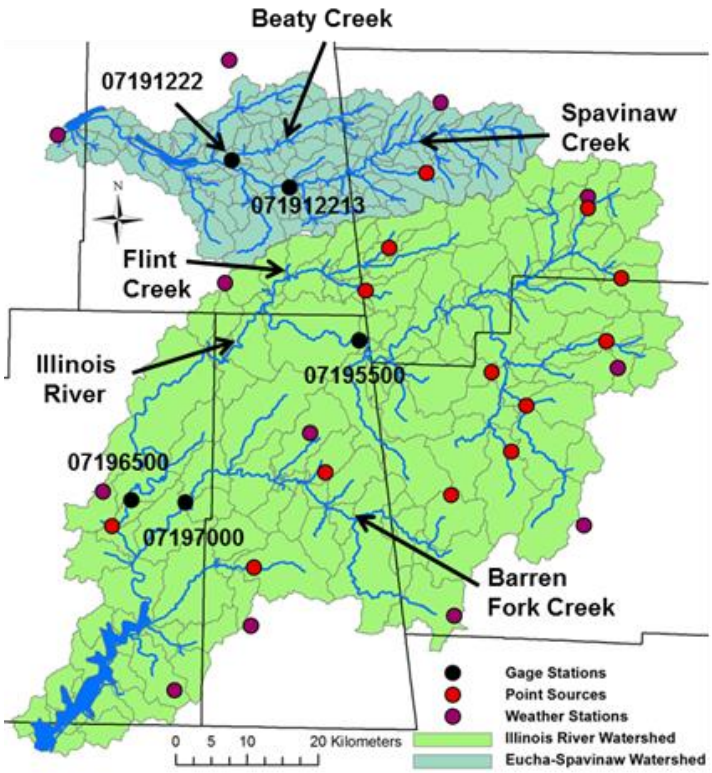

Figure 3.3. United States Geological Survey gage stations, weather stations and point sources used in the Illinois River and Eucha-Spavinaw watersheds SWAT models.

3.3.4 SWAT Land Cover Management

The most important management data for the two watersheds were cattle stocking rates, litter application rates and STP. Cattle density was calculated using the average population from 1987 to 2007 (USDA Census of Agriculture, 2012), which yielded 101,700 and 24,500 heads of cattle for the Illinois River and Eucha-Spavinaw watersheds, respectively. Well-managed pastures were assumed to be grazed at 0.60 animal units ha^{-1} and overgrazed pastures at 1.25 animal units ha^{-1} (Redfearn, D., personal communication, March 12, 2012).

In order to estimate the quantity of litter applied per subbasin, the location of poultry houses, the total quantity of litter applied and the STP were required. The most recent poultry house location data for the Illinois River watershed were 2005, which

included 1,958 active houses (Figure 3.4), 361 abandoned houses, 838 inactive houses, 110 removed houses and 294 unknown status houses (Fisher et al., 2009). Since poultry house data were not available for the Eucha-Spavinaw watershed (ANRC, 2014; ODAFF, 2013), the poultry houses were manually digitized using 2010 NAIP images (Figure 3.4). A total of 908 houses were digitized. The number of active and inactive houses for the Eucha-Spavinaw watershed was also unknown. Therefore, assuming 80,000 birds per house and the ratio of active to inactive houses in the Illinois River watershed (Fisher et al., 2009), 508 poultry houses were considered active.

Based on the average number of broilers, turkeys, hens and pullets in the watersheds and the quantity of waste produced per bird (NMP, 2007), a total of 316,000 and 81,000 tons (dry weight basis) of litter were produced annually in the Illinois River

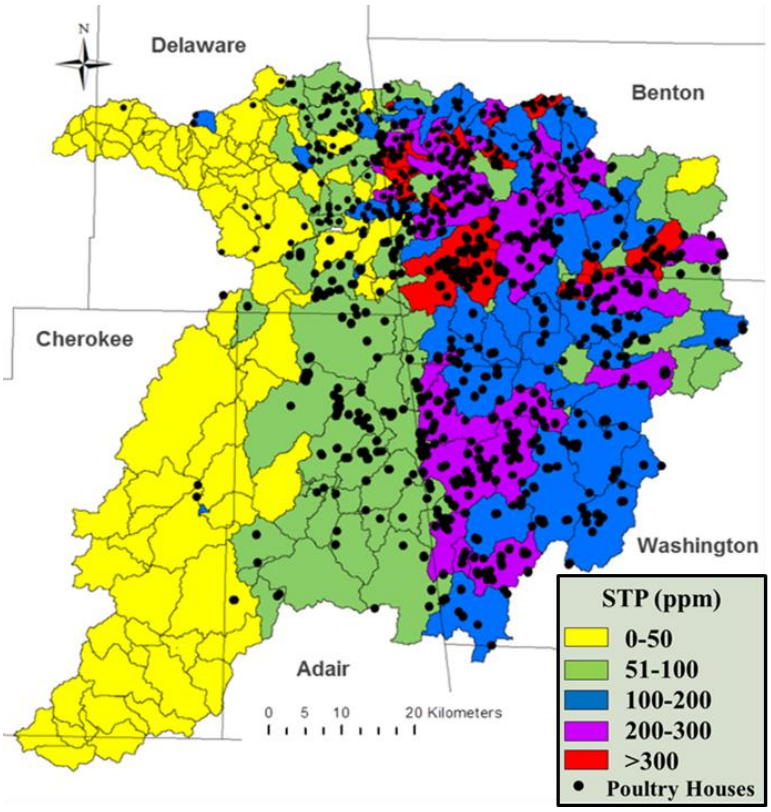

Figure 3.4. Poultry house locations and average Mehlich III soil test phosphorus (STP) for pastures in the Illinois River and Eucha-Spavinaw watersheds based on 2005 data and a 2013 National Agricultural Imagery Program image.

and Eucha-Spavinaw watersheds, respectively, during the calibration and validation periods. Starting in 2003, the export of litter increased substantially over time in the two watersheds based on data obtained from the Oklahoma Poultry Waste Report and Arkansas 2013 Poultry Waste Report (ODAFF, 2013; ANRC, 2014). Using these data, regression equations for litter export vs time for the Illinois River watershed (Equation 3.2) and the Eucha-Spavinaw watershed (Equation 3.3) were developed:

$$Litter\ Export\ (\%) = 0.512473 * X^{2.1214} \tag{3.2}$$

$$Litter\ Export\ (\%) = (2.6107 * X) + 59.352 \tag{3.3}$$

where X is the year (1-10). For the Illinois River watershed calibration period (2001 to 2010), equation 3.4 was integrated and divided by the total number of years (9 years) to estimate the average annual litter export used in the model, given as:

$$Average\ Annual\ Litter\ Export\ (\%) = \frac{\int_{1}^{10} 0.512473 * X^{2.1214} dx}{9} = 24 \tag{3.4}$$

Twenty four percent was subtracted from the total poultry litter produced in each county to obtain an average annual poultry-litter application of 240,000 tons. For the Eucha-Spavinaw watershed calibration period (2002 to 2010), Equation 3.5 was integrated and divided by the total number of years (8 years) to estimate the average annual litter export used in the model, given as:

$$Average\ Annual\ Litter\ Export\ (\%) = \frac{\int_{2}^{10} (2.6107 * X + 59.352) dx}{8} = 75 \tag{3.5}$$

Seventy five percent was subtracted from the total litter produced in each county to obtain an average annual litter application of 20,250 tons.

Reasonable estimates of the total poultry litter applied in the watersheds are given above; however, no data were available on the locations of poultry litter application. Litter was relatively expensive to transport and thus was typically only transported short distances. Therefore, litter was assumed to be applied to pastures in close proximity to the point of production, i.e. within the same subbasin as the poultry houses. Pastures with the highest STP were assumed to receive more poultry litter over time. Oklahoma STP data for pastures were based on county level data from the Oklahoma State University (OSU) Soil, Water, and Forage Analytical Laboratory. Data for Adair, Cherokee, Delaware and Sequoyah Counties were available from October 1994 to December 2008. Arkansas STP data were obtained from the University of Arkansas (UA) Soil Testing Laboratory for Benton and Washington Counties from 1999 to 2011.

The first step to determine poultry-litter application rates by subbasin was to estimate the average pasture and hay meadow STP for each subbasin, which was assumed to have a minimum of 15 mg kg^{-1} (Engel et al., 2013). STP for each subbasin was calculated based on the ratio of poultry house numbers to pasture and hay meadow area. The more poultry houses per ha of pasture and hay meadow in a subbasin, the higher the STP assigned to the subbasin. Based on the county average observed STP for pasture and hay meadow, the subbasin STP values were uniformly adjusted to match the observed county average. This method was validated by comparing observed STP data for 529 samples on the Oklahoma side of the Eucha-Spavinaw watershed (Storm and Mittelstet, 2015).

The quantity of poultry litter applied to each HRU was based on the average STP of the subbasin and the quantity of poultry litter produced in the county. Assuming all of

the litter produced in a county was applied to all pasture and hay meadows within that county, the amount applied to each subbasin was estimated using an area weighted average of the subbasin STP values. As an example, assume a county had two subbasins, subbasin A and B, 2,500 ha of pasture, 2,500 ha of hay meadow, and STP of 100 and 300 mg kg^{-1}, respectively. If the total litter produced in the county were 1,000 tons, then subbasin A would receive 1,000*300/400 = 250 tons of litter or 0.1 t ha^{-1} and subbasin B would receive 1,000*300/400 = 750 tons or 0.3 t ha^{-1}. Since the litter was applied to all pasture and hay meadows, the application rate per ha was lower than reality since some fields do not receive litter each year.

3.3.5 Model Calibration and Validation

Calibration is the process by which parameters are adjusted to make predictions agree with observations. SWAT was designed for use on large un-gaged basins and can be used without calibration; however, calibration generally improves the reliability and reduces the uncertainty of model predictions. Validation is similar to calibration except model parameters are not modified. Validation evaluates the calibrated model with observed data that are not used in the calibration process, and preferably under conditions outside the calibration period. For both the calibration and validation models, a five year warm-up was added to ensure that the model represents reasonable initial conditions at the beginning of each simulation, e.g. aquifer levels, soil water conditions, vegetative growth, etc.

3.3.5.1 Streamflow

Manual calibration was used to calibrate the daily and monthly baseflow, peak flow and total flow at three USGS gage stations in the Illinois River watershed

(07195500, 07196500 and 07197000) and two in the Eucha-Spavinaw watershed (07191222 and 071912213) (Figure 3.3). Streamflow was calibrated and validated from 1990 to 2010 and 1980 to 1989, respectfully, for the Illinois River watershed. For the Eucha-Spavinaw watershed, flow was calibrated from 2005 to 2010 and validated from 1999 to 2004 for gage station 07191222 and 2006 to 2010 and 2002 to 2005 for gage station 071912213. Hydrograph separation program (HYSEP), a product of the USGS, was used to estimate baseflow (Sloto and Crouse, 1996).

A sensitivity analysis was conducted on eleven parameters based on previously used calibration parameters and SWAT documentation (Neitsch et al., 2009). Parameters were adjusted within the SWAT recommended range and the model sensitivity analyzed to determine the influence that each parameter had on peak flow and baseflow.

The model performance was evaluated using the coefficient of determination (R^2), Nash-Sutcliffe Efficiency (NSE) and relative error. R^2 is the square of Pearson's product-moment correlation coefficient and represents the proportion of total variance in the observed data that can be explained by a linear model of observed vs predicted values. NSE is a normalized statistic that determines the relative magnitude of the residual variance compared to the measured data variance (Nash and Sutcliffe, 1970), given as:

$$NSE = \left(1 - \frac{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim})^2}{\sum_{i=1}^{n}(Y_i^{obs} - Y^{mean})^2}\right) * 100\% \tag{3.6}$$

where Y_i^{obs} is the ith observation for the constituent being evaluated, Y_i^{sim} is the ith simulated value for the constituent being evaluated, Y_i^{mean} is the mean of the observed data for the constituent being evaluated and n is the total number of observations. Model

performance ratings for NSE for total monthly flow were the following: Very good >0.75, Good 0.65-0.75, Satisfactory 0.50-0.65, Unsatisfactory <0.50 (Moriasi et al., 2007).

3.3.5.2 Phosphorus

The SWAT in-stream P model was calibrated and validated on daily and monthly time steps from 2001 to 2010 and 1995 to 2000, respectfully, at the three USGS gage stations in the Illinois River watershed. Due to limited P data, the two USGS gage stations in the Eucha-Spavinaw watershed were calibrated from 2002 to 2010 with no validation. The five gages were calibrated for total P, dissolved P and particulate P. Similar to flow, the model was calibrated using the R^2, NSE and relative error simultaneously until the best fit was obtained.

Unlike flow data, daily observed P data were not available. Therefore Load Estimator (LOADEST) was used to estimate daily loads (Runkel et al., 2004). The latest version estimates loads for a user defined time period from discrete water quality samples and measured daily flow using a formulated regression model (Runkel et al., 2004). Goodness-of-fit for LOADEST predictions at the five USGS gage stations are presented in Table 3.3.

Table 3.3. Goodness-of-fit statistics for LOADEST total and dissolved phosphorus load estimates for the Illinois River and Eucha-Spavinaw watersheds at five United States Geological Survey gage stations. NSE is Nash-Sutcliff efficiency.

Watershed	Gage Station	Period of Record	Goodness of Fit	Total Phosphorus	Dissolved Phosphorus
Illinois River	07195500	1990-2010	R^2	0.45	0.38
			NSE	0.43	0.37
	07196500	1990-2010	R^2	0.71	0.68
			NSE	0.69	0.65
	07197000	1990-2010	R^2	0.62	0.63
			NSE	0.58	0.60

Eucha-Spavinaw	071912213	2002-2010	R^2 NSE	0.55 0.59	0.55 0.59
	07191222	2002-2010	R^2 NSE	0.46 0.42	0.45 0.31

3.3.6 Updated SWAT Model

After calibrating the two SWAT models, they were updated to represent current conditions. The major updates included poultry litter application rates (Table 3.4) (ODAFF, 2014; ANRC, 2014), point source discharges and weather. Although average discharge from the five largest point sources in the Illinois River watershed increased over time (Table 3.5), the upgraded WWTPs resulted in average total P load and concentration reductions of 88% and 90%, respectively. The average total P concentration at Decatur, Arkansas also decreased from 1.8 to 0.22 mg L^{-1} in 2002 to 2013, respectively.

Table 3.4. Poultry litter applied (tons yr^{-1}) to pasture and hay meadows within each county in the Illinois River and Eucha-Spavinaw watersheds for the updated (2013) SWAT model (ODAFF, 2014; ANRC, 2014).

Watershed	County					Total
	Cherokee	Adair	Delaware	Benton	Washington	
Illinois River	1,400	4,100	2,250	23,400	30,600	59,750
Eucha-Spavinaw	N/A	N/A	5,400	6,700	N/A	12,100

Table 3.5. Updated (2013) and historical discharge (1990-2010), total phosphorus (P) loads and concentration for the five largest point sources in the Illinois River watershed (USEPA, 2012).

Facility	Time Period					
	2013			1990-2010		
	Discharge (m^3)	Total P (kg d^{-1})	Total P (mg l^{-1})	Discharge (m^3)	Total P (kg d^{-1})	Total P (mg l^{-1})
Fayetteville	24,000	2.7	0.11	16,000	7.0	0.44
Rogers	25,200	5.8	0.23	20,200	40.4	2.0
Springdale	50,600	11.9	0.25	41,500	121.4	3.9
Siloam Springs	9,500	3.8	0.40	10,300	33.2	3.2
Tahlequah	8,900	1.0	0.11	10,200	8.0	0.80
Total	118,200	25.2	Mean: 0.22	108,000	210	Mean: 2.1

Weather stations in the updated SWAT model added data from 2011 to 2013. Since the weather data from 2004-2013 represented drought and wet years, all SWAT scenario model runs were simulated with weather data from 2004-2013. The average rainfall over this period was 1,150 mm yr^{-1} at the Jay, Oklahoma Mesonet station with a range from 848 to 1,720 mm yr^{-1}. The purpose of using weather from this time period was not to predict loads from 2004 to 2013, but to simulate loads using typical representative precipitation.

3.3.7 Meeting Water Quality Standards

The Oklahoma Conservation Commission and the Oklahoma Department of Environmental Quality were interested in determining if the State of Oklahoma was meeting water-quality standards, and if not, what agricultural conservation practices could be adopted in order to do so. The updated SWAT model was used to estimate current P concentrations in the three subwatersheds: Barren Fork Creek, Flint Creek and the Illinois River. The model was also used to estimate P loads entering Lake Eucha and P loads and flow entering Oklahoma from Arkansas.

The total P criterion for the Oklahoma Scenic Rivers, which included the Illinois River, Barren Fork Creek and Flint Creek, is 0.037 mg L^{-1} (OKLA. ADMIN. CODE § 785:46-15-10(h)). To meet the P standard, fewer than 25% of the daily geometric means, calculated using the present month and two previous months, must not exceed 0.037 mg L^{-1} (OKLA. ADMIN. CODE § 785:46-15-10(h), 2014). These concentrations and loads were used to determine if Oklahoma was meeting the water-quality standards. When calculating the 90-day geometric means for total P, the daily total flow and P loads entering Oklahoma from Arkansas were removed from the three subwatersheds. This

allowed the evaluation of P loads from only the Oklahoma portion of the watershed. The total P loads, geometric means and percent exceedances were then calculated for each subwatershed separately.

The Oklahoma water quality standard for Lake Eucha requires the long-term average total P concentration to not exceed 0.0168 mg L^{-1} at 0.5 m below the water surface (OKLA. ADMIN. CODE § 785:45-5-10(8), 2014). The City of Tulsa maintains four water quality sampling sites on the reservoir (Figure 3.5). Each site was sampled monthly 0.5 m below the surface and 0.5 m from the bottom of the reservoir bed. From 2004 to 2013, a combined 1,227 samples were taken at the four sites. The concentration in the reservoir was highest at sampling site EUC03 near the Spavinaw Creek confluence with the reservoir, 0.046 mg L^{-1}, and lowest at sampling site EUC01 near the dam, 0.032 mg L^{-1}, after P settled and was diluted by the forest dominated tributaries around the reservoir.

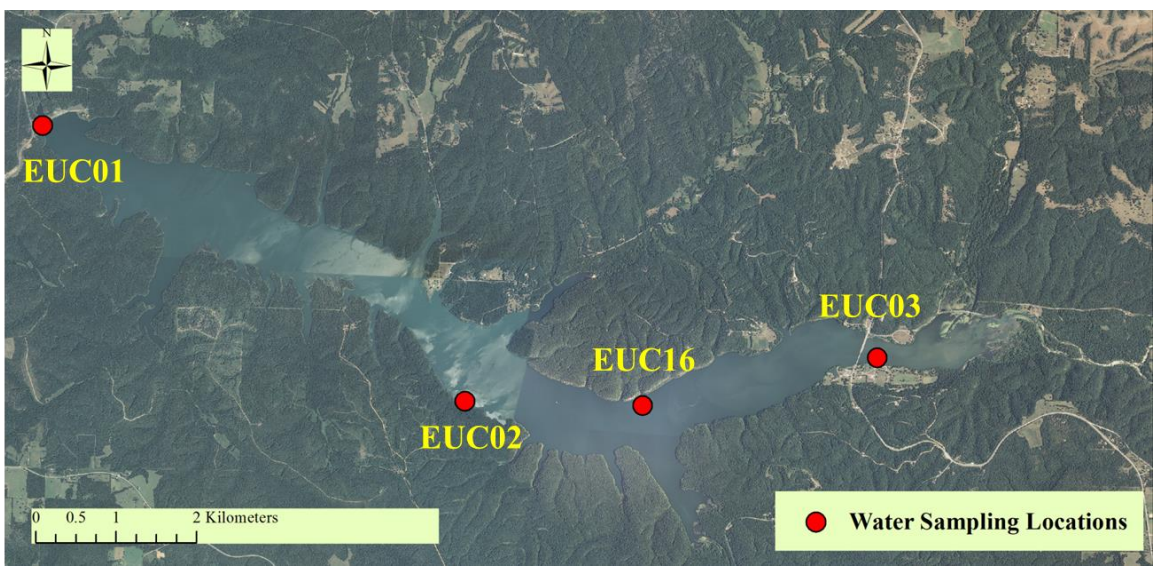

Figure 3.5. Lake Eucha and the four City of Tulsa water sampling locations.

To determine if Oklahoma was meeting the water quality standard in Lake Eucha, a method was needed to predict the P contribution from Oklahoma. There are three P load contributions to the reservoir: Oklahoma sources, Arkansas sources and internal loadings. First, the P load entering from Arkansas was subtracted from the P load entering Lake Eucha to obtain the P loads originating from Oklahoma. Second, equation 3.7 was used to determine the P contribution to the reservoir's water column originating from each of the three P sources:

$$TP_{res} = P_{OK} + P_{AR} + P_{IN} \tag{3.7}$$

where TP_{res} is the total P loading to Lake Eucha (Mg), P_{OK} is the P load contribution from Oklahoma (Mg), P_{AR} is the load contribution from Arkansas (Mg) and P_{IN} is the internal P loading from the reservoir (Mg). Third, equation 3.8 was applied to determine the P concentration in the reservoir if only P originating from Oklahoma was considered:

$$OK_{con} = \frac{P_{OK}}{TP_{res}} * Tot_{con} \tag{3.8}$$

where OK_{con} is the concentration in Lake Eucha if only Oklahoma's contribution was considered (mg L^{-1}), P_{OK} is the P load originating from Oklahoma (Mg), TP_{res} is the total P loading to Lake Eucha (Mg) and TP_{con} is the average P concentration at sampling site EUC03 (mg L^{-1}). Finally, OK_{con} was compared to the water-quality standard to determine if the standard was being met. We chose to use sampling site EUC03 since it had the highest P concentration in the reservoir. Therefore, if the standard was met at the inlet, it would be met throughout the reservoir.

If Oklahoma was not meeting the water quality standards, the following management practices were simulated to analyze the reduction in P concentration and loads for each watershed: 100% litter export, no overgrazing, hay converted to forest,

pasture converted to hay, pasture converted to forest, stocking rate reduced to 0.60 AU ha^{-1}, STP reduced to 33 mg kg^{-1} and crops converted to forest.

3.4 Results and Discussion

3.4.1 Flow Calibration and Validation

During the calibration process, parameters were manually adjusted to obtain the best goodness-of-fit statistics for each gage station. Ultimately seven parameters were modified in the final calibration (Table 3.6). The total flow calibration results were all 'very good' with monthly NSEs all greater than 0.75 (Table 3.7). For the validation period, three of the five sites had NSEs greater than 0.75. The remaining two had NSEs of 0.69 and 0.65. Time series for the two gage stations furthest downstream in each watershed are given in Figure 3.6. Daily calibration and validation NSEs ranged from 0.45 to 0.59, except for gage 07196500. Gage 07196500 was downstream of all 14 point sources and therefore received the cumulative error of the point source inputs, which were reported on a monthly basis. During summer months, the point source flow can be a large percent of the total flow. The average daily flow from 1990 to 2010 at gage 07196500 was 30 $m^3 s^{-1}$ compared to the average daily point source discharge of 1.5 $m^3 s^{-1}$. Seven and 32 percent of the daily flows from 1990 to 2010 were less than 5 $m^3 s^{-1}$ and 10 $m^3 s^{-1}$, respectively.

Table 3.6. Original and calibrated parameter values used to calibrate the SWAT model for the Illinois River and Eucha-Spavinaw watersheds.

Parameter	Original Value or Range	Calibrated Value or Range		Description
		Illinois River Watershed	Eucha-Spavinaw Watershed	
ESCO	0.95	1.0	0.70	Soil evaporation compensation coefficient
RCHRG_DP	0.05	0.33-0.75	0.20-0.38	Aquifer percolation coefficient
ALPHA_BF	0.048	0.30-0.36	0.35	Baseflow Alpha Factor (days)
SOL_AWC	0.08- 0.23	+0.10	+0.10	Soil available water capacity
CN2	39-94	-4 to +2	-6 to -2	SCS curve number adjustment
CH_K2	0	20-35	18-34	Effective hydraulic conductivity in main channel alluvium (mm hr^{-1})
CH_K1	0.5	40-150	75-150	Effective hydraulic conductivity in tributary channel alluvium (mm hr^{-1})

Table 3.7. SWAT model daily and monthly flow calibration and validation results for the Illinois River and Eucha-Spavinaw watersheds on a daily and monthly time scale.

Time	Gage Station	Coefficient of Determination (R^2)	Nash-Sutcliffe Efficiency	Coefficient of Determination (R^2)	Nash-Sutcliffe Efficiency
Illinois River Watershed		**Calibration**		**Validation**	
Daily	07196500	0.32	0.20	0.39	0.21
	07195500	0.57	0.50	0.65	0.57
	07197000	0.50	0.49	0.49	0.45
Monthly	07196500	0.82	0.81	0.80	0.79
	07195500	0.80	0.80	0.79	0.79
	07197000	0.80	0.77	0.66	0.65
Eucha-Spavinaw Watershed		**Calibration**		**Validation**	
Daily	071912213	0.64	0.53	0.64	0.59
	07191222	0.59	0.59	0.47	0.47
Monthly	071912213	0.85	0.85	0.84	0.69
	07191222	0.81	0.80	0.80	0.75

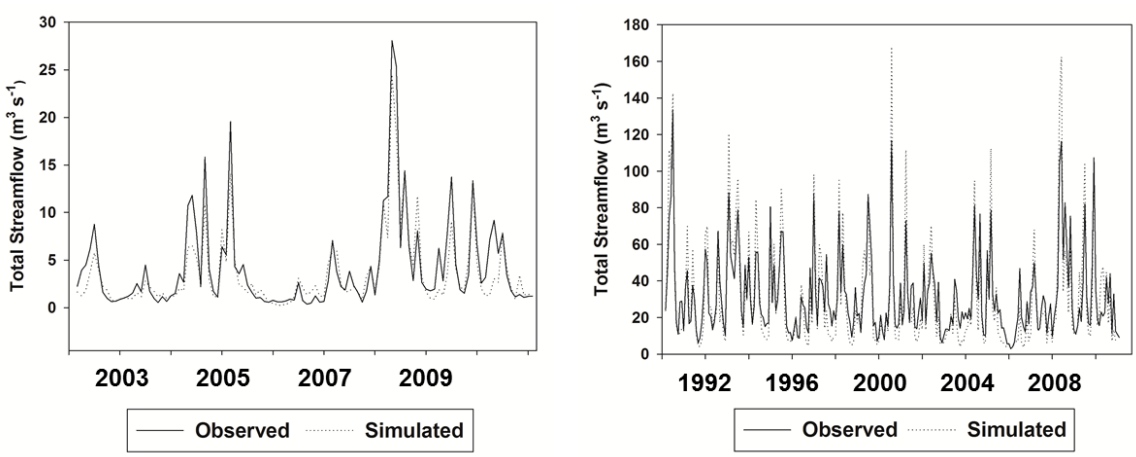

Figure 3.6. Total streamflow calibration results for monthly SWAT simulations at the United States Geological Survey gage stations 071912213 (left) and 07196500 (right).

3.4.2 Phosphorus Calibration and Validation

For the Illinois River watershed, the monthly total P NSE was 0.80, 0.75 and 0.51 for the calibration period and 0.66, 0.81 and 0.65 for the validation periods for gage stations 07196500, 07195500 and 07197000, respectively. The Eucha-Spavinaw SWAT model produced similar results. The total P NSE for the calibration period was 0.67 and 0.73 for gage stations 071912213 and 07191222, respectfully. The relative errors for total P were relatively small, as illustrated in Figure 3.7.

The lower NSEs for gage station 07197000 on the Barren Fork Creek near Eldon, Oklahoma was due to under predicting peak flows during large storm events, which was believed to result from not accounting for streambank erosion. For example, the average observed and predicted flow for October 2009 was 37.2 and 35.9 $m^3 s^{-1}$, respectively, yet the observed and predicted P were 92,800 and 17,700 kg, respectfully. Heeren et al. (2012) observed a large amount of streambank erosion during the October 2009 storm event and Miller et al. (2014) found that P derived from streambank erosion in the Barren Fork Creek watershed was significant; thus, supporting the hypothesis.

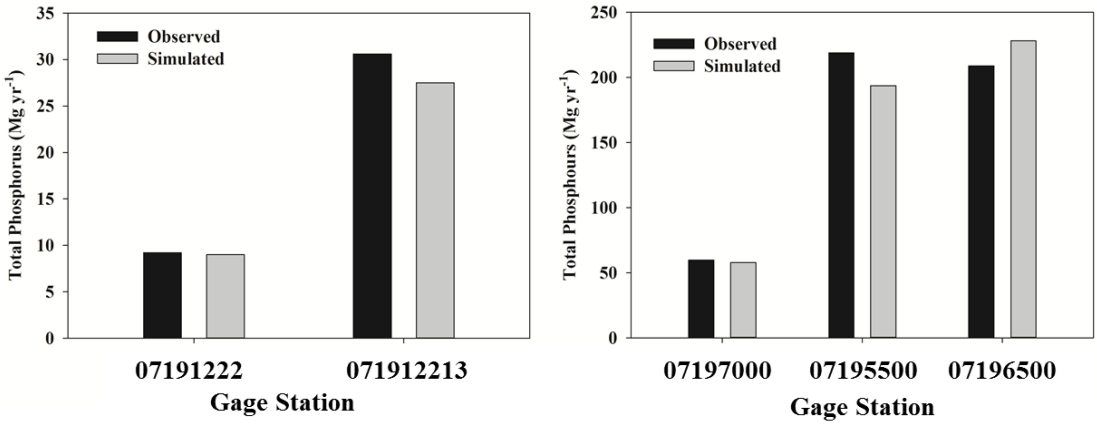

Figure 3.7. Calibrated SWAT observed vs simulated total phosphorus (P) loads for the Eucha-Spavinaw (left) and Illinois River (right) United States Geological Survey gage stations.

3.4.3 Current Phosphorus Sources

An average total P of 190 Mg yr^{-1} entered Lake Tenkiller for the period 2004 to 2013, the watershed outlet for the Illinois River watershed. Based on SWAT predictions, the major P sources reaching Lake Tenkiller were pasture, hay meadow and STP (Figure 3.8), which contributed 65% of all P reaching the reservoir. The major sources in the watershed have changed in the last 10 to 20 years. Storm et al. (2010) found that these sources contributed only 35% from 1990 to 2006; with the major sources being point sources (40%) and poultry litter (13%). Due to improvements to the WWTPs and the increase in poultry litter export, the poultry litter and point sources currently only contribute nine and seven percent, respectively. The increase in the fraction of P from pasture and STP was not due to an increase in their P loads, but rather an overall reduction in P from other sources. P concentrations and loads have been declining in recent years due to the significant changes in the watershed and the implementation of conservation practices (Scott et al., 2011; Haggard et al., 2010).

The total P reaching Lake Eucha was 30 Mg yr^{-1} for the period 2004 to 2013. The major sources of P were also well-managed pasture, hay meadow and STP (69%) (Figure 3.8). In previous SWAT modeling efforts (Storm et al., 2001), STP, point sources and poultry litter were the main P sources. While STP is still a major source, point sources and poultry litter are not due to improvements at the Decatur, Arkansas WWTP and the increase in poultry litter export.

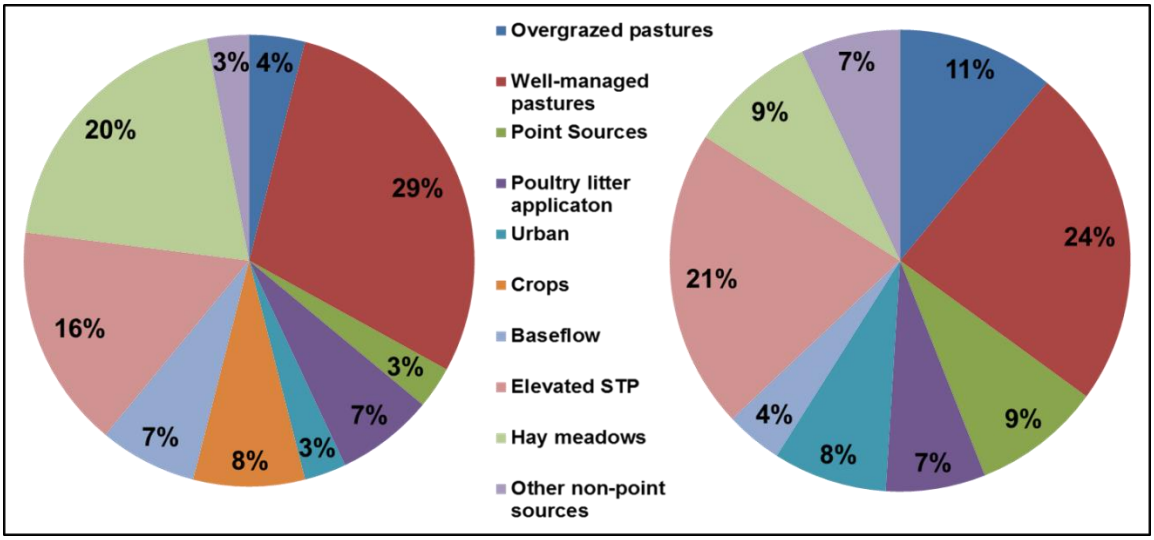

Figure 3.8. SWAT predicted major phosphorus sources reaching Lakes Eucha (left) and Tenkiller (right), at the watershed outlets for the Eucha-Spavinaw and Illinois River and watersheds for the time period 2004 to 2013.

3.4.4 Meeting Water Quality Standards

In order to determine if Oklahoma was meeting the water quality standards, all flow and P entering Oklahoma from Arkansas were removed (Figures 3.9). Only the flow and P entering the waterbodies originating from Oklahoma were considered. The 90-day geometric mean and the total number of days exceeding the Oklahoma Scenic River P criterion 0.037 mg l^{-1} were calculated for each of the three subwatersheds in the Illinois River watershed using daily SWAT predictions. Fewer than 25% of the daily flows can exceed 0.037 mg L^{-1} for the standard to be met (OKLA. ADMIN. CODE § 785:46-15-10(h)). At current conditions, only the Illinois River above the point source discharge (Figure 3.9) met the standard with only 12% exceedances. The percent exceedances for the Illinois River below the point source discharge was 33%, Flint Creek was 37%; and the Barren Fork Creek 44%.

Based on the updated SWAT model, the total P load entering Lake Eucha from external sources was 30 Mg yr^{-1}, 78% from Arkansas and 22% from Oklahoma. The

internal P loading to the reservoir was 12 Mg yr^{-1} (Haggard et al., 2005). Therefore, the average total P loading to Lake Eucha was 42 Mg yr^{-1}. Applying Equation 3.8, the average percent contribution and P loading originating from Oklahoma and Arkansas was 6.6 Mg yr^{-1} or 15.7% and 23.4 Mg yr^{-1} or 55.7%, respectively. Neglecting internal P loads and those originating in Arkansas, the P concentration in Lake Eucha was 0.008 mg L^{-1}. This concentration is less than 0.0168 mg L^{-1} and therefore Oklahoma meets the water-quality standard. The average concentration considering only P loads from Arkansas or internal loads was 0.021 and 0.014 mg L^{-1}, respectively; therefore Arkansas is not meeting the water quality standard, but internal reservoir loadings do.

Based on these results, the standard is being met for the Illinois River above the point source and for the Eucha-Spavinaw watershed. The reason these two watersheds meet the standard, while Flint Creek and the Barren Fork Creek do not, is due to a combination of land use (Figure 3.9), STP and poultry house density. Note that these refer only to the Oklahoma portion of the watershed. Both the Illinois River and Eucha-Spavinaw watershed have a larger percent of forest and less pasture than Flint Creek and the Barren Fork Creek watersheds (Figure 3.10). The Illinois River and Eucha-Spavinaw watersheds both have lower STP, 40 and 47 mg kg^{-1}, respectively compared to 52 mg $kg^{-1}$for Flint Creek and 65 mg $kg^{-1}$for the Barren Fork Creek. While the poultry house density in the Illinois River watershed is only 0.0012 poultry houses per ha pasture, the density in the Eucha-Spavinaw watershed is 0.0095 houses per hectare. This compares to 0.0092 and 0.0050 houses per ha pasture in the Flint Creek and Barren Fork Creek watersheds, respectively.

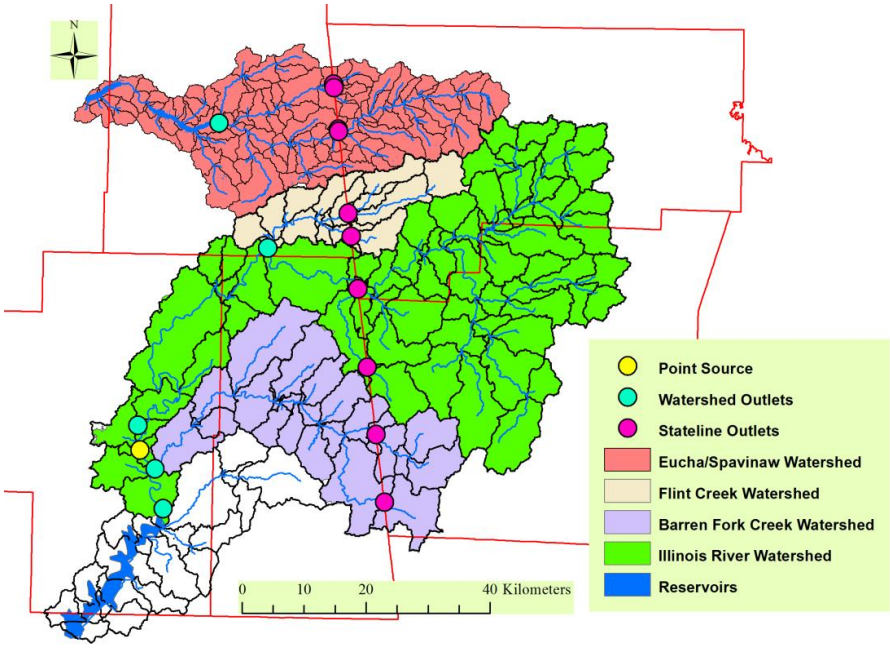

Figure 3.9. Eucha-Spavinaw, Flint Creek, Illinois River and Barren Fork Creek watersheds and the reach outlets on the Arkansas/Oklahoma state line and at the subwatershed outlets, including above the point source on the Illinois River.

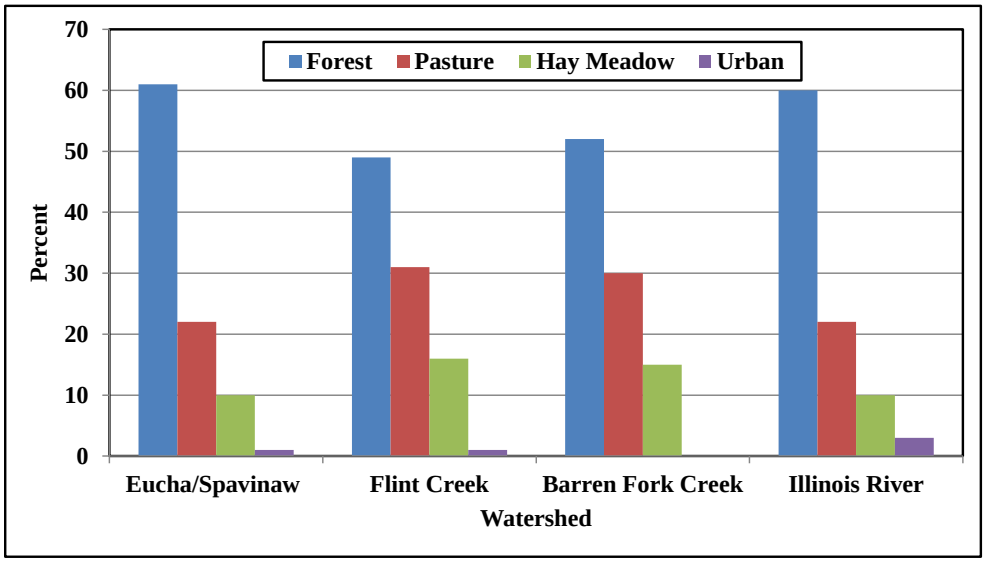

Figure 3.10. Percent forest, pasture, hay meadow and urban for the Eucha-Spavinaw watershed and the Flint Creek, Barren Fork Creek and Illinois River subwatersheds.

Since the water-quality standard is not being met in the three Illinois River subwatersheds, a sensitivity analysis was conducted to determine the percent reduction in standard violations based on several scenarios (Figure 3.11). While most of the scenarios were impractical to implement, the predicted load reductions provide insight into

influences that each of the management practices and land use changes have on P concentrations. The land use conversions that produced the largest reductions were the conversion of pastures and hay meadow to forest. Reducing STP to 32.5 mg kg^{-1} reduced the percent violation by nearly 40% in the Barren Fork Creek subwatershed while reducing cattle density reduced the percent violations by approximately 20% in the Illinois River and Flint Creek subwatersheds, respectively.

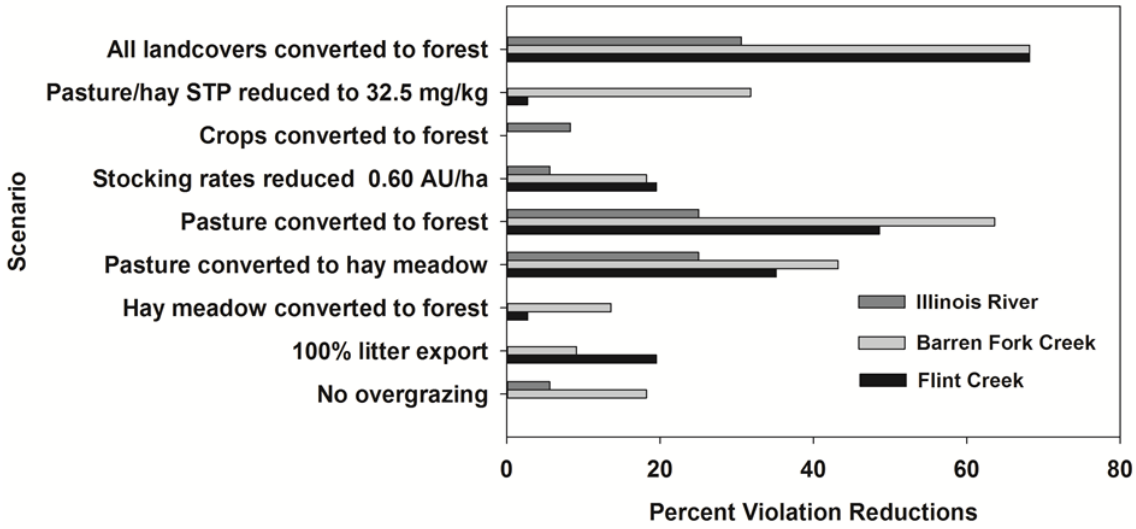

Figure 3.11. SWAT predicted percent reduction in water quality violations for several alternative scenarios for the three subwatersheds in the Illinois River watershed. STP is soil test phosphorus; AU is animal units.

Guided by the results of the sensitivity analysis, a mixture of the scenarios was simulated for each subwatershed to determine the type and number of management and land use changes required to meet the water quality standard in the Barren Fork Creek and Flint Creek subwatersheds and the Illinois River subwatershed below the point source. The following management changes were simulated individually and in various combinations:

1) 100% litter export
2) Hay meadow converted to forest
3) Pasture converted to hay meadow
4) Pasture converted to forest

5) Pastures with slope > 3% converted to forest
6) Stocking rate reduced to 0.60 AU ha^{-1}
7) STP reduced to 32.5 mg kg^{-1}
8) Crops converted to forest
9) No overgrazing
10) No urban P fertilizer
11) No litter application on fields with slopes greater than 3%
12) Conversion of all pasture and hay meadow with slopes greater than 10% to forest
13) Hay meadow with slope > 3% converted to forest

Figure 3.12 shows the percent standard exceedances after the incorporation of several management and land use changes. The standard required fewer management changes to meet in the Illinois River subwatershed since forest made up 60% of the watershed compared to 49% and 52% for the Flint Creek and Barren Fork Creek watersheds, respectively. The standard was most challenging to meet in the Flint Creek watershed since pasture makes up over 30% of the watershed and it has a higher poultry house density.

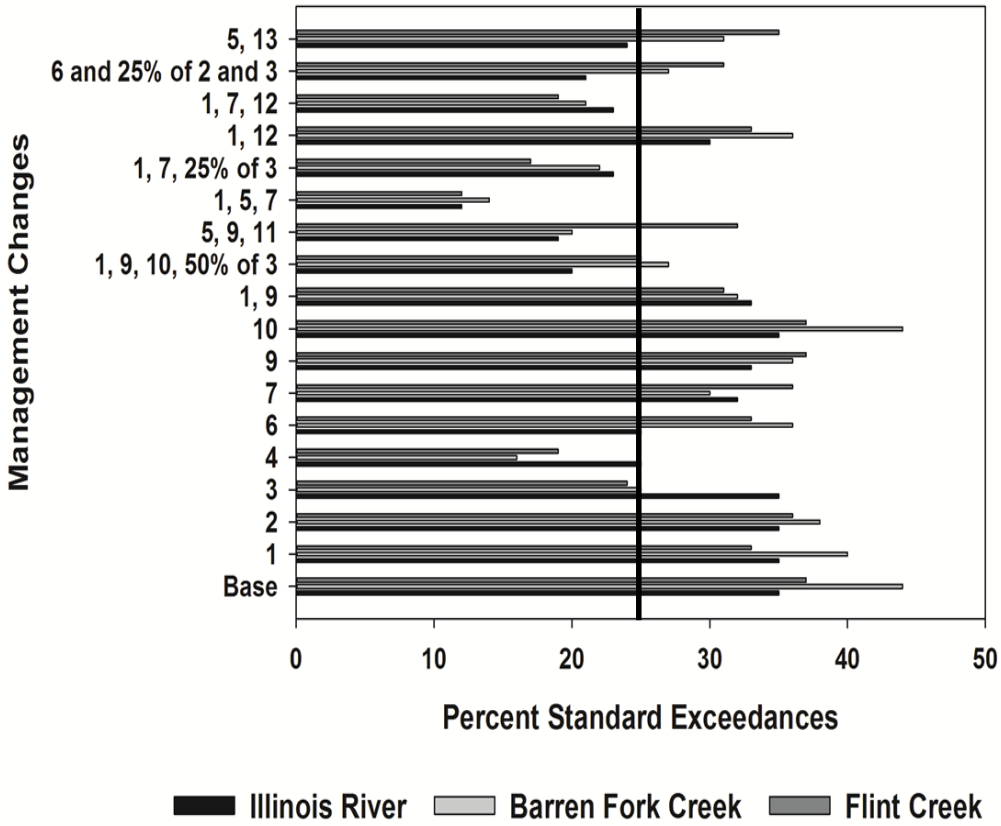

Figure 3.12. Individual and a combination of management changes and the resulting percent standard exceedances for the three subwatersheds in the Illinois River watershed. The black line represents the standard (25% exceedances). The following scenarios were simulated: 1) 100% litter export; 2) Hay converted to forest; (3) Pasture converted to hay; (4) Pasture converted to forest; (5) Pastures with slope > 3% converted to forest; (6) Stocking rate reduced to 0.25AU ha^{-1}; (7) STP reduced to 32.5 mg kg^{-1}; (8) Crops converted to forest; (9) No overgrazing; (10) No urban P fertilizer; (11) No litter application on fields with slopes greater than 3%; (12) Convert all pasture and hay with slopes greater than 10% to forest; (13) Hay with slope > 3% converted to forest.

3.5 Conclusions

The new in-stream P routine for the SWAT model was calibrated and validated at five gage stations in the Illinois River and Eucha-Spavinaw watersheds. The Nash-Sutcliffe Efficiencies for total P ranged from 0.51 to 0.80. Poultry-house density and county-level STP were used to characterize subbasin litter-application rates and STP. Although there is not a reservoir component to SWAT, a method was developed to determine if Oklahoma was meeting the water-quality standard in Lake Eucha. Due to

recent land management changes in the two watersheds, Oklahoma is meeting the water quality standard in Lake Eucha. However, more changes will be required in the Illinois River watershed for the three designated Scenic Rivers to meet the 0.037 mg L^{-1} water-quality standard. With the recent reduction in litter application rates and improvements in wastewater treatment plants, meeting the water-quality standard will require more focus on cattle and legacy P. Currently, STP is responsible for a significant quantity of P reaching the reservoirs. This will continue to be a major source of P for many years and future management practices will need to either stabilize or remove this P.

As the number of waterbodies in the US with numeric nutrient water-quality standards continues to increase, watershed models, such as SWAT, will be an invaluable tool to aid in the development of watershed-based plans. The SWAT model proved to be a capable tool in the evaluation of various management changes and conservation practices required to meet numeric water-quality standards in the watersheds studied. The SWAT model can aid watershed managers in identifying select fields where P load reductions will be maximized, such as overgrazed pastures with elevated STP and slopes. This research demonstrated the effectiveness of a watershed-scale model in providing critical information in the development of watershed-based plans for streams and reservoirs with numeric water-quality standards.

CHAPTER IV

USING SWAT TO PREDICT WATERSHED-SCALE STREAMBANK EROSION ON COMPOSITE STREAMBANKS

4.1 Abstract

Streambanks can be a significant source of sediment and P to aquatic ecosystems. Although the streambank-erosion routine in the Soil and Water Assessment Tool (SWAT) has improved in recent years, the lack of site or watershed-specific streambank data increases the uncertainty in SWAT predictions. There were two primary objectives of this research: (1) improve and apply the current streambank-erosion routine in SWAT on composite streambanks and (2) compare SWAT-default channel parameters to field-measured values and assess their influence on erosion. Three modifications were made to the current streambank-erosion routine: replaced the empirical applied-shear stress equation with a process-based equation, replaced bankfull width and depth with top width and bank height and incorporated an area-adjustment factor to account for heterogeneous trapezoidal cross-sections. The updated streambank-erosion routine was tested on the gravel-dominated streambanks of the Barren Fork Creek in northeastern Oklahoma. The study used data from 28 cross-sectional surveys, including bank height and width, bank

slope, bank-gravel d_{50} and bank composition. Gravel d_{50} and k_d - τ_c relationships were used to estimate the critical shear stress (τ_c) and the erodibility coefficient (k_d), respectively. Incorporating the process-based shear stress equation increased erosion by 85%, the area-adjustment factor increased erosion by 31% and the erosion decreased 30% when using top width and bank height. Incorporating the process-based applied shear stress equation, sinuosity, radius of curvature and measured bed slope improved the predicted vs observed Nash-Sutcliffe Efficiency and R^2 at the ten study sites from -0.33 to 0.02 and 0.49 to 0.65, respectively. Although the process-based applied shear stress equation was the most influential modification, incorporating the top width, bank height and area-adjustment factor more accurately represented the measured irregular cross-sections and improved the model predictions compared to observed data.

4.2 Introduction

Sediment is a primary pollutant to surface waters and the fifth leading cause of water quality impairment in the US (USEPA, 2015). Though erosion is a natural process, the rate of erosion has been accelerated due to anthropogenic activities, such as farming and urbanization. Although sediment loss from agricultural fields, deforestation, and construction sites is significant, in some watersheds streambank erosion can be the most significant contributor of sediment to rivers and streams (Simon and Darby, 1999; Simon et al., 2002; Wilson et al., 2008). Streambank erosion has been observed to increase 10 to 15 times with the advent of European settlement. Rates cited range from 37% to up to 92% (Walling et al., 1999; Simon, et al., 1996). Excess sediment in our streams and reservoirs affects water chemistry, water clarity, increases the cost of treating drinking water, harms fish gills and eggs, reduces benthic macroinvertebrates densities and

diversities and increases turbidity. Increased turbidity not only affects the water aesthetics, but reduces photosynthesis and organisms' visibility. Siltation alters flow in streams and decreases the storage area in our reservoirs, which in turn affects flooding, drinking water and recreation.

Although streambank erosion can contribute a significant quantity of sediment and phosphorus to stream systems (Miller et al., 2014; Kronvang et al., 2012), most watershed-scale models are limited in their ability to predict streambank erosion (Merritt et al., 2003). Two types of models are used to predict streambank erosion: empirical and process-based (Lai et al., 2012). Empirical models, those that predict erosion based on data alone, do a poor job of predicting erosion with changing boundary conditions (Narasimhan et al., 2015). Process-based models simulate the streambank erosion processes, i.e. subaerial processes, fluvial erosion and mass wasting. While process-based models, such as the Bank-Stability and Toe-Erosion Model (BSTEM) (USDA ARS, 2013; Daly et al., 2015a) and CONservation Channel Evolution and Pollutant Transport System (CONCEPTS) (USDA-ARS, 2000), estimate erosion on a single cross-section or reach (Staley et al., 2006), data requirements on a watershed scale are vast and often not practical for most projects. While HEC-RAS recently incorporated BSTEM into the watershed-scale model (Gibson, 2013), few projects have the resources to gather and incorporate the required data. In order to estimate streambank erosion for an entire watershed and require relatively simple inputs, the Soil and Water Assessment Tool (SWAT) model (Arnold et al., 1998) uses both process-based and empirical routines. This combination of processes allows SWAT to model the physical properties involved in streambank erosion, yet make it more practical to use for large watersheds.

4.3. Background

4.3.1 Streambank Erosion Routine and Parameter Estimation

The current streambank erosion routine from SWAT 2005 (Neitsch et al., 2011) only permits streambank erosion if there is sufficient transport capacity and after the deposited sediment from the previous time step is removed (Table 4.1). The routine uses the excess shear stress equation (Partheniades, 1965; Neitsch et al., 2011) to calculate the streambank erosion rate, ε (m s^{-1}), given as:

$$\varepsilon = k_d\left(\tau_e - \tau_c\right) \tag{4.1}$$

where k_d is the erodibility coefficient (cm^3 N-s^{-1}), τ_e is the effective shear stress (N m^{-2}), and τ_c is the soil's critical shear stress (N/m^2). The k_d and τ_c coefficients are functions of numerous soil properties. SWAT estimates the critical shear stress based on silt and clay content (Julian and Torres, 2006) using the following equation:

$$\tau_c = 0.1 + 0.1779(SC) + 0.0028(SC)^2 - 0.0000235(SC)^3 \tag{4.2}$$

where *SC* is the percent silt and clay content. SWAT predicts k_d using the relationship proposed by Hanson and Simon (2001) based on 83 *in situ* jet erosion tests:

$$k_d = 0.2 * \tau_c^{-0.5} \tag{4.3}$$

Effective shear stress is calculated using the following equations (Eaton and Miller, 2004):

$$\frac{\tau_e}{\gamma * d * s} = \frac{SF_{bank}}{100}\left(\frac{(W + P_{bed}) * \sin\theta}{4 * d}\right) \tag{4.4}$$

$$\log(SF_{bank}) = -1.40 * \log\left(\frac{P_{bed}}{P_{bank}} + 1.25) + 2.25\right) \tag{4.5}$$

where SF_{bank} is the proportion of shear force acting on the bank (N m^{-2}), γ is the specific weight of water (9800 N m^{-3}), d is the depth of water in the channel (m), W is the top width of the bank (m), P_{bed} is the wetted perimeter of the bed (m), P_{bank} is the wetted perimeter of the channel bank (m), θ is the angle of the channel bank from horizontal and s is the slope of the channel (m m^{-1}).

SWAT uses a digital elevation model (DEM) to estimate bed slope and drainage area, assumes the channel has a 2:1 side slope and uses regression equations to estimate bankfull height and width (Neitsch et al., 2011). Currently the same equations are applied worldwide to estimate bankfull width, *BW*, and bankfull height, *BH*, given as:

$$BW = 1.278 * A^{0.6004} \tag{4.6}$$

$$BD = 0.1291 * A^{0.4004} \tag{4.7}$$

where *BW* and *BD* are in meters, and *A* is the drainage area in km^2.

The current streambank-erosion routine has several limitations. Although streambanks on the outside of a meander experience more shear stress (Sin et al., 2012) and erosion (Purvis, 2015), the current routine does not account for the sinuosity of the stream system. The routine does a poor job of redefining channel dimensions after streambank erosion occurs. Therefore, most users assume a balance between erosion and deposition at a cross-section and thus channel dimensions remain constant. Unlike BSTEM and CONCEPTS, which can model multiple bank layers and simulate mass wasting, SWAT assumes a uniform bank and only considers fluvial erosion. Modeling only one layer can lead to large errors in erosion estimates if the critical shear stress and erodibility coefficients of a multilayer streambank are significantly different. Modeling on a large spatial scale leads to many assumptions and simplifications since data are not

often available. Some assumptions include average shear stress on the bank, *BW* and *BD* correctly define channel dimensions and the channel is homogeneous and symmetrical.

4.3.2 Proposed Streambank Erosion Routine

Streambank erosion dependent on transport capacity and bed erosion can underestimate the erosion and does not represent the actual processes. A proposed routine (Narasimhan et al., 2015), currently being beta tested, also uses the excess shear stress equation, but erodes the streambank independent of transport capacity and bed erosion (Table 4.1). The new routine increases the applied shear stress based on the radius of curvature and sinuosity of the reach. The maximum effective shear stress occurs on the outside of the meander and is affected by the degree of sinuosity. Sin et al. (2012) developed a dimensionless multiplication bend factor to adjust the effective shear stress on the meander, which was the ratio of the maximum shear stress experienced at the bends divided by the average channel shear. The dimensionless bend factor (K_b) is estimated using (Sin et al., 2012; Narasimhan et al., 2015):

$$K_b = 2.5\left(\frac{R_c}{W}\right)^{-0.32} \tag{4.8}$$

where R_c is the radius of curvature (m) and W is the top width (m). R_c is estimated using the empirical relationship based on several studies and has a wide range of applicability over widths ranging from 1.5 m (Friedkin, 1945) to 2,000 m (Fisk, 1947) (Williams, 1986), given as:

$$R_c = 1.5 * W^{1.12} \tag{4.9}$$

The maximum effective shear stress on the outside of the meander, τ_e^*, is calculated using:

$$\tau_e^* = K_b * \tau_e \tag{4.10}$$

To calculate the total mass of sediment eroded from streambanks, the channel is divided into straight and meandering reaches. The length of the reach affected by meandering is calculated using the inverse of the sinuosity (ratio of channel length to the straight-line length). The effective shear stress of the reach affected by the sinuosity is then multiplied by K_b while the straight section is not. For the meandering section of a reach, erosion is only calculated from the critical bank while both banks erode for the straight section.

Table 4.1. Streambank erosion processes and equations for the current version (SWAT 2005), the 2015 beta version and the proposed modifications to the beta version.

Process	2005 SWAT	2015 Proposed Subroutine	2015 Proposed Subroutine Modifications
Streambank erosion	Excess shear stress equation; function of transport capacity	Excess shear stress equation	Excess shear stress equation
Applied shear stress equation	Equations 4.4, 4.5	Equations 4.4, 4.5	Equations 4.15, 4.16
Incorporates sinuosity	No	Yes	Yes
Bank dimensions	Bankfull width/depth	Bankfull width/depth	Top width/bank depth
Channel heterogeneity	No	No	Yes; area adjustment factor

4.3.3 Objectives

The proposed routine has only been tested on cohesive soils in the Cedar Creek watershed in North-Central Texas with lateral bank erosion rates ranging from 0.025 to 0.37 m yr^{-1}. More testing is needed before the routine is incorporated into the official SWAT release and used by watershed modelers worldwide. Although the proposed routine addressed some of the current model limitations, several additional limitations and assumptions remain. Therefore, three modifications were made to the proposed routine and tested on the Barren Fork Creek watershed in northeastern Oklahoma. The Barren Fork Creek watershed has non-cohesive soils and lateral bank erosion rates

ranging from 0.5 to 8.7 m yr^{-1} (Heeren et al., 2012; Midgley et al., 2012; Daly et al., 2015a). The Barren Fork Creek is representative of noncohesive gravel-dominated channels and will add important information to the streambank erosion routine validation and assessment.

At a watershed-scale there is typically limited site specific streambank data, both spatially and temporally. While stream reaches range in length from a few hundred meters to several kilometers, only one value for each parameter may be used to characterize the reach in SWAT. Gathering data for channel parameters by reach is a daunting task and for most projects is not feasible; therefore, the most critical parameters need to be identified to focus data collection efforts. Although there is considerable uncertainty in each of these parameters (Chaubey et al., 2005; Wechsler, 2007; Bieger et al., 2015), no study has compared field-measured to SWAT derived parameters and their influence on streambank erosion.

The objectives of this research were to (1) improve the current SWAT streambank erosion routine, (2) test the routine on the composite streambanks and (3) compare SWAT-default channel parameters to field-measured values and assess their influence on erosion. Results of this study will provide recommendations to watershed modelers and managers to focus data collection and parameter estimation efforts on the most critical streambank erosion parameters, thus providing more accurate model predictions.

4.4 Methods

4.4.1 Proposed SWAT Streambank Erosion Modifications

Three proposed modifications were made to the SWAT 2015 streambank-erosion routine beta version to address some of the model's current limitations. The first

replaced the empirical applied shear stress equation with a process-based equation. The second replaced the bankfull width and depth with the top width and bank depth. Finally, the third added an area-adjustment factor to account for heterogeneous stream channels (Table 4.1).

To accurately predict streambank erosion, a good estimate of the applied shear stress is essential. Currently, SWAT uses an empirical equation derived from laboratory studies using symmetrical trapezoidal channels (Eaton and Miller, 2004). This can introduce error when used outside the conditions under which the equation was developed. The proposed replacement equation is process-based and used by CONCEPTS (USDA-ARS, 2000):

$$\tau = \gamma * R * S_f \tag{4.11}$$

where R is the hydraulic radius (m) and S_f is the friction slope (m m^{-1}). The friction slope is computed using the following equation:

$$S_f = \frac{n^2 * Q^2}{A^2 * R^{\frac{4}{3}}} \tag{4.12}$$

where Q is the average flow rate (m^3 s^{-1}), n is Manning's roughness coefficient and A is the cross-sectional area (m^2).

SWAT currently assumes a symmetric trapezoidal channel with dimensions derived from bankfull width and depth. There are two primary reasons to replace bankfull parameters with top width and bank height. First, identifying and measuring bankfull width is subjective and thus carries considerable uncertainty (Johnson and Heil, 1996). Second, bankfull measurements are often less than top width and bank height measurements, thus resulting in inaccurate modeling of stream flow depth (Figure 4.1).

In summary, replacing bankfull parameters with top width and bank height more accurately defines the stream system being modeled.

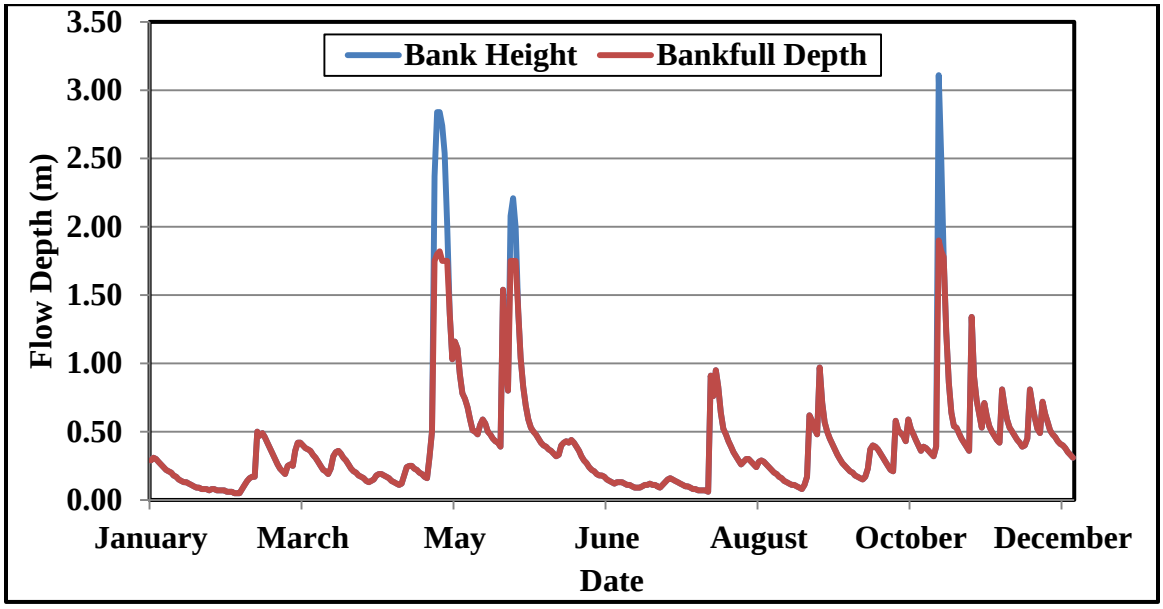

Figure 4.1. SWAT simulated flow depth when using bankfull depth or bank depth to define the channel cross section on the Barren Fork Creek for 2011.

To accurately model streambank erosion, channel dimensions must mimic those of the studied stream system. Although the current SWAT model is constrained by its symmetrical trapezoidal channel dimensions, a simple area-adjustment factor to account for a heterogeneous channel cross-section is proposed (Figure 4.2). No natural channel is symmetrical with a flat and level streambed, and thus assuming a trapezoidal channel will result in errors predicting flow depth. The proposed equation is:

$$A_{adj} = a * A \tag{4.13}$$

where A_{adj} is the adjusted channel cross-sectional area (m^2), A is the irregular cross-sectional area (m^2), and a is a dimensionless adjustment factor less than or equal to 1.0. The variable a is calculated by dividing the irregular cross-sectional area by the trapezoidal area. The trapezoidal area is based on the SWAT input for top width, channel depth and side slope.

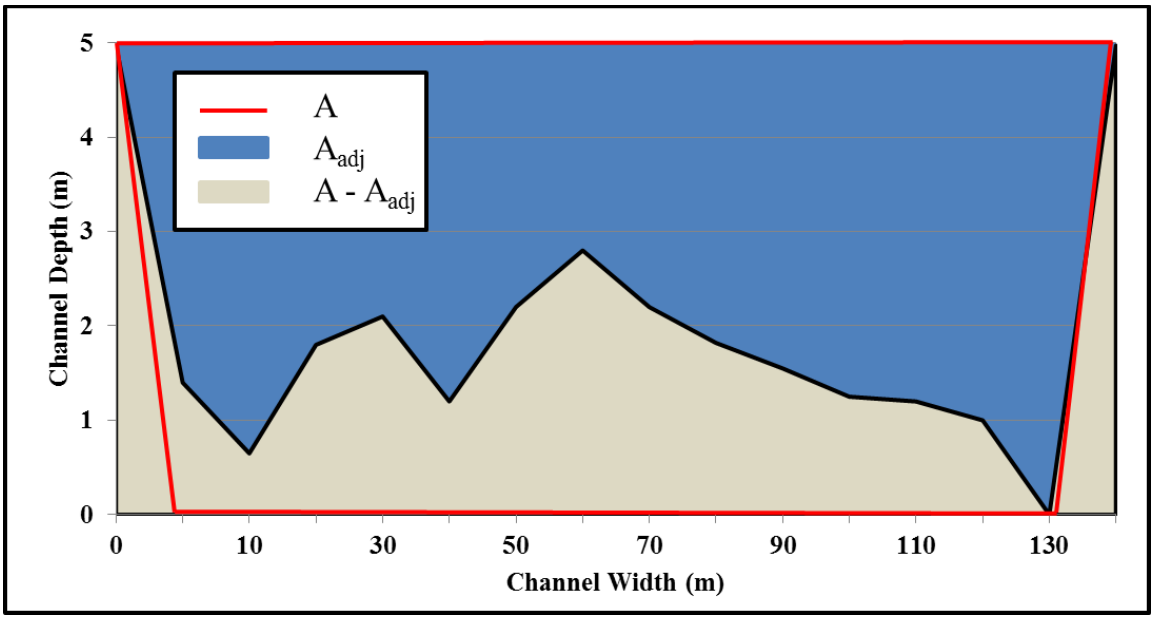

Figure 4.2. SWAT trapezoidal and measured stream cross sections at the United States Geological Survey gage station 07197000 used to adjust cross sectional area and calibrate flow depth. A_{adj} is the measured cross-sectional area of the natural channel, A is the cross-sectional area of an assumed trapezoidal channel, A-A_{adj} is the difference between the trapezoidal and measured cross sections.

4.4.2 Study Site

The streambank erosion routine was tested on the Barren Fork Creek watershed, located in the Ozark Highland Ecoregion in northeast Oklahoma and northwest Arkansas. Recent research on the Barren Fork Creek, an Oklahoma designated Scenic River, has shown that streambank erosion is a significant P source (Miller et al., 2014). Miller et al. (2014) estimated that 36% of the streambanks in the Barren Fork Creek watershed were unstable and eroding. In another study by Heeren et al. (2012), lateral bank erosion on 23 reaches on the Barren Fork Creek and Spavinaw Creek, approximately 50 km north, averaged more than 7 m from 2003 to 2008, with one reach losing 55 m.

The watershed has a drainage area of 890 km^2 (Figure 4.3) and is composed of 55% forest, 30% pasture and 13% hay meadow (Storm and Mittelstet, 2015). The headwaters begin in Washington County, Arkansas, flow through Adair County, Oklahoma before discharging into the Illinois River in Cherokee County, Oklahoma just

north of Ferry Tenkiller Lake. The streambanks consist of a fining upward sequence of basal gravels and overlying silts and clays derived from overbank deposition (Figure 4.4). Due to readily available information, the ten study sites from Miller et al. (2014) were used in this study (Figure 4.3). Available information for each site included pebble counts used to define the median particle size (d_{50}), bank height, and streambank total and water soluble soil P. Seven of the ten sites historically had riparian vegetation protection while three were unprotected. Since SWAT only models one streambank layer, the entire streambank was modeled as a gravel layer. Although fluvial erosion is the dominant streambank process in the watershed, ignoring mass wasting of the cohesive layer may lead to the underprediction of the streambank erosion, especially during those events where the top cohesive layer becomes saturated and unstable (Fox and Wilson, 2010).

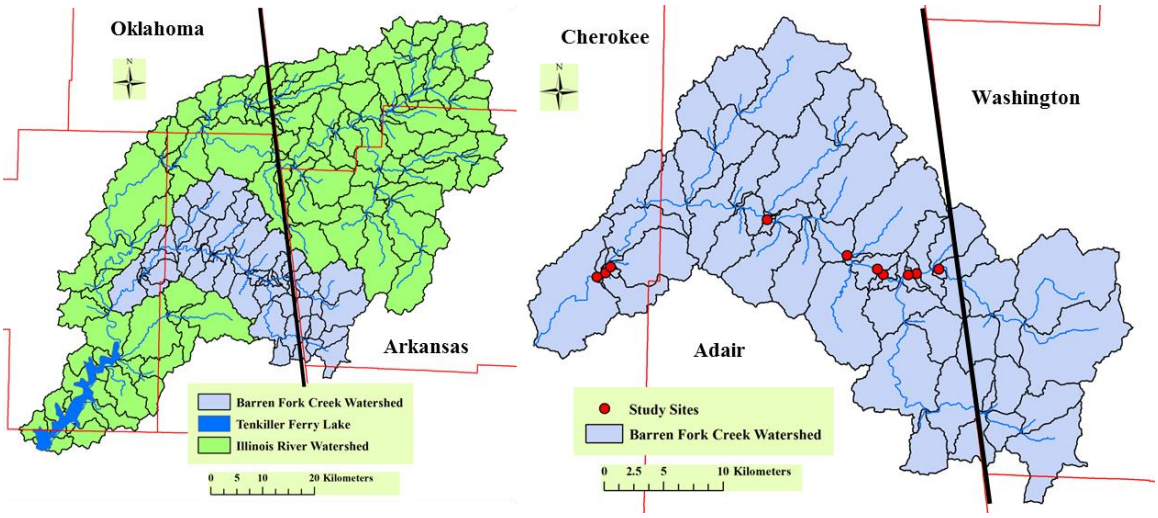

Figure 4.3. Illinois River and Barren Fork Creek watersheds in Oklahoma and Arkansas (left) and the Barren Fork Creek watershed showing ten study sites (right).

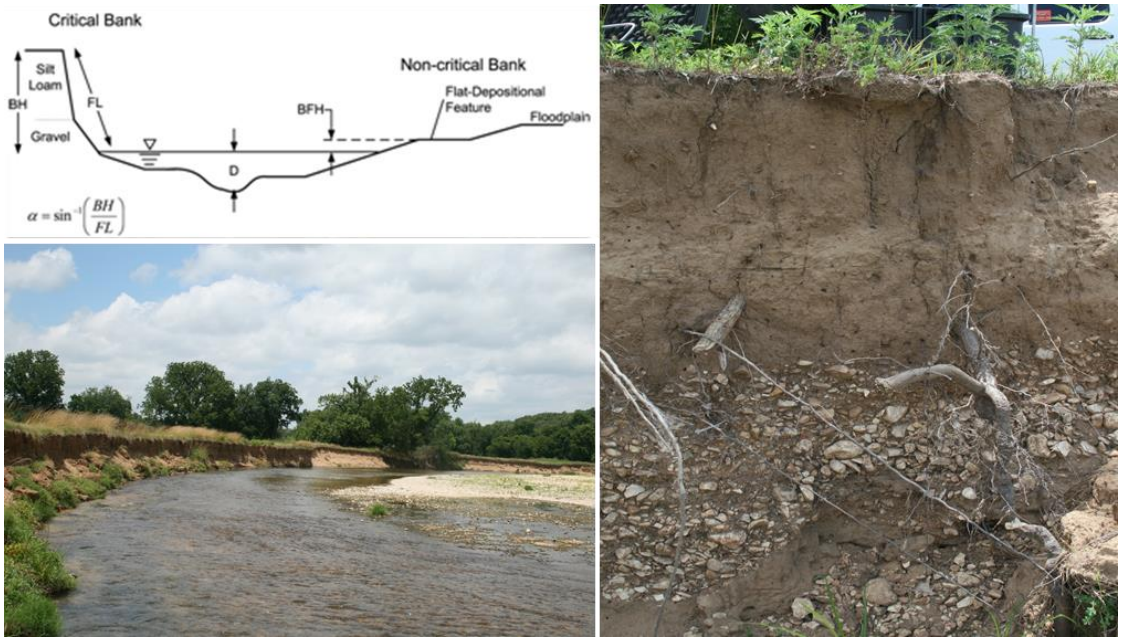

Figure 4.4. Typical stream channel profile in the Barren Fork Creek with one critical bank and one non-critical bank. Right image illustrates the underlying gravel layer and the silty loam topsoil for the critical bank (Heeren et al., 2012).

4.4.3 Parameter Measurement

Parameter measurement was divided into two categories, data mining and field data collection. Data mining included existing online digital data and derivatives, such as bed slope, R_c and sinuosity. Field data included measured stream and streambank information, i.e. *BW*, *BD*, top width, bank height, side slope and τ_c .

Kocian (2012) found that aerial images and topographic maps were highly correlated with measured data. Therefore, bed slope for each study site reach was calculated using 1:24,000 USGS topography maps and National Aerial Imagery Program (NAIP) aerial images to estimate elevation change and stream length, respectively. Both sinuosity and R_c were calculated using NAIP images from 2003, 2008 and 2013 and averaging the calculated values. The R_c was calculated for each of the meandering reaches by visually overlaying and fitting a circle to each bend (Figure 4.5), and then comparing estimates obtained from Equation 4.9 using *BW* and top width.

A total of 28 stream cross-sections, starting from the Oklahoma/Arkansas state line to the confluence of Barren Fork Creek and the Illinois River (Figure 4.6; Appendix A) were surveyed using a laser level, measuring tape and survey rod; eight at cross-over points, nine at meanders and eleven at straight cross sections (Figures 4.6 and 4.7). Locations of cross-sections were based on available access points. Cross-over points were defined as the river reaches where the thalweg crossed from one side of the channel centerline to the other, straight reaches were defined as reaches with a sinuosity less than 1.1 (Dey, 2014) and meanders were the remaining reaches with a sinuosity greater than 1.1. Two of the straight reaches included surveys completed at the USGS gage stations near Eldon, Oklahoma (07197000) and Dutch Mills, Arkansas (07196900). At each of the 28 sites, the following data were collected: *BW*, *BD*, top width, bank height and side slope.

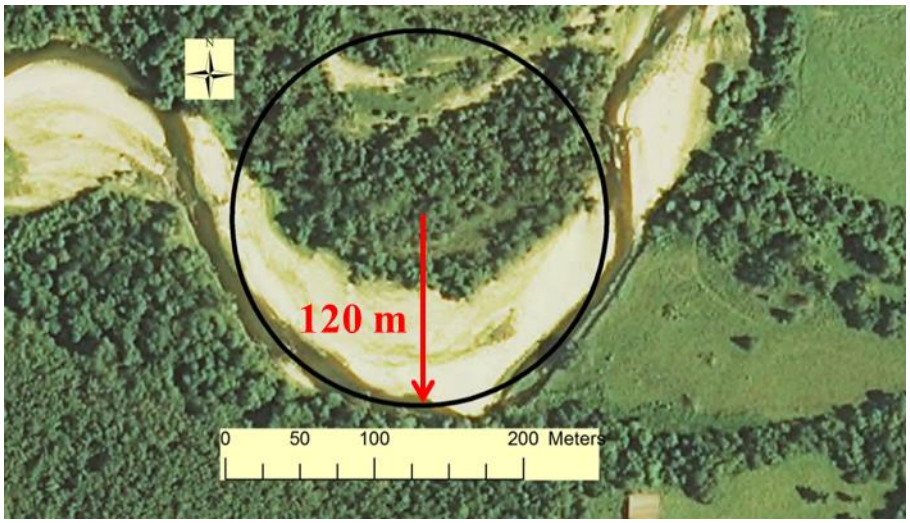

Figure 4.5. Radius of curvature estimate at site F on the Barren Fork Creek using a 2013 National Agriculture Imagery Program image.

The measured irregular channel cross section for each of the straight and meandering reaches were compared to the trapezoidal cross section, which was calculated from the measured top width and side slope to obtain the *a*. FlowMaster V8 (Bentley,

2015) was used to estimate the water depth of the irregular cross-section versus the water depth using a trapezoidal cross-section with and without using *a*. Three representative cross-sections were chosen: meander, and heterogeneous and homogenous straight reaches. Flow depths were calculated assuming uniform flow and Manning's formula.

BW was identified by physical stream indicators, such as change in elevation, deposited sediment and vegetation (USGS, 2004). The bankfull area, calculated using the cross-sectional survey, was divided by *BW* to obtain the average *BD*. The measured bankfull parameters were compared to the values calculated by SWAT as well as two equations proposed by Bieger et al. (2015). The equations currently used by the SWAT model to estimate *BW* and *BD* were derived several years ago based on limited measured data. Bieger et al. (2015) compiled *BW* and *BD* data from 51 studies across the US, one equation for the entire US and eight regional equations based on physiographic divisions. The entire US equations for BW_{us} and BD_{us}, in m, are (Bieger et al., 2015):

$$BW_{US} = 2.70 * A^{0.352} \tag{4.14}$$

$$BD_{US} = 0.30 * A^{0.213} \tag{4.15}$$

Dutnell (2000) developed regional equations for the Internal Highland Region, which includes the Barren Fork Creek, for BW_{ihr} and BW_{ihr}, in m, given as:

$$BW_{ihr} = 23.23 * A^{0.121} \tag{4.16}$$

$$BD_{ihr} = 0.27 * A^{0.267} \tag{4.17}$$

Measured d_{50} coupled with an alternative τ_c equation (Millar, 2005) were used to estimate τ_c for the streambank gravel layer using the following algorithm developed specifically for non-cohesive gravel particles (Millar, 2005):

$$\tau_c = 0.05 * \tan(\varphi) * \rho * g(SG-1)d_{50} * \sqrt{1-\frac{\sin^2 * \theta}{\sin^2 * \phi}} \tag{4.18}$$

where ρ is the density of water (1000 kg m^{-3}), g is gravitational acceleration (9.81 m s^{-2}), SG is the specific gravity of the bank soil (assumed to be 2.65 for all soils), d_{50} is the mean particle diameter of the soil (m), φ is the angle of repose (degrees), and θ is the bank angle (assumed to be 25° for all streambank soils and 0° for all streambed sediments) (Daly et al., 2015a). Although equation 4.3 was derived using cohesive soils, the equation was successfully used for gravel layers at similar sites by Daly et al. (2015a) and Midgley et al. (2012) and thus will be used in this study.

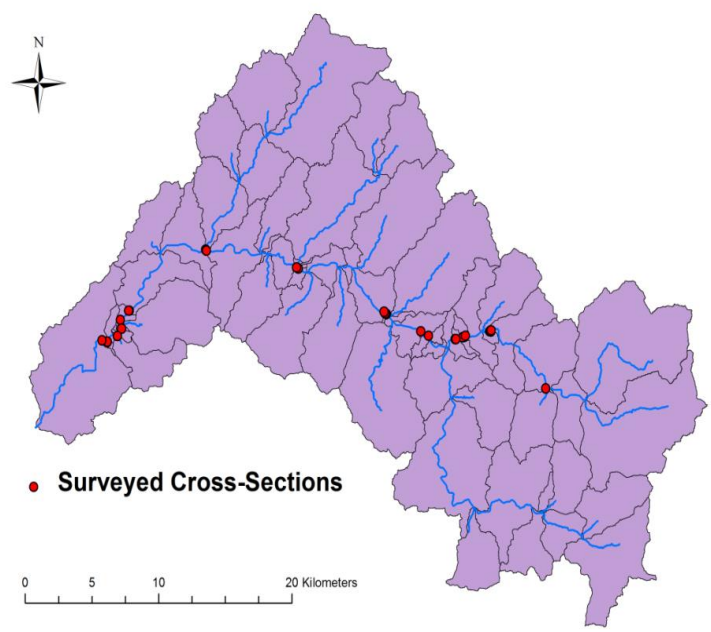

Figure 4.6. Location of 28 surveyed cross-sections surveyed on the Barren Fork Creek 2015.

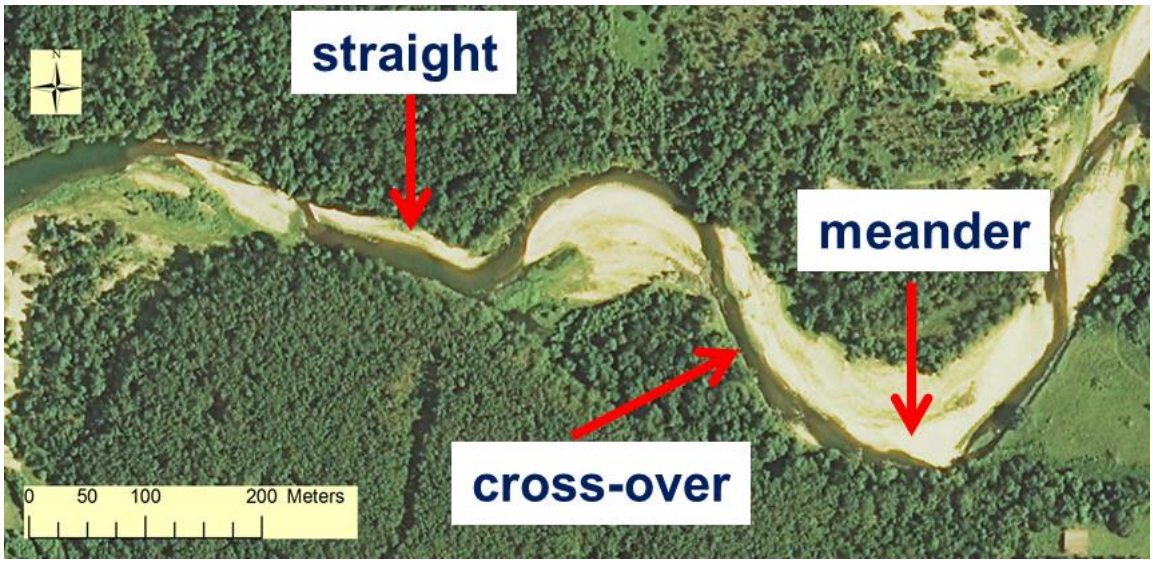

Figure 4.7. Examples of straight, meandering and cross-over stream reaches on a 2013 National Agriculture Imagery Program image.

4.4.4 SWAT Model Setup

A SWAT model for the Barren Fork Creek watershed was created similar to the Illinois River watershed SWAT model (Chapter 3). The landcover dataset, developed from 2010 and 2011 Landsat images, was used as well as the 10-m USGS DEM and SSURGO soil data. The watershed had minor point sources at Westville, Oklahoma and Lincoln, Arkansas, two USGS stream gages located near Eldon, Oklahoma and Dutch Mills, Arkansas, and three weather stations (Figure 4.8). Outlets were added to the model upstream and downstream of the ten study sites (Miller et al., 2014) to produce SWAT output files for each study reach to predict stream flow and streambank erosion. Management practices, litter application rates and Soil Test Phosphorus for each subbasin were obtained from the (Chapter 3). The final SWAT model consisted of 73 subbasins, 2,991 HRUs and eight land covers. The primary land covers were forest (55%), pasture (30%) and hay meadow (13%).

4.4.5 Model Evaluation

<u>Streamflow and Flow Depth</u>

The SWAT model was calibrated to observed daily and monthly baseflow, peak flow and total flow at USGS gage stations 07197000 and 07196900. Since Oklahoma's Mesonet began in November 1994, streamflow was calibrated and validated from 2004 to 2013 and 1995 to 2003, respectively. The USGS Hydrograph Separation Program (HYSEP) was used to estimate baseflow (Sloto and Crouse, 1996). Channel dimensions, obtained from the cross-sectional surveys at the two USGS gage stations, were used in the SWAT model along with an initial Manning's *n* of 0.025 (Daly et al., 2015a). Manning's *n*, the only value not measured, was manually adjusted to calibrate flow depth. The Coefficient of Determination (R^2) and Nash Sutcliff Efficiency (NSE) (Nash and Sutcliffe, 1970) were used to evaluate the model's performance (Moriasi et al., 2007).

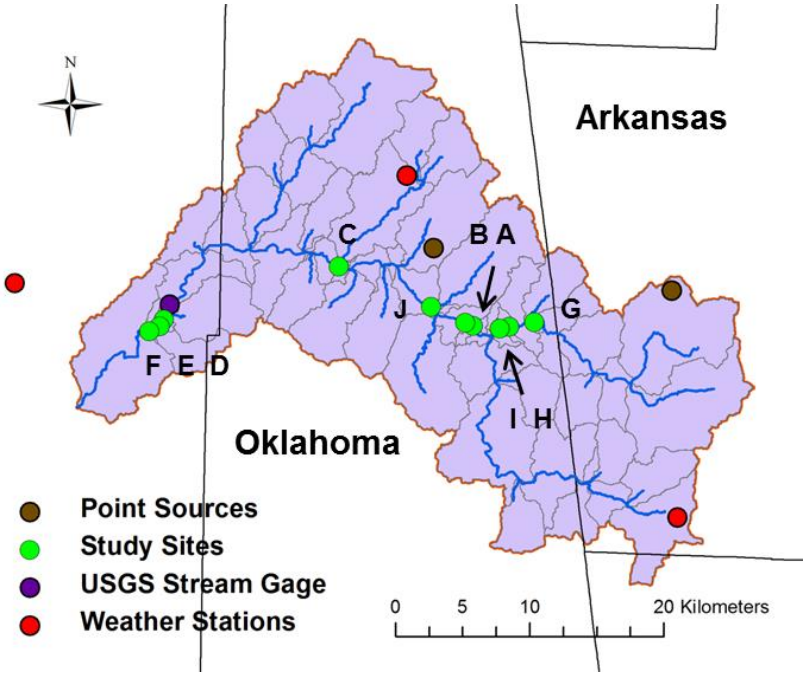

Figure 4.8. United States Geological Survey gage station, weather stations and stream reach study sites for the Barren Fork Creek watershed.

Streambank Erosion

NAIP images from 2003 to 2013 were used to estimate the lateral streambank retreat (Figure 4.9) (Heeren et al., 2012; Miller et al., 2014). The NAIP images were used to estimate the eroded streambank widths and lengths, and to calculate the eroded surface area (*EA*). Streambank depth (D_{ts}), in m, was based on Miller et al. (2014) and the 28 surveys, which was used to calculate the total sediment loading (*TS*), in kg, from each reach using:

$$TS = EA * D_{ts} * \rho_b \tag{4.19}$$

where ρ_b is the soil bulk density (g cm^{-3}). A weighted ρ_b based on the bank composition (Miller et al., 2014) was used to estimate the average ρ_b for the bank.

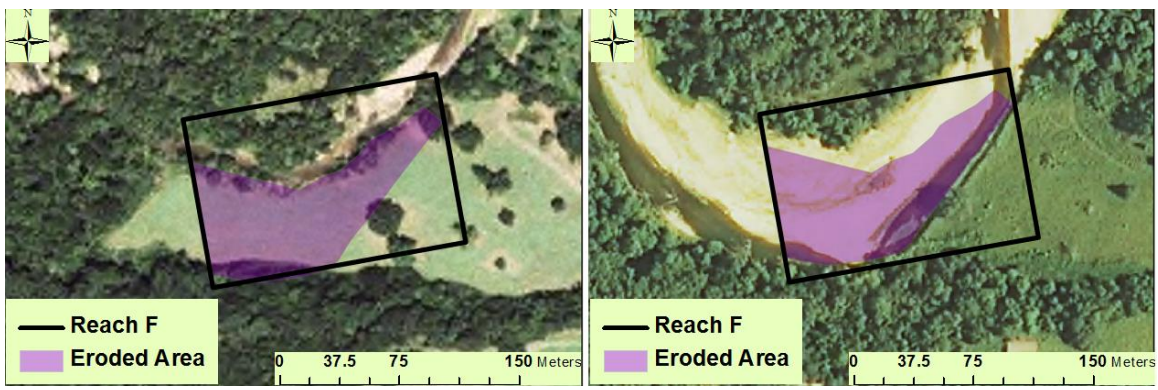

Figure 4.9. 2003 (left) and 2013 (right) National Agricultural Imagery Program aerial images with polygons illustrating the streambank retreat (purple) during the period for study Site F on the Barren Fork Creek.

4.5 Results and Discussion

4.5.1 Area Adjustment Factor Verification

Figure 4.10 illustrates differences in *a* and flow depth for three cross-sectional reaches: meander (a=0.72), heterogeneous straight reach (a=0.77) and homogenous straight reach (a=0.93). Due to land cover changes and deforestation, gravel has eroded

from the upland areas throughout the Barren Fork Creek watershed. Much of this gravel has reached the Barren Fork Creek, resulting in changes in the channel dimensions and flow dynamics of the creek. The highly irregular cross-sections (Figure 4.10A and B) were more representative of the cross-sections on the Barren Fork Creek. The more irregular the measured channel cross section, the more important *a* becomes in accurately estimating the flow depth. For each cross-section, the flow depth was simulated more accurately when using a.

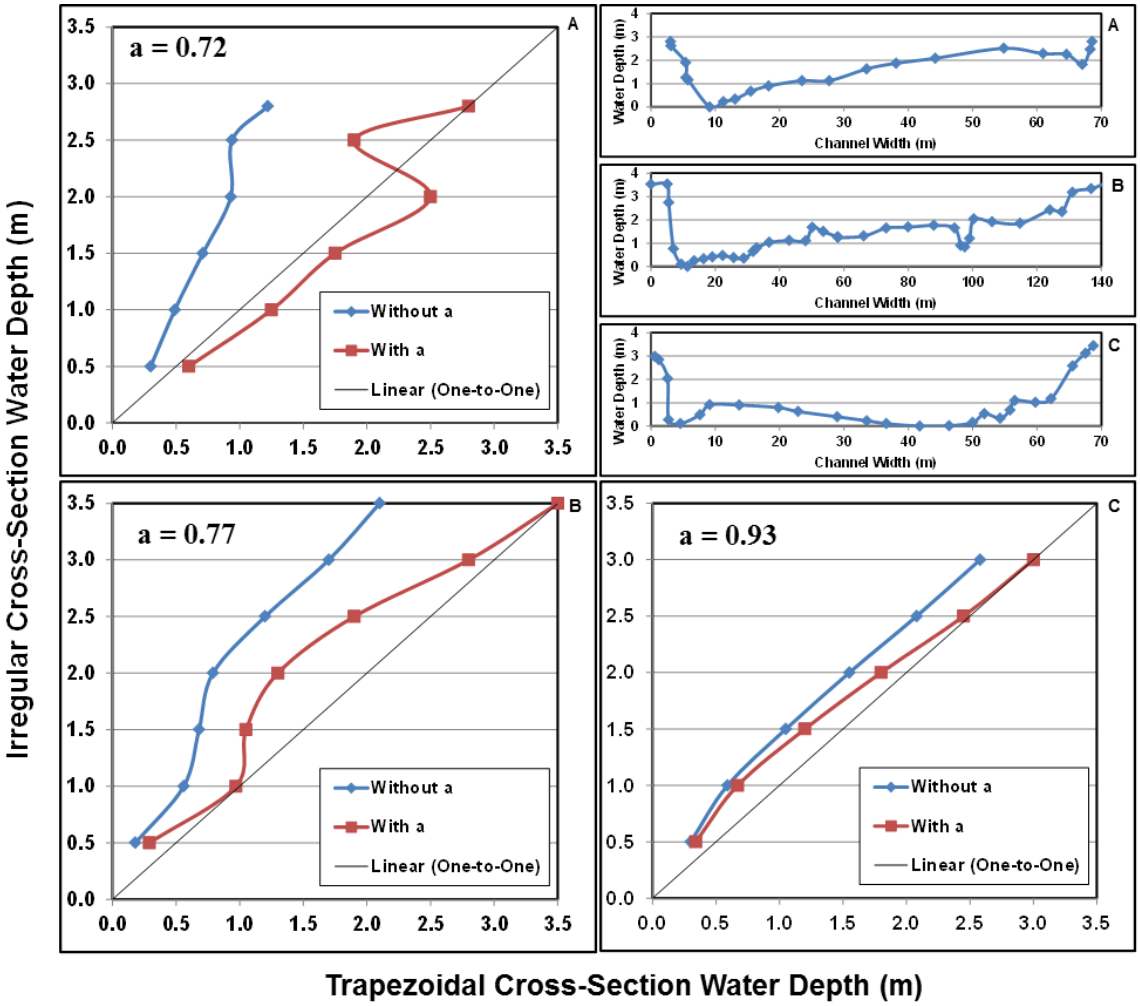

Figure 4.10. FlowMaster-calculated flow depth for the irregular cross-section compared to the trapezoidal cross-section with and without the area adjustment factor (a). Cross-section A is a meander, B is a heterogeneous straight reach and C is a homogenous straight reach.

4.5.2 Flow and Flow Depth Calibration

Streamflow calibration predictions were 'very good' (Moriasi et al., 2007) with monthly R^2 and NSE for the calibration (2004 to 2013) and validation (1995 to 2003) periods ranging from 0.78 to 0.82. Based on the cross-sectional surveys, a trapezoidal channel with a top width of 136 m, D_{ts} of 4.97 m and side slopes of 1.35 m m^{-1} were used to calculate *A* at USGS gage station 07197000. This *A* was then compared and adjusted using *a* (Equation 4.13) until it matched the irregularly-shaped surveyed *A* (see Figure 4.2). An *a* of 0.66 was calculated, which signifies that water is not flowing in 34% of the trapezoidal *A* at a flow depth of 4.97 m. The procedure was repeated at the upstream USGS gage station 07196900 using an *a* of 0.95.

Flow-depth calibration at the two USGS gage stations yielded the same Manning's *n*, 0.05, which was applied to each reach in the watershed. The calibrated daily flow depth at gage station 07197000 had an R^2 of 0.64 and NSE of 0.56 (Figure 4.11), while the USGS gage station upstream near Dutch Mills, Arkansas had an R^2 and NSE of 0.49. The calibrated Manning's *n* of 0.05 was in the range for other gravel bed streams (Chow, 1959; USGS, 1989) based on the procedure developed by Cowan (1956).

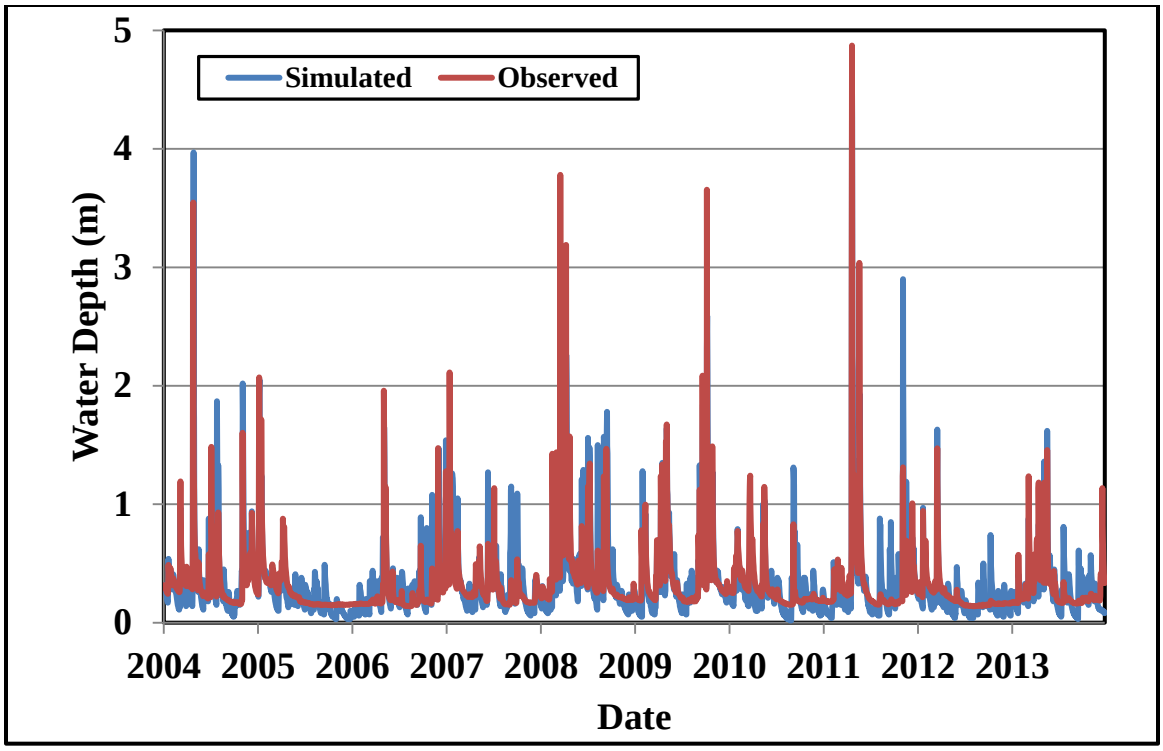

Figure 4.11. Observed and simulated water depth at the United States Geological Survey gage station 07197000 for the period 2004 to 2013.

4.5.3 SWAT Calculated vs Measured Parameters

4.5.3.1 Data Mining Parameters

The estimated bed slope using topographic maps and NAIP aerial images were not normally distributed; therefore, a Mann-Whitney Rank Sum Test was used to compare bed slopes. At a 95% confidence level, the bed slope calculated using the topographic maps and NAIP aerial images was not significantly different than the bed slope estimated from the 10-m DEM (Figure 4.12). However, the DEM underestimated the bed slope near the watershed outlet and overestimated the bed slope in the head waters. Kocian (2012) also found low accuracy with the 10-m DEM in estimating bed slope compared to LIDAR and topographic maps. Based on these findings and those by Kocian (2012), the bed slope measurements derived from aerial images and topographic maps were utilized.

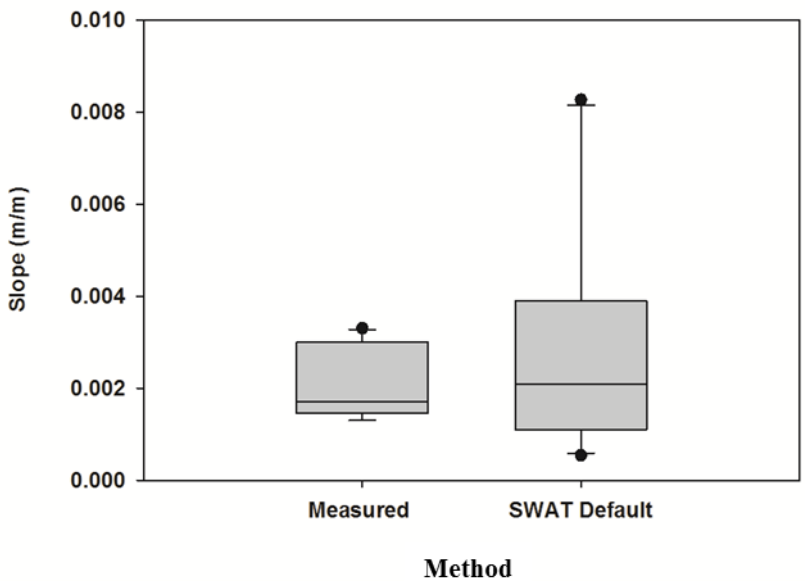

Figure 4.12. Channel bed slope calculated from the topographic map and aerial images (measured) and digital elevation model (SWAT default) for the Barren Fork Creek.

The sinuosity at the ten study sites ranged from 1.0 to 2.5 with an average of 1.3. Of the ten study sites, four were classified as straight reaches (less than 1.1), three sinuous (1.1-1.5) and three meandering (greater than 1.5) (Dey, 2014). Note that Equation 4.9 was valid for reaches with a sinuosity greater than 1.2 (Williams, 1986). The average radius of curvature for the four study reaches with a sinuosity greater than 1.2 was 151 m. Applying equation 4.9, the average R_c of the four sites was 131 m and 216 m using *BW* and top width, respectfully (Figure 4.13). An analysis of covariance was conducted at a 95% confidence level to compare the measured R_c versus those derived from Equation 4.9 and the top width or *BW*. Neither the slope nor slope intercept were significantly different for either the top width or *BW*.

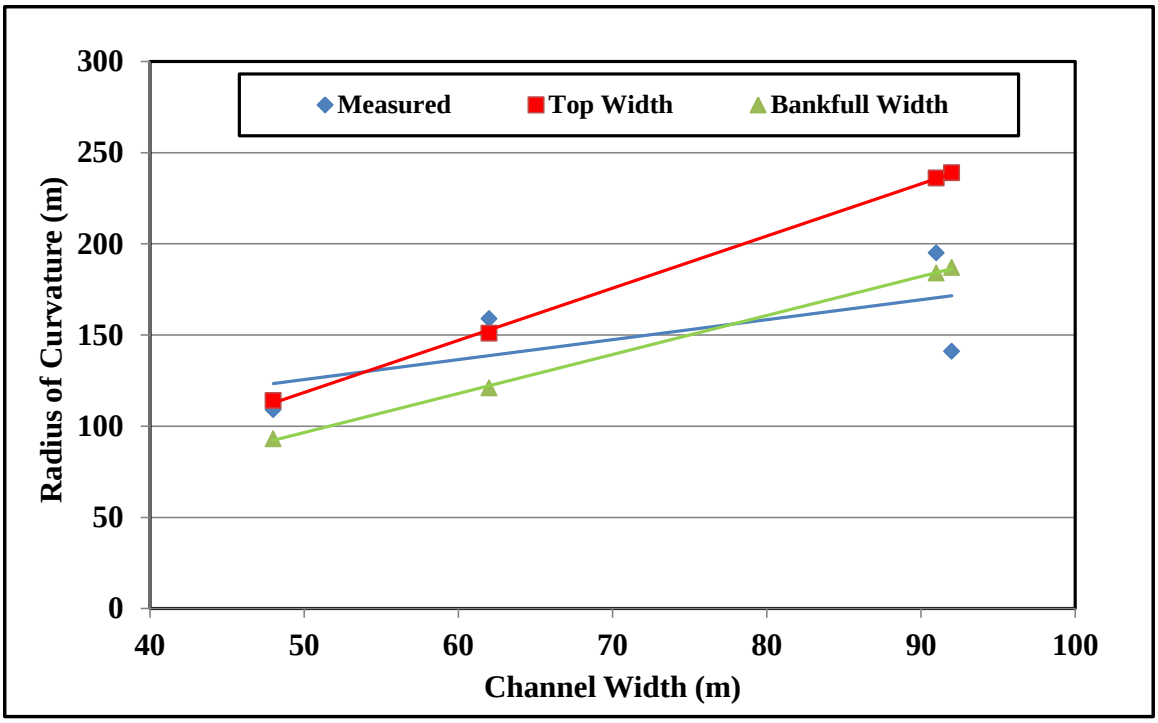

Figure 4.13. Measured and calculated radius of curvature for four reaches with a sinuosity greater than 1.2 on the Barren Fork Creek. The radius of curvature was calculated using Equation 4.9 ($R_c = 1.5*W^{1.12}$), where *W* is the measured bankfull width or top width.

4.5.3.2 Field-measured Parameters

Field measurements at cross-over points and the corresponding drainage area were used to derive equations for *BW* and *BD* (Dutnell, 2000). The measured *BW* had an R^2 of 0.72 and was compared to the values derived from the three empirical equations using an analysis of covariance with a 95% confidence level (Figure 4.14). Neither the slope nor the slope intercept for the SWAT global regression (equation 4.6) were significantly different with p-values of 0.23 and 0.07, respectively. For the proposed regional regression (Equation 4.17), the slope was significantly different, but the slope intercept was not with a p-value of 0.08. Both the slope and slope intercept were significantly different for the proposed US regression (equation 4.15).

The measured *BD* versus *DA* had an R^2 of 0.66 and was also compared to the values derived from the three empirical equations using an analysis of covariance with a

95% confidence level (Figure 4.15). The slope was not significantly different for the SWAT global regression (Equation 4.6), yet the slope intercept was significantly different with p-values of 0.07 and 0.02, respectively. For the proposed regional and US regression (Equation 4.17 and 4.15), neither the slope nor the slope intercept were significantly different with p-values of 0.49 and 0.11 for the proposed regional regression and 0.19 and 0.72 for the US regression, respectively.

These results support the findings by Bieger et al. (2015) that concluded that the regional curves were more reliable than the US equations. The regional equations can be improved by incorporating additional sites, especially for the Internal Highlands (seven sites) and Laurentian Upland (six sites) (Bieger et al., 2015). With the large number of SWAT users outside the US, there is a need for counties outside the US to develop their own regional or watershed specific regression equations; however, in this study the global regression estimated the bankfull parameters adequately.

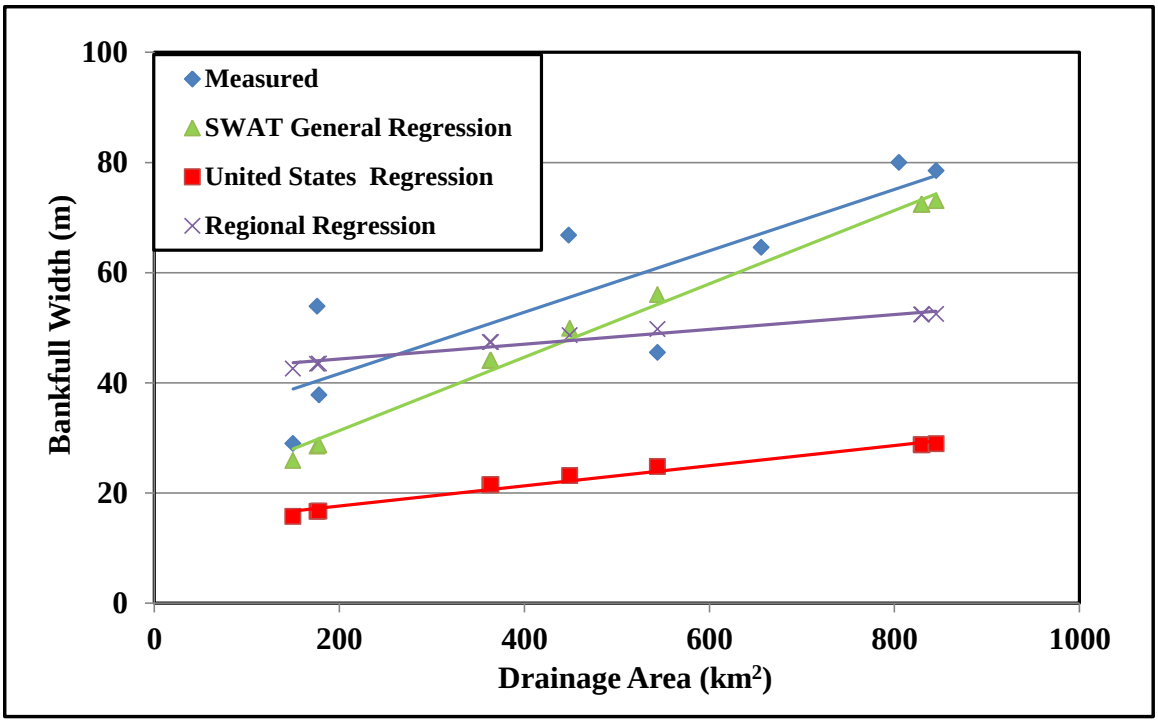

Figure 4.14. Measured bankfull width and calculated bankfull width using three empirical equations vs drainage area for the Barren Fork Creek.

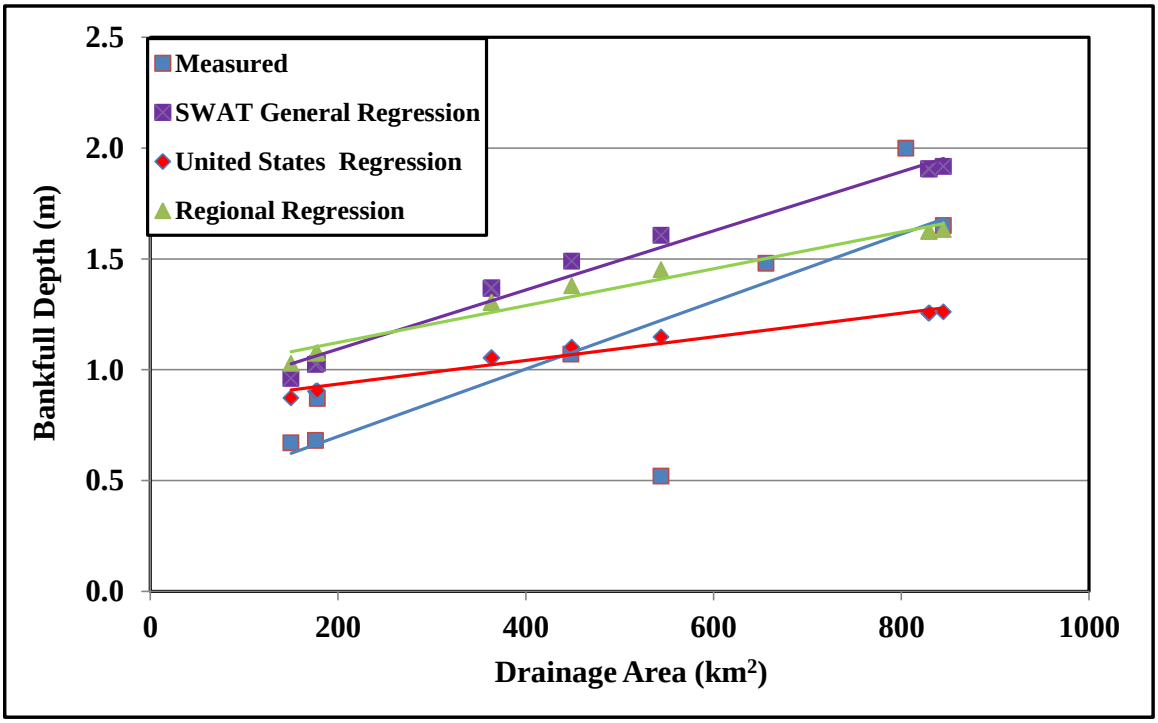

Figure 4.15. Measured bankfull depth and calculated bankfull depth using three empirical equations vs drainage area for the Barren Fork Creek.

SWAT defined the gravel bank containing 65% gravel, 15% sand, 15% silt and 5% clay, which was similar to the ten study sites that measured 68% gravel, 15% sand, 10% silt and 7% clay. Based on the measured *SC* content of the banks (Julian and Torres, 2006), τ_c was 4.6 Pa and k_d was 0.093 $cm^3\ N^{-1}\ s^{-1}$ (Equations 4.2, 4.3). Using the measured d_{50} of the ten study sites (1.3 to 2.5 cm) and Equation 4.18, τ_c ranged from 3.5 Pa to 8.7 Pa with an average of 5.6 Pa. Both methods produced similar results for τ_c, 4.6 versus 5.6, which agrees with Daly et al. (2015b).

The field surveys measured stream channel side slope, top width and D_{ts}. Average measured side slopes for the straight reaches and meanders were 4.8:1 and 1.4:1, respectively (Figure 4.16). Based on an ANOVA with a Tukey's multiple comparison test at a 95% confidence level, the measured side slopes from straight and meandering reaches and SWAT default values were all significantly different. Top width measurements taken at straight reaches were used to characterize all the stream reaches (Figure 4.17). Measurements were attempted at cross-over and meandering reaches, but many of the cross sections had 25 to 100 m of thick vegetation preventing accurate measurements. Based on an analysis of covariance at a 95% confidence level, the measured *BW* and top width were not significantly different. However, both the slope and slope intercept were significantly different for the measured D_{ts} and *BD* (Figure 4.18).

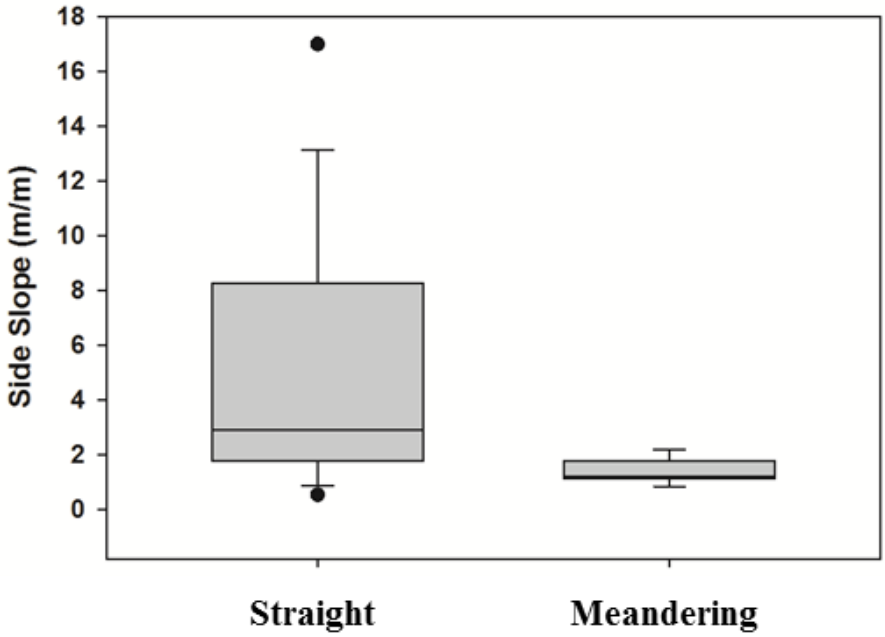

Figure 4.16. Measured side slopes for straight and meandering reaches on the Barren Fork Creek.

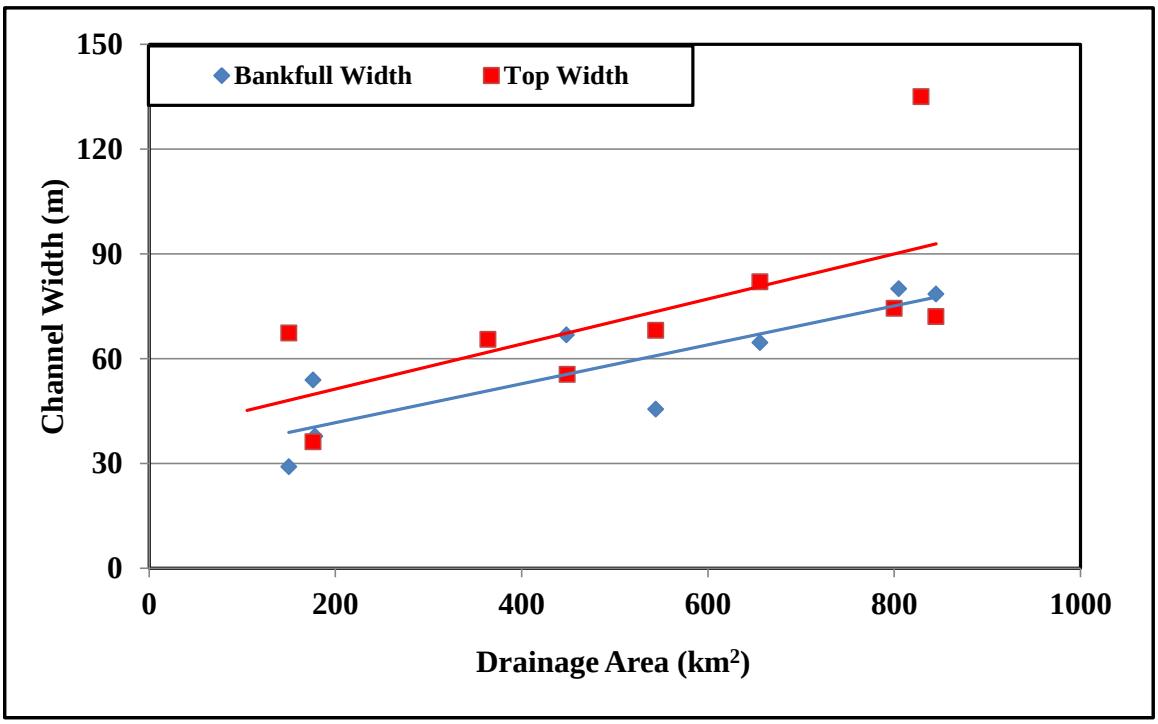

Figure 4.17. Measured straight reach top width and bankfull width for the Barren Fork Creek.

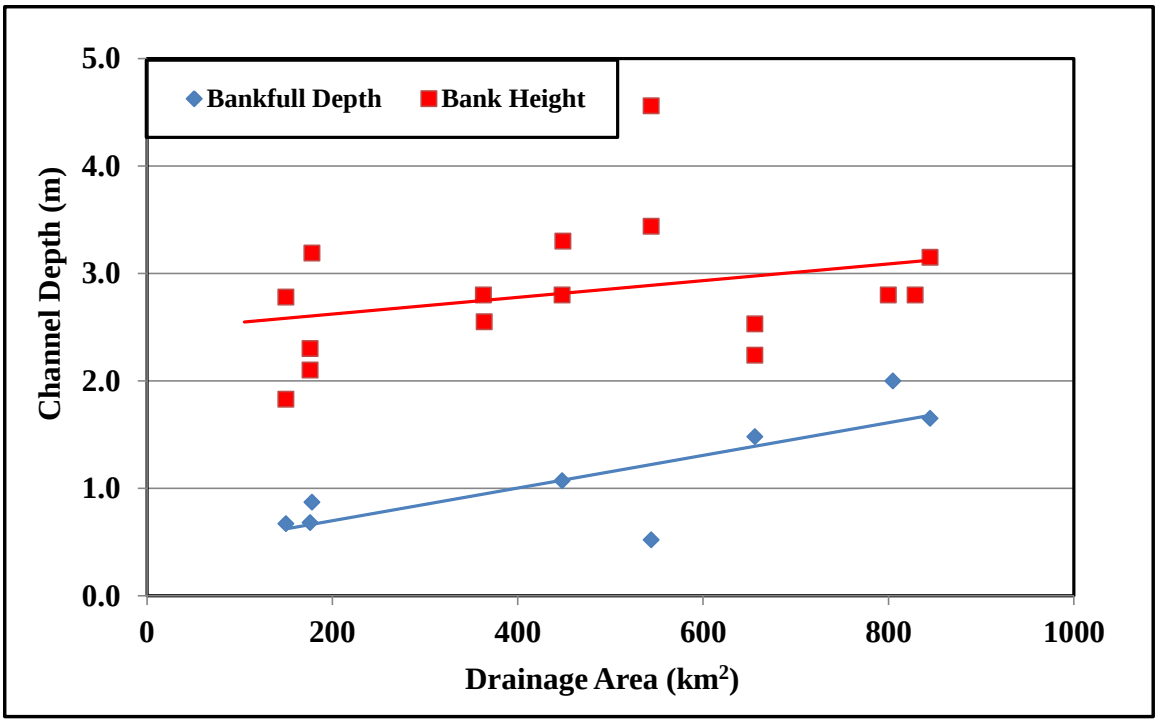

Figure 4.18. Measured straight reach bankfull depth and bank height for the Barren Fork Creek.

4.5.4 Observed vs Simulated Streambank Erosion

SWAT-estimated parameters were replaced with parameter estimates based on measured data using a regression equation with watershed area as the independent variable or an average measured value. The following regression equations were derived using measured bed slope and top width:

$$BS = 4.3*10^{-9}*WA^2 - 6.7*10^{-6}*DA + 0.00369 \quad (4.20)$$

$$TW = 0.0787*DA + 35.384 \quad (4.21)$$

where *BS* is the bed slope in m m^{-1}, *TW* is the top width in m and *DA* is the watershed area in km^2. The sinuosity measured at each site using aerial photographs was used in the model. However, R_c could not be measured using aerial photographs for large reaches. Therefore, equation 4.9 was used to estimate the R_c based on *DA*. It should be noted that the R_c measurements were taken from the aerial photographs were not significantly

different at the 95% confidence level from the estimates using equation 4.9. Since there was no longitudinal trend with *DA* along the length of the Barren Fork Creek, the average τ_c (5.6 Pa), k_d (0.085 cm^3 N^{-1} s^{-1}), side slope (3.1:1), D_{ts} (2.8 m) and *a* (0.78) were used for each reach in the model simulations.

The average observed streambank erosion (gravel and topsoil) from 2004 to 2013 at the ten sites was 2,830 Mg yr^{-1}, and ranged from 219 Mg yr^{-1} at site J to 10,300 Mg yr^{-1} at site F (Figure 4.19). Using the SWAT model with default parameters, the SWAT 2015 streambank erosion routine beta version was tested using two methods, the empirical and proposed applied shear stress equations. The average simulated streambank erosion using the empirical equation was 1,360 Mg yr^{-1} compared to 2,510 Mg yr^{-1} for the process-based equation (Figure 4.19). Both models under predicted the streambank erosion at sites F and E and over predicted the erosion at several other sites, such as D and J. Though the correlation with observed erosion was poor for both equations, the NSE was better for the proposed shear stress equation (Table 4.2).

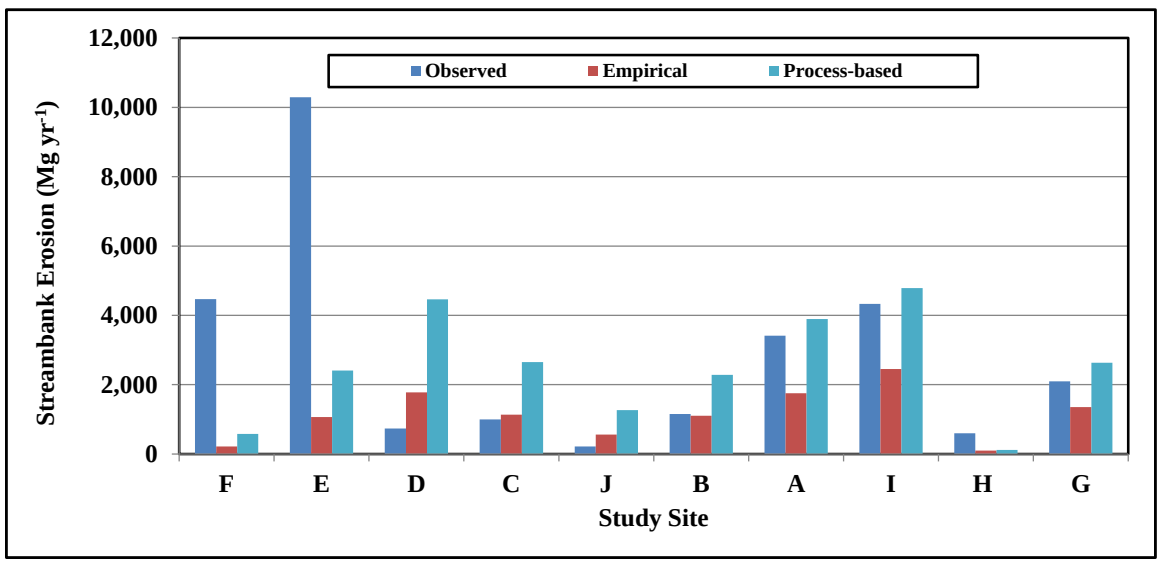

Figure 4.19. Measured and simulated streambank erosion using empirical and process-based applied shear stress equations using the SWAT model with default parameters at ten study sites on the Barren Fork Creek from 2004 to 2013. Empirical is the applied shear stress equation currently used by the SWAT model and process-based is the proposed process-based applied shear stress equation.

4.5.4.1 Data Mining

Incorporating measured *BS* into the model resulted in an improvement in both the R^2 and NSE (Table 4.2). Much of this improvement was due to the incorporation of measured *BS* for sites E and F. Based on the SWAT default using DEM, the *BS* at sites E and F were 0.00095 and 0.00054, respectively. The measured values using the topographic maps and NAIP images were 0.0015 for both sites, which were slope increases of 58 and 180 percent. Incorporating the measured sinuosity and R_c further improved model predictions. Though the average erosion for the data mining scenario decreased overall by 4 to 5% using the two applied shear stress equations, the simulated erosion at the meandering reaches (sites E and F) increased as did the R^2 and NSE (Table 4.2). Based on these results, model simulations can be improved by incorporating measured *BS*, sinuosity and R_c, which can all be measured without field-collected data. The correlation between observed and measured streambank erosion for both the empirical and process-based model had an R^2 of 0.65, even though the average erosion was under predicted using the empirical equation.

4.5.4.2 Bankfull Parameters

Replacing SWAT default *BW* and *BD* with measured values resulted in an average streambank erosion reduction of 41% and 30% for the empirical and process-based equations, respectively. While the *BW* and *BD* from the proposed regional equation reduced the average erosion by only 4 to 10%, the quantity of erosion increased 46 to 126% when the bankfull parameters derived from the US equation were incorporated into the model (Table 4.3). Using an ANOVA and Tukey's comparison test at 95% confidence level, none of the simulation results using the proposed shear stress

equation were significantly different, yet the simulation results using the empirical shear stress equation and the US regression equations was significantly different compared to the other simulation results using the empirical equations. This re-enforces the need for US SWAT applications to use the regional regression equation instead of the US regression equation.

Table 4.2. SWAT simulated streambank erosion using different methods to estimate streambank erosion parameters using both the empirical and proposed process-based equations for Barren Fork Creek. Empirical is the empirical applied shear stress equation currently used by the SWAT model. Process-Based is the proposed process-based applied shear stress equation. Methods include SWAT default parameters and replacing default parameters with several measured parameters: bed slope, literature based and bankfull width and depth. Literature based parameters include bed slope, sinuosity and radius of curvature. NS=Nash Sutcliff Efficiency.

Parameter	Applied Shear Stress Equation					
	Empirical			Process-Based		
	Erosion (Mg yr^{-1})	R^2	NSE	Erosion (Mg yr^{-1})	R^2	NSE
SWAT default	1,150	0.02	-0.33	2,510	0.01	-0.16
Bed slope	1,000	0.03	-0.20	2,230	0.57	0.38
Literature based	1,090	0.02	-0.12	2,410	0.65	0.49
Measured bankfull parameters	680	0.01	-0.55	1,750	0.05	-0.14
Regional bankfull regression	1,100	0.55	-0.35	2,260	0.01	-0.26
Proposed United States regression	2,600	0.65	-0.47	3,660	0.01	-0.92

4.5.4.3 Field Data

Incorporating measured τ_c into the model resulted in a 22 to 25% reduction in the predicted average erosion for the two applied shear stress equations. Increasing τ_c by just one Pa influenced the erosion significantly and corroborates the findings by Narasimhan et al. (2015) that streambank erosion is very sensitive to τ_c. This supports the need for further research evaluating τ_c and k_d using empirical equations and field-measured data. Although the τ_c using the silt and clay content was within the range of measured values in this study, Daly et al. (2015b) found out the Julian and Torres (2006) relationship predicted a smaller range of values over a large range of silt and clay content for cohesive soils.

Replacing the SWAT default side slope of 2:1 with the field-measured side slope of 3.1:1 increased erosion at each site by 34% and 80% for the empirical and applied shear stress equation, respectively (Table 4.3). Issues arise when adjusting side slope, but not the *W* and bank height. Modifying the side slope, but using the smaller bankfull width instead of the *W*, decreases the stream channel *A* and results in excessive shear stress applied to the banks. Replacing the default *BW* and *BH* with the measured *W* and bank height increased the stream channel *A* and reduced the erosion by approximately 30% for the two applied shear stress equations. Replacing all of the measured values, side slope, T*W* and D_{ts}, with the measured values only increased the erosion by 15% using the empirical equation and reduced the erosion by 2% using the process-based equation. Incorporating *a* resulted in an increase of 172% for the empirical equation and 28% for the process based equation. The sensitivity of the empirical applied shear stress equation to decreases in the *A* is a result of more shear stress applied to the streambank instead of the streambed (Equations 4.4 and 4.5). Although replacing the default values with field measurements did not improve model predictions in this study (Table 4.3), more confidence can be given to the model predictions. Further research is needed to determine if replacing the *BS*, sinuosity and R_c is sufficient or if cross-sectional surveys should be conducted.

Table 4.3. Influence field-measured parameters have on simulated streambank erosion using both the empirical and proposed process-based applied shear stress equations. Empirical is the empirical applied shear stress equation currently used by the SWAT model and Process-Based is the proposed process-based applied shear stress equation. Each method includes literature based parameters, which includes bed slope, sinuosity and radius of curvature. All measured data includes the following: critical shear stress, side slope, top width and bank height. A_{adj} = area adjustment factor.

Parameter	Applied Shear Stress Equation					
	Empirical			Process-Based		
	Erosion (Mg yr^{-1})	R^2	NSE	Erosion (Mg)	R^2	NSE
Literature based (baseline)	1,090	0.65	-0.12	2,410	0.65	0.49
Critical shear stress	850	0.27	-0.37	1,800	0.32	0.10
Side slope	1,960	0.38	0.16	3,240	0.35	0.31
Top width and bank height	720	0.30	-0.42	1,740	0.46	0.15
All measured data	1,250	0.28	-0.14	2,350	0.46	0.32
All measured data + A_{adj}	2,960	0.34	0.31	3,080	0.47	0.41

4.5.4.4 Cover Factor

Seven of the ten study sites were protected with riparian vegetation while three sites (F, E, and A) were unprotected (Miller et al., 2014). The average observed erosion from 2003 to 2013 at the three unprotected sites was 6,160 Mg yr^{-1} compared to 1,450 Mg yr^{-1} for the protected sites. Although quantifying the impact of riparian vegetation on streambank erosion is challenging on a watershed scale, vegetation can significantly impact the streambank erosion (Daly et al., 2015a; Harmel et al., 1999). While vegetation does not reduce the erodibility of the gravel layer, the stability of the cohesive top layer increases with root density. Micheli and Kirchner (2002) studied similar banks in California and found that the protected sedge banks only failed after the bank was significantly undercut. After the geotechnical streambank failure, the overbank soil remained partially attached providing temporary armoring against further erosion. The unprotected meadow banks failed more frequently and detached completely from the bank, thus preventing temporary armoring. Although the gravel layer is not affected by vegetation, the streambank erosion of the top cohesive soil layer is reduced. Therefore,

due to the current limitations of the model, the τ_c was increased for the seven banks with riparian protection based on the following equation (Julian and Torres, 2006):

$$\tau_c^* = \tau_c * CH_{\text{cov}} \tag{4.22}$$

where τ_c^* is the effective critical shear stress (N m^{-2}) adjusted for vegetative cover and CH_{COV} is the multiplication factor called channel cover factor. Based on Narasimhan et al. (2015), we chose to use a CH_{COV} of two for forest. Therefore, the τ_c for the seven protected sites was increased from 5.6 to 11.2 N m^{-2} and the k_d was decreased to 0.06 cm^3 N^{-1} s^{-1} using Equation 4.3. Including the channel CH_{COV} improved the R^2 and overall model predictions (Figure 4.20). R^2 and NSE were 0.58 and 0.42 using the empirical equation and 0.66 and 0.52 using the process-based equation, respectfully. Both shear stress equations using the CH_{COV} adequately predicted streambank erosion except at reaches E and I. Reach E had an unusually large quantity; more than twice as much as the other two unprotected sites. Although reach I had good riparian protection in 2003 (Figure 4.21), it had 4,330 Mg yr^{-1} streambank erosion compared to a combined total of 5,800 Mg yr^{-1} for the remaining six protected sites. Results from these two reaches demonstrate that models cannot account for all processes occurring in the natural world.

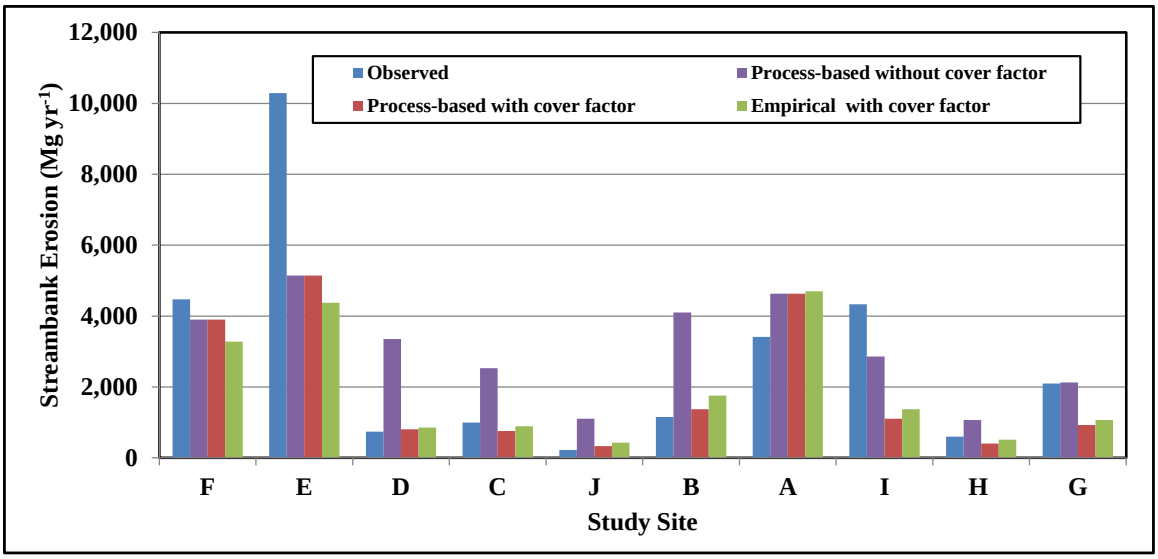

Figure 4.20. Observed streambank erosion compared to SWAT simulated erosion with and without the streambank cover factor for the Barren Fork Creek from 2004 to 2013. Empirical equation is the applied shear stress equation currently used by the SWAT model and process-based is the proposed process-based applied shear stress equation.

Figure 4.21. Streambank erosion at reach I on the Barren Fork Creek from National Agricultural Imagery Program aerial 2003 (left) to 2013 (right) images. The red line is the location of the reach in 2003.

4.6 Conclusions

The modified streambank-erosion routine for the SWAT model improved the predicted streambank erosion for composite streambanks. Although the process-based applied shear stress equation was the most influential modification, incorporating the top width, streambank depth and area-adjustment factor more accurately represented the measured irregular cross-sections and improved the model predictions compared to

observed data. Since field-data collection is not feasible for every project, simulations were performed using literature and field-based data.

If collecting stream data to estimate channel parameters is not possible due to financial, geographic or time constraints, literature-based data can provide good streambank-erosion estimates. The current SWAT and proposed regional regression equations adequately estimated bankfull width and bankfull depth. The proposed US equation, on the other hand, produced poor results and therefore should not be used for the conditions studied. While equation 9 provided an adequate estimate of the radius of curvature, the measured bed slope using aerial images and topography maps should be used in place of the DEM-derived estimates. Incorporating the radius of curvature, sinuosity, bed slope and the global or regional bankfull parameters improved model predictions at the ten study sites. The R^2 increased from 0.01 to 0.65 and the NSE increased from -0.92 to 0.49.

Although results from this study demonstrated that using field-measured parameter estimates may not statistically improve model predictions for the conditions studied, other time periods or watersheds may be different. If limited field work can be conducted, multiple measurements of the critical shear stress (τ_c) are recommended. The τ_c was one of the most sensitive parameters and it can be incorporated into the model without affecting the cross-sectional area of the stream channel. If resources permit, complete cross-section surveys should be conducted throughout the stream system to quantify the top width, streambank depth, side slope and area-adjustment factor. Each of these parameters affects the cross-sectional area and should be replaced together. In

general, the more watershed-specific measured data incorporated into the model, the more confident the user can be in the model predictions.

Further testing of the ability to predict τ_c using the silt and clay content is needed as well as exploring other τ_c and erodibility coefficient relationships. More research is also needed to quantify how root density from different types of riparian vegetation impact τ_c . Future research also needs to address the streambank-erosion routine limitations, specifically incorporating multiple-layer banks and the modification of channel dimensions throughout the simulation.

CHAPTER V

ESTIMATING STREAMBANK EROSION AND PHOSPHORUS LOADS FOR THE BARREN FORK CREEK WATERSHED USING A MODIFIED SWAT MODEL

5.1 Abstract

Phosphorus (P) and streambank erosion are problematic in the Barren Fork Creek watershed in northeast Oklahoma and northwest Arkansas. Previous SWAT modeling efforts of the watershed have not accounted for the contribution of stream banks as a P source due to lack of field data and model limitations. This is believed to be the cause for under predicting total and particulate P during large storm events. The objectives of this research were to model the streambank erosion and P for the Barren Fork Creek using a modified SWAT model. Measured streambank and channel parameters were incorporated into a flow-calibrated SWAT model and used to estimate streambank erosion and P for the Barren Fork Creek using the latest streambank-erosion routine and newly incorporated process-based applied shear stress equation. The predicted streambank erosion was 215,000 Mg yr^{-1} versus the measured 160,000 Mg yr^{-1}. Streambank erosion contributed 47% of the total P to the Barren Fork Creek and also improved P predictions compared to observed data, especially during the high flow

events. Due to this influx of streambank P to the system and the current in-stream P routine's limitations, the in-stream P routine was modified by introducing a long-term storage coefficient, thus converting some of the particulate P to long-term storage. Of the total P entering the stream system, approximately 65% left via the watershed outlet and 35% was stored in the floodplain and stream system. This study not only provided local, state and federal agencies with accurate estimates of streambank erosion and P contributions for the Barren Fork Creek watershed, it demonstrated how watershed-scale model, such as SWAT, can be used to predict both upland and streambank P.

5.2 Introduction

Excess phosphorus (P) and sediment are two major stream and reservoir pollutants. Often non-point sources, such as livestock, urbanization and commercial fertilizer, and point sources are responsible for elevated P and turbidity. Currently, over $3.7 billion is spent in the United States annually on natural resource conservation (Monke and Johnson, 2010; White et al., 2014), with much of this spent on the implementation of conservation practices to reduce the quantity of P and sediment reaching waterways from agricultural activities. White et al. (2014) found that row crops and point sources were the most significant contributors of P reaching the Gulf of Mexico, although in some watersheds, streambanks can contribute up to 80% of the total sediment (Simon et al., 1996) and a significant quantity of total P (Kronvang et al., 2012; Laubel et al., 2003; Langendoen et al., 2012). Conservation practices aimed at reducing P runoff from agricultural land and point sources will thus be less effective if streambank erosion is not addressed.

One area of concern is the highly-sinuous stream system of the Barren Fork Creek in northeast Oklahoma and northwest Arkansas. The Barren Fork Creek, along with its receiving waterbodies Illinois River and Tenkiller Ferry Lake, are on the Oklahoma 303(d) list of impaired waters due to excess P (DEQ, 2012). In the last sixty years, the once-clear waters have become eutrophic due to pollutant loads from urbanization and livestock production, especially poultry (Cooke et al., 2011). Although tens of millions of dollars have been spent on improving the water quality of one of Oklahoma's few state-designated scenic rivers, most of these monies have been used for the implementation of conservation practices in the upland areas. In previous SWAT modeling efforts of the Illinois River watershed (Storm et al., 2006; Storm et al., 2010; Storm and Mittelstet, 2015), streambank erosion was not addressed due to lack of data and model limitations. Due to the meandering stream system and highly erosive streambanks, P derived from streambank erosion is hypothesized to be the cause for underestimating P during the high flow events. Recent work by Miller et al. (2014) has strengthened this hypothesis. They found that 36% of the streambanks on the Barren Fork were unstable and contribute approximately 90 Mg of TP annually, almost half the total P reaching the watershed outlet.

In the last decade, the streambank-erosion routine in the Soil and Water Assessment Tool (SWAT) (Arnold et al, 1998) has undergone considerable improvements. The latest beta version, previously only tested on cohesive soils in the Cedar Creek watershed in Texas (Narasimhan et al., 2015), uses an excess shear stress equation to calculate the erosion rate, ε (m/s), given as:

$$\varepsilon = k_d\left(\tau_e - \tau_c\right) \tag{5.1}$$

where k_d is the erodibility coefficient ($cm^3\ N^{-1}\ s^{-1}$), τ_e is the effective shear stress ($N\ m^{-2}$), and τ_c is the soil's critical shear stress ($N\ m^{-2}$). The k_d and τ_c coefficients are functions of numerous soil properties. Improvements on predicting applied shear stress to streambanks were accomplished by incorporating sinuosity and radius of curvature to account for the effects of meander. Though the current routine uses an empirical equation to estimate the applied shear stress (Eaton and Millar, 2004), Mittelstet (Chapter 4) proposed an alternative process-based equation (USDA-ARS, 2000) for SWAT:

$$\tau = \gamma * R * S_f \tag{5.2}$$

where γ is the specific weight of water (N/m^3), R is the hydraulic radius (m) and S_f is the friction slope (m/m). The friction slope is computed using the following equation:

$$S_f = \frac{n^2 * Q^2}{A^2 * R^{\frac{4}{3}}} \tag{5.3}$$

where Q is the average flow rate (m^3), n is Manning's roughness coefficient and A is the area (m^2).

This study will test and validate this updated routine on a flow-calibrated SWAT model of the Barren Fork Creek watershed. Specifically the objectives of this study are (1) to predict streambank erosion for the Barren Fork Creek using the proposed streambank-erosion routine (Chapter 4), (2) model P in the watershed with and without incorporating P derived from streambank erosion and (3) determine the significance of streambank erosion relative to upland P sources.

5.3 Methods

5.3.1 Study Site

The Barren Fork Creek watershed has a drainage area of 890 km^2 and is composed of approximately 55% forest, 24% well-managed pasture, 6% over-grazed pasture and 13% hay meadow (Storm and Mittelstet, 2015). The Barren Fork Creek, a fourth-order stream, is approximately 73 km in length and is located in the Ozark Highland Ecoregion in northeast Oklahoma and northwest Arkansas (Figure 5.1). The headwaters begin in Washington County, Arkansas, and flow through Adair County, Oklahoma before discharging into the Illinois River in Cherokee County, Oklahoma just north of Tenkiller Ferry Lake. Barren Fork Creek is a State-designated Scenic River and is on the Oklahoma 303(d) list for nutrient and sediment related impairments (U.S. EPA, 2015). Typical of the Ozark Highland Ecoregion, the watershed is characterized by cherty soils and gravel-bed streams (Heeren et al., 2012). The highly dynamic streambanks consist of alluvial gravel deposits underlying silty loam topsoil (Figure 5.2). The sinuous stream often has a critical bank on the outside of the meander and a gravel bed on the inside bank.

Figure 5.1. Illinois River and Barren Fork Creek watersheds in northeast Oklahoma and northwest Arkansas.

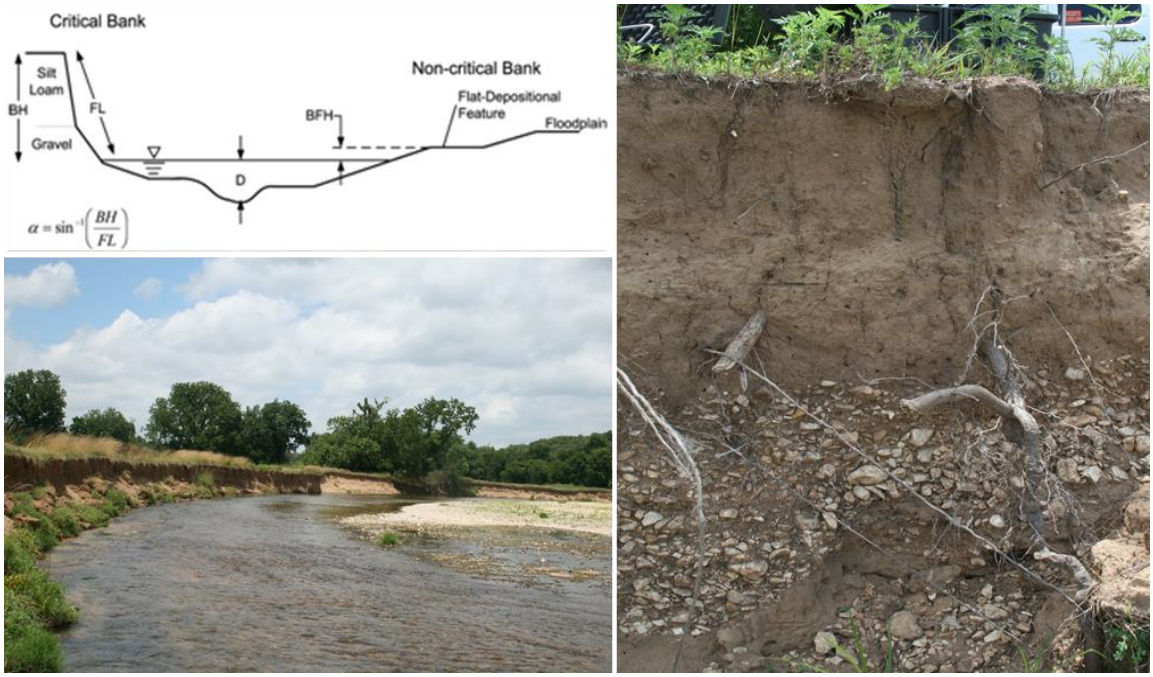

Figure 5.2. Typical stream channel profile in the Barren Fork Creek with one critical bank and one non-critical bank. Right image illustrates the underlying gravel layer and the silty loam topsoil for the critical bank (Heeren et al., 2012).

5.3.2 SWAT Model Description

SWAT is a basin-scale hydrological/water-quality model used to predict streamflow and pollutant losses from watersheds composed of mixed land covers, soils and slopes. The model was developed to assist water resource managers to assess water quantity and/or quality in large river watersheds and as a tool to evaluate the impact of agricultural conservation practices. The SWAT model, a product of over 30 years of model development by the US Department of Agriculture Agricultural Research Service, has been extensively used worldwide (Gassman et al., 2007, 2014). The model is process-based and can simulate the hydrological cycle, crop yield, soil erosion, and nutrient transport.

An ArcGIS interface can be used to develop model input of land cover, soils, elevation, weather, and point sources, and define the flow network. The interface divides the watershed into subbasins, which are further split into hydrological response units (HRUs). Each HRU has one soil type, one land use and one slope. The model uses the Modified Universal Soil Loss Equation (MUSLE) to calculate sediment yield for each HRU. This sediment, along with nutrients, are combined for each subbasin and routed through the stream reach. The water and sediment, along with any other pollutants, are routed from reach to reach until arriving at the watershed outlet. Many field-scale activities, such as planting dates, irrigation, fertilization, grazing, harvesting and tillage, are utilized by SWAT as management options scheduled by date. Further details on the model inputs and the theoretical aspects of hydrology, nutrient cycling, crop growth and their linkages are provided in Neitsch et al. (2009).

This study used SWAT 2012 version 583 and the recently incorporated simplified in-stream P routine (White et al., 2012), which consists of two components. The first component represents the transformation of soluble P to particulate P (i.e. the uptake of soluble P by algae and P precipitation) and its interactions with sediment, which is based on an equilibrium P concentration (EPC). EPC is the concentration at which there is no net sorption or desorption from benthic sediments into the water column. If the EPC is greater than the concentration of soluble P in the water column, P moves from the benthos to the water column; the reverse occurs if the EPC is less than the soluble P. The second component represents the deposition and scour of particulate P (sediment-bound P and algal P) to/from the benthos, which is based on the ratio of flow to bankfull discharge.

5.3.3 SWAT Model Modifications

As figure 5.3 illustrates, the Barren Fork Creek is very dynamic. Within ten years, sediment was deposited on the gravel bar and the riparian vegetation became fully established (see yellow arrows). Much of eroded particulate P, from both uplands and streambanks, is deposited on the floodplain or within the stream system, particularly on the non-critical bank. Since the water only overtopped its bank a few times from 2004 to 2013, most of the excess P is believed to be stored in the stream system.

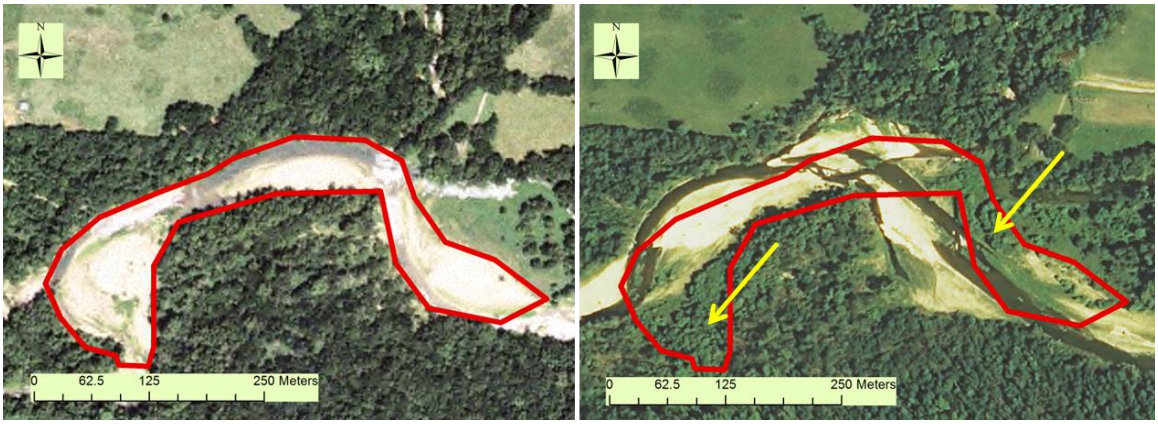

Figure 5.3. Barren Fork Creek reach illustrating the large quantity of streambank erosion and deposition that occurred from 2003 (left) to 2013 (right). Red lines illustrate the location of the gravel bar in 2003 and the yellow arrows show the newly established riparian vegetation.

A floodplain ratio, currently in the beta version of the streambank-erosion routine (Narasimhan et al., 2015), calculates the sediment and particulate P that settles on the floodplain using:

$$FP_{ratio} = \frac{area_1 - area_2 - area_3}{area_1} \tag{5.4}$$

where FP_{ratio} is a fraction of sediment and particulate P deposited in the floodplain, $area_1$ and $area_2$ are the total and top of the bank submerged cross sectional area (m^2), respectfully, and $area_3$ is the submerged cross sectional area from the top of bank to the total water depth multiplied by the top width (m^2). This equation assumes the velocity and particulate P are uniformly distributed.

The in-stream P routine scours all benthic P during large storm events, although much of the P deposited within the stream system is believed to remain stored in the stream system (Figure 5.3). Thus, in order to simulate the long-term storage of the particulate P, the in-stream P routine was modified.

Two new variables were added to the subroutine, F_{stor} and S_{max}. F_{stor} is the fraction of bankfull flow when P from the benthic pool is converted to long-term P

storage, and ranges from 0 to 1. The flow corresponding to long-term P storage, Q_{stor} in $m^3 s^{-1}$, is calculated using:

$$Q_{stor} = F_{stor} * Q_{bankfull} \tag{5.5}$$

where $Q_{bankfull}$ is the flow when the water reaches the top of the bank ($m^3 s^{-1}$). When the flow exceeds Q_{stor}, a storage ratio, S_{ratio}, is calculated using:

$$S_{ratio} = \frac{Q_{stor}}{Q} \tag{5.6}$$

where Q is the stream flow in $m^3 s^{-1}$. The quantity of P moved from the benthic P storage into the long term P storage is calculated using:

$$P_{lts} = (1 - S_{ratio}) * P_{benthic} \tag{5.7}$$

where P_{lts} is P moved to long term storage (kg), and $P_{benthic}$ in the P stored in the benthic pool (kg). Note that P in long term storage is stored indefinitely. To limit the quantity of P converted to long term storage, S_{max} is the maximum allowable S_{ratio}.

A sensitivity analysis was conducted on F_{stor} and S_{max}. Each parameter was varied from 0.25 to 1.0 and the results compared to the SWAT-predicted total P load without the new parameters, i.e. baseline conditions (Table 5.1). The greatest change occurred when both variables were at 0.25. As F_{stor} increases, more flow is required to convert P to long-term stored P. As S_{max} converges to 1.0, less P is converted to long term stored P.

Table 5.1. Sensitivity of instream-phosphorus routine proposed parameters F_{stor} and S_{max} on SWAT predicted total phosphorus load. At baseline F_{stor} and S_{max} are equal to 0.35 and 0.25, respectively.

F_{stor}	S_{max}	Total P (kg yr^{-1})	Percent Change
Baseline	Baseline	101,200	N/A
0.25	0.25	74,200	-26.7
0.50	0.25	83,500	-17.5
0.75	0.25	89,900	-11.2
1.0	0.25	93,100	-8.0
0.35	0.25	78,600	-22.3
0.35	0.50	80,100	-20.8
0.35	0.75	84,200	-16.8
0.35	1.0	101,200	0

5.3.4 SWAT Model Setup

A SWAT model for the Barren Fork Creek watershed was created similar to the Illinois River watershed SWAT model (Chapter 3). The landcover dataset, developed from 2010 and 2011 Landsat images, was used as well as the 10-m USGS DEM and SSURGO soil data. The watershed had minor point sources at Westville, Oklahoma and Lincoln, Arkansas, two United States Geological Survey (USGS) stream gages located near Eldon, Oklahoma and Dutch Mills, Arkansas and three weather stations (Figure 5.4). The two point sources contributed an average of 2.5 kg of dissolved P and 0.63 kg of particulate P daily from 2004 to 2013. Management practices, litter application rates and Soil Test Phosphorus (STP) for each subbasin were obtained from the Illinois River SWAT model. The final SWAT model consisted of 73 subbasins, 2,991 HRUs and eight land uses: forest (55%), well-managed pasture (24%), over-grazed pasture (5.8%) hay meadow (13%) and other (2.2%).

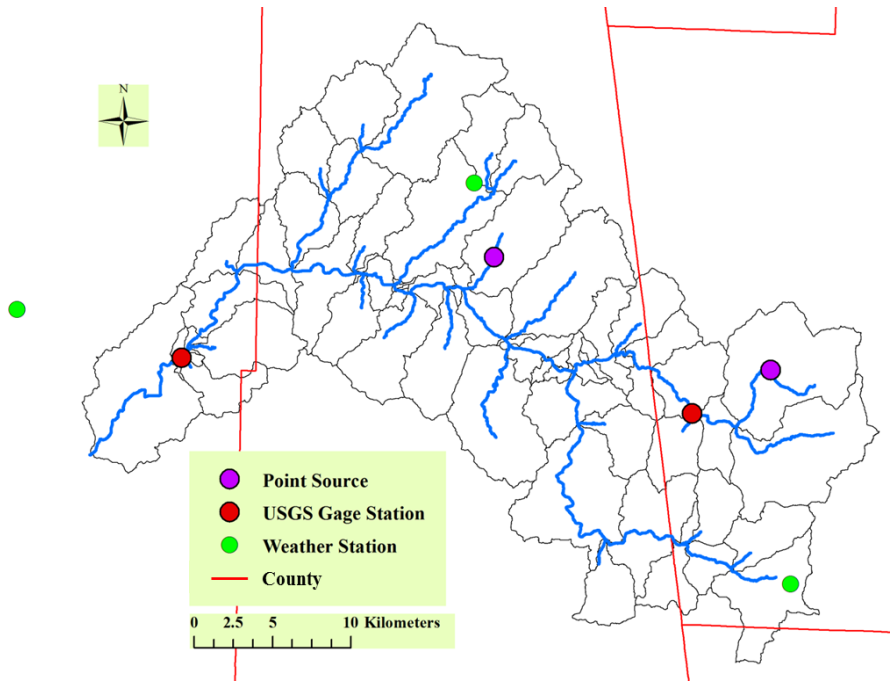

Figure 5.4. United States Geological Survey (USGS) gage stations, weather stations and point sources located in the Barren Fork Creek watershed in northeast Oklahoma and northwest Arkansas.

Of the 73 subbasins, 36 were on the Barren Fork Creek. Streambank erosion for tributaries was ignored. Data to characterize each stream reach were obtained from aerial images, topography maps, 28 cross-sectional surveys (Chapter 4) and previous studies (Miller et al., 2014; Narasimhan et al., 2015). These data included bed slope, cover factor, sinuosity, radius of curvature, top width, streambank depth, area-adjustment factor, bank composition, side slope, τ_c , k_d and total and dissolved P. For each measured parameter, the values for each reach were derived either from (1) a longitudinal trend relating the variable to watershed area or distance to confluence with the Illinois River or (2) an average from measured data.

The bed slope was measured using National Agricultural Imagery Program (NAIP) images and 1:24,000 topography maps, and used to derive the following equation:

$$BS = 4.3*10^{-9} * DA^2 - 6.7*10^{-6} * DA + 0.00369 \qquad (5.8)$$

where *BS* is the bed slope (m m^{-1}) and *DA* is the drainage area (km^2).

Previous streambank modeling results showed that riparian protection significantly impacted the quantity of erosion in the watershed (Daly et al., 2015a; Chapter 4). In Chapter 4, a channel cover factor of 2.0 for the protected sites and a channel cover factor of 1.0 for the unprotected sites were used. Since only a portion of the streambank reaches were protected, a value between 1.0 and 2.0 was assigned to each reach proportional to the percentage of riparian protection (Narasimhan et al., 2015). The critical shear stress was then modified based on the equation proposed by Julian and Torres (2006):

$$\tau_c^* = \tau_c * CH_{\text{cov}} \tag{5.9}$$

where τ_c^* is the effective critical shear stress (N m^{-2}) adjusted for vegetation and CH_{COV} is channel cover factor (Julian and Torres, 2006).

The sinuosity for each reach was calculated by measuring both the stream length and straight-line distance for each reach using NAIP images. Based on the sinuosity, SWAT divided each reach into the fraction of straight (*1/sinuosity*) and meandering (*1-*(*1/sinuosity*)) reach sections. For example, for a 100 m reach with a sinuosity of 1.5, 67% (*1/1.5*) of the reach is defined as straight, or 67 m. The remaining reach section (*1-*(*1/1.5*)) or 33 m would be defined as a meander. Streambank erosion occurs on both banks for the straight reaches, but only one bank for the meandering sections. In this example, streambank erosion would occur on both banks for 67 m of the reach and on one bank for 33 m of the reach. Effective shear stress, calculated from equations 5.2 and 5.3, is multiplied by a dimensionless bend factor, K_b, (Sin et al., 2012; Narasimhan et al., 2015) for the meandering section of each reach using:

$$K_b = 2.5 * \left(\frac{R_c}{W} \right)^{-0.32} \tag{5.10}$$

$$R_c = 1.5 * W^{1.12} \tag{5.11}$$

where R_c is the radius of curvature (m) and W is the top width (m).

Data from the cross-sectional surveys were used to estimate the W, streambank depth, side slope, area-adjustment factor and bank composition for each reach. These data were used with drainage area to derive:

$$W = 0.0765 * DA + 35.6 \tag{5.12}$$

where W is top width (m) and DA is the drainage area (km^2). Since there was no longitudinal trend, the average side slope (3.1:1) and streambank depth (2.84 m) were used for each reach. Since SWAT assumes a simple trapezoidal channel cross section, an area-adjustment factor was proposed (Chapter 4) to account for the heterogeneous cross-section given as:

$$A_{adj} = a * A \tag{5.13}$$

where A_{adj} is the adjusted channel cross-sectional area (m^2), a is a dimensionless adjustment factor less than or equal to 1.0 and A is the trapezoidal cross-sectional area. An average a of 0.78 was found for the surveyed cross sections (Chapter 4), which signifies that when flow is at the top of the bank, only 78% of the cross-sectional area is submerged**.** The percentage of gravel for each measured bank ranged from 0 to 100% with an average of 62% gravel and 38% cohesive (Figure 5.5).

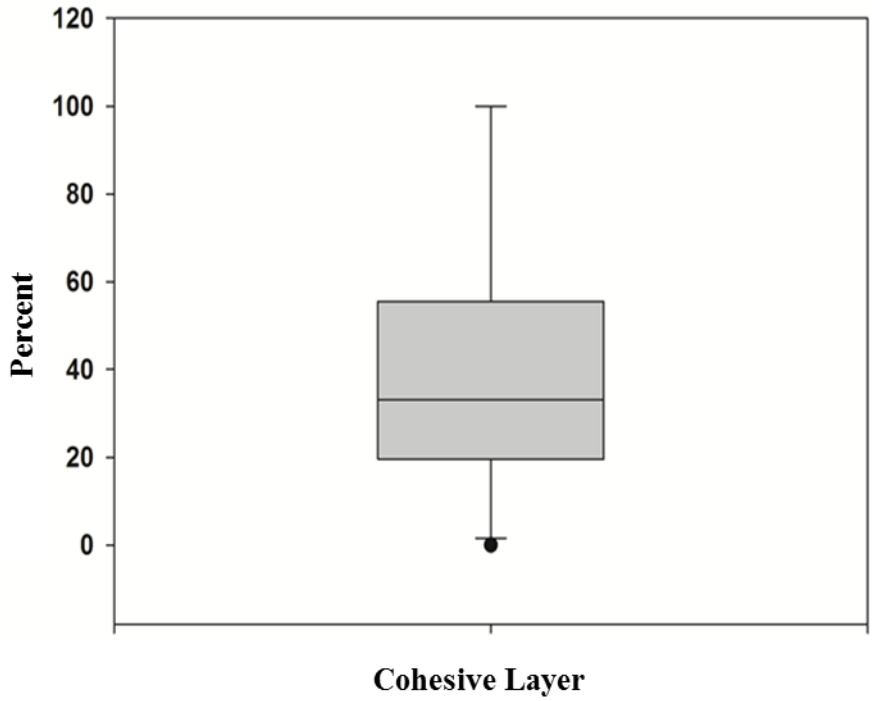

Figure 5.5. Percent cohesive layer for each of the surveyed banks.

Streambank data obtained from Miller et al. (2014) included τ_c and total and water soluble P for the soil. There was no longitudinal trend relating the τ_c with the *DA*. Therefore, an average τ_c of 5.6 Pa, a function of the measured d_{50}, was used. The k_d was calculated based on the k_d to τ_c relationship proposed by Hanson and Simon (2001):

$$k_d = 0.2 * \tau_c^{-0.5} \tag{5.14}$$

Although equation 5.14 was derived using cohesive soils, the equation was successfully used for gravel layers at similar sites by Daly et al. (2015a) and Midgley et al. (2012) and thus will be used in this study. Total P concentrations for the streambanks from Miller et al. (2014) ranged from 250 to 427 mg P kg^{-1} soil, which were similar to Tufekcioglu (2010) (246 to 349 mg P kg^{-1} soil) and Zaimes et al. (2008) (360 to 555 mg P kg^{-1} soil). Water soluble P concentrations ranged from 1.2 to 8.1 mg P kg^{-1} soil. Total and water-

soluble P for the streambank soil was obtained using the regression equations 5.14 and 5.15:

$$TP = 1.7546 * d + 249.49 \tag{5.15}$$

$$WSP = 0.1121 * d + 0.3278 \tag{5.16}$$

where *TP* and *WSP* are the total and water soluble P in the streambank (mg P kg^{-1} soil) and *d* is the distance from the confluence of the Illinois River (km) (Figure 5.6). The P concentrations are higher upstream, believed to be a result of the higher density of poultry houses in Arkansas. The quantity of P eroded was adjusted based on the percentage of the bank containing cohesive soil, since gravel was assumed to not contain P.

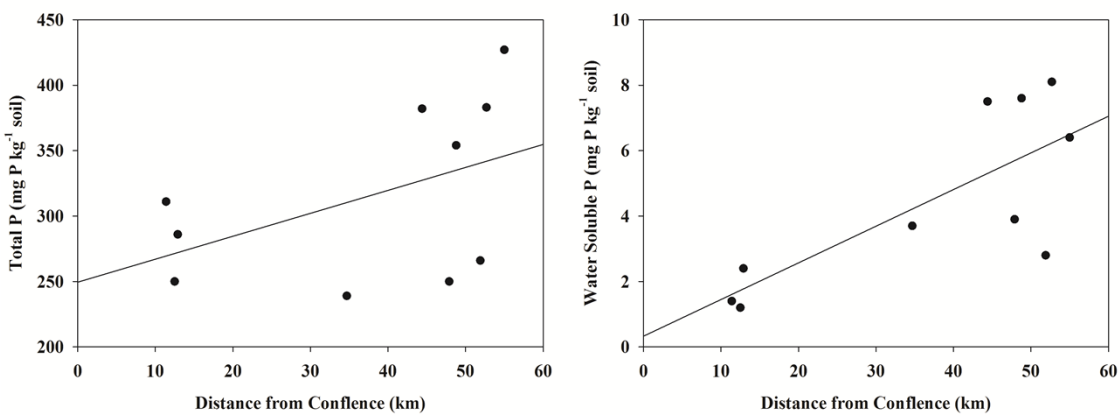

Figure 5.6. Total and water soluble phosphorus (P) concentrations for streambanks with distance from the Barren Fork Creek to the confluence with the Illinois River in Oklahoma.

5.3.5 Model Evaluation

5.3.5.1 Streamflow

SWAT was manually calibrated for monthly baseflow, peak flow and total flow at the USGS gage stations 07197000 and 07196900. A sensitivity analysis was conducted on eleven parameters based on previously used calibration parameters and SWAT documentation (Neitsch et al., 2009). Parameters were adjusted within the SWAT

recommended range. Their sensitivity was calculated and used to determine the influence each parameter had on peak flow and baseflow. The streamflow was calibrated and validated from 2004 to 2013 and 1995 to 2003, respectively. The USGS Hydrograph separation program (HYSEP) was used to estimate baseflow (Sloto and Crouse, 1996). Coefficient of Determination, R^2, and Nash-Sutcliffe Efficiency (NSE) were used to evaluate the model's performance (Moriasi et al., 2007). Model performance ratings for NSE for total monthly flow were the following: Very good >0.75, Good 0.65-0.75, Satisfactory 0.50-0.65, Unsatisfactory <0.50 (Moriasi et al., 2007).

5.3.5.2 Phosphorus

The SWAT in-stream P routine was calibrated and validated on a monthly time step from 2009 to 2013 and 2004 to 2008, respectfully, at the USGS gage station 07197000. The USGS gage station 07196900 was not used due to poor LOADEST results (Miller et al., 2014). R^2 and NSE were used to evaluate model performance. Note that the model was calibrated prior to and after the incorporation of the streambank erosion.

5.3.5.3 Streambank Erosion

Using a method by Heeren et al. (2012) and Miller et al. (2014), streambank erosion was measured using 2003 and 2013 NAIP images for each of the 36 SWAT defined reaches on the Barren Fork Creek (Figure 5.7). The NAIP images were used to estimate the average eroded width and length and then used to calculate the eroded area (*AE*, m^2). The total sediment loading (*TS, kg*) from each reach was calculated using:

$$TS = EA * D_{ts} * \rho_b \tag{5.17}$$

where D_{ts} is the streambank depth (m) from Miller et al. (2014) and Chapter 4, and ρ_b is the soil bulk density (g cm^{-3}). A weighted ρ_b based on the bank composition (Miller et al., 2014) was used to estimate the average ρ_b for the bank.

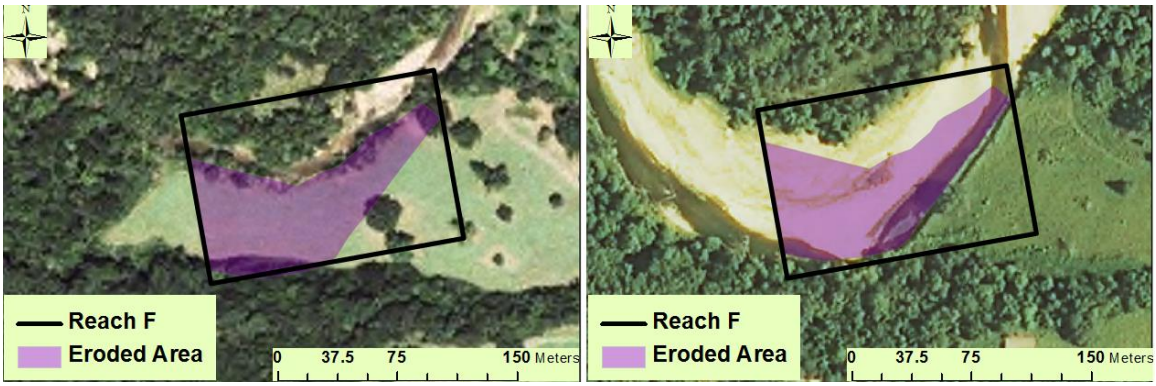

Figure 5.7. National Agricultural Imagery Program (NAIP) aerial images for 2013 (left) and 2013 (right) with polygons showing the bank retreat (purple) during the period.

5.4 Results and Discussion

5.4.1 Streamflow

During calibration, six parameters were modified to obtain the best goodness-of-fit statistics for each gage station (Table 5.2). SWAT predictions at USGS gage station 07197000 were 'very good' (Moriasi et al., 2007) for the calibration and validation periods, with NSE of 0.82 and 0.78, respectfully. R^2 for the calibration and validation periods were 0.82 and 0.80, respectfully. At the upstream USGS gage station (0719690), calibration and validation predictions 'good' (Moriasi et al., 2007) based on the NSE of 0.72 and 0.70 for the calibration and validation periods, respectfully. R^2 for the calibration and validation periods were 0.72 and 0.71, respectfully.

Table 5.2. SWAT default and calibrated parameter estimates used to calibrate flow on the Barren Fork Creek watershed SWAT model.

Original Value or Range	Calibrated Value or Range	Parameter	Description
0.95	0.85	ESCO	Soil evaporation compensation coefficient
0.05	0.25	RCHRG_DP	Aquifer percolation coefficient
0.048	0.75	ALPHA_BF	Baseflow Alpha Factor (Days)
39-94	-4	CN2	SCS curve number adjustment
0	10	CH_K2	Effective hydraulic conductivity in main channel alluvium (mm hr^{-1})
0.5	105	CH_K1	Effective hydraulic conductivity in tributary channel alluvium (mm hr^{-1})
0.014	0.05	Manning's n	Manning's 'n' in main channel

5.4.2 Total Phosphorus without Streambank Erosion

Each of the in-stream P parameters was manually adjusted during P calibration (Table 5.3). Overall the model performed exceptionally well predicting total P, except for some of the peaks loads (Figure 5.8). During the calibration process, any attempt to increase the predicted total P for the peaks resulted in an over prediction for a number of smaller events (Figure 5.8, see arrows). For the 2009 to 2013 calibration period, the R^2 was 0.82 and the NSE 0.60. The lower NSE was due to the under prediction of the total P during the large storm event in April 2011. The R^2 and NSE for the 2004 to 2008 validation period was 0.80 and 0.77, respectively. The predicted average annual P load from 2004 to 2013 originating from the uplands was 53.9 Mg yr^{-1}, with 42% from well-managed pasture, 32% from overgrazed pasture, 21% from hay meadows and 5.6% from forest.

Table 5.3. SWAT default and calibrated in-stream phosphorus (P) model parameter estimates for the Barren Fork Creek watershed SWAT model.

Parameter	Default Value	Calibrated Value	Description
DI	250	90	Period of influence (d)
K_{in}	0.10	0.15	Soluble P transformation into the benthic sediment (hr^{-1})
K_{out}	0.10	0.001	Soluble P transformation out of the benthic sediment (hr^{-1})
F_{dep}	0.01	0.01	Fraction of bankfull discharge at which there is 100% deposition
F_{eq}	0.15	0.26	Fraction of bankfull discharge at which scour and deposition of particulate P is at equilibrium
F_{scr}	0.80	0.75	Fraction of bankfull discharge at which all P is scoured from the streambed
SPT	0.01	0.001	Soluble to particulate transformation coefficient

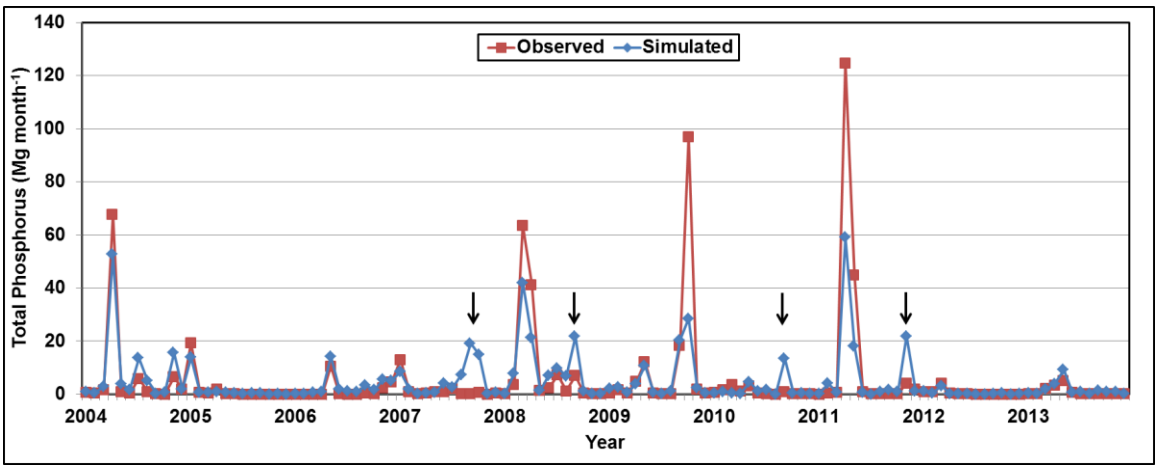

Figure 5.8. Time series illustrating monthly SWAT predicted and observed total phosphorus (P) load from 2004 to 2013 at the United States Geological Survey gage station 07197000 on the Barren Fork Creek. Black arrows indicate storm events where the SWAT model over predicted P.

5.4.3 Streambank Erosion

The measured streambank erosion for the Barren Fork Creek from 2003 to 2013 was 160,000 Mg yr^{-1}. The reach-weighted streambank erosion was 42 kg m^{-1} compared to 34 kg $m^{-1}$for Spavinaw Creek (Purvis, 2015), approximately 60 km north of the Barren Fork Creek. The Barren Fork Creek streambank erosion increased further downstream as reaches approached the confluence of the Illinois River. For example, the average erosion 0 to 25 km from the confluence with the Illinois River was 78 kg m^{-1}, compared to 28 kg m^{-1} 25 to 65 km from the confluence. Therefore, future streambank stabilization projects should focus their efforts on the lower 25 km of the creek.

The uncalibrated cover factors for the 36 reaches ranged from 1.0 to 2.0 with an average of 1.6 (Figure 5.9). Using these cover factors, the uncalibrated SWAT predictions compared to measured streambank erosion resulted in an R^2 and NSE of 0.36 and 0.33, respectively (Figure 5.10). SWAT simulated mass of eroded soil was 215,000 Mg yr^{-1} or a reach-weighted 40 kg m^{-1} from 2004 to 2013, which compares to the measured erosion of 160,000 Mg yr^{-1} or 42 kg m^{-1}. Some of this over prediction was due to assumptions in estimating the streambank-erosion parameters and failing to account for the armored banks. From personal observations, approximately 5% of the banks are armored, with the majority located in the head waters of the Barren Fork Creek. Armored banks, with a k_d of 0.0 cm^3 N^{-1} s^{-1}, would reduce the simulated erosion by approximately 10,800 Mg yr^{-1} and the relative error for the measured versus simulated erosion from 34 to 27%.

SWAT-predicted streambank erosion was then calibrated by adjusting the cover factor, which modified τ_c and k_d. The average calibrated cover factor was 1.9 (Figure 5.9), which equates to τ_c of 11 Pa and k_d of 0.06 cm^3 N^{-1} s^{-1}.

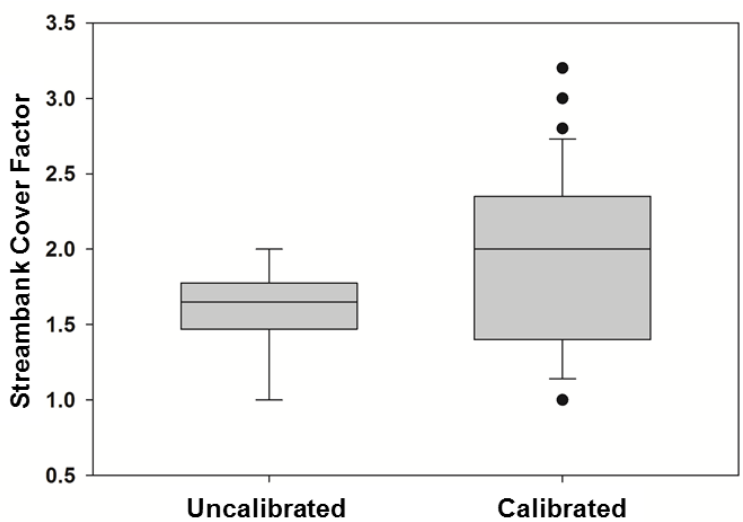

Figure 5.9. Uncalibrated and calibrated cover factors for the 36 reaches on the Barren Fork Creek.

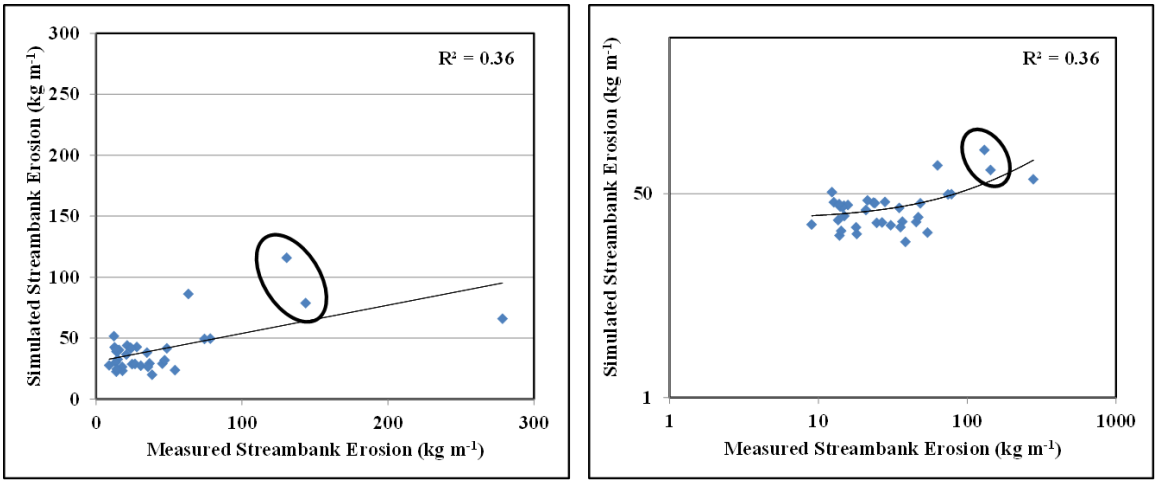

Figure 5.10. Measured vs uncalibrated SWAT streambank erosion predictions for the Barren Fork Creek from 2004 to 2013 on linear (left) and log (right) scales. The two circled points are two of the ten study sites from Miller et al. (2014), which were two of the most erosive reaches of the SWAT-defined 36 reaches on the Barren Fork Creek.

5.4.4 Total Phosphorus with Streambank Erosion

The calibrated streambank erosion contributed a total of 48 Mg yr^{-1} of total P from 2004 to 2013, which is approximately half the total P estimated by Miller et al. (2014). The higher estimate by Miller et al. (2014) was likely due to the ten study sites not being representative of the entire creek. Two of their study sites had the second and third most erosion per length of stream (see ovals in Figure 5.10). The total P from the combined uplands and Barren Fork Creek streambanks from 2004 to 2013 was 103 Mg yr^{-1}, of which 47% originated from streambanks. Langendoen et al. (2012) found that 36% P entering Missisquoi Bay was from streambank erosion. Streambanks in Denmark contributed 21 to 62% of the annual P loads (Kronvang et al., 2012). This study supports other studies around the world that P derived from streambank erosion can be a significant source of P in a watershed. It should be noted that while the quantity of particulate P from streambank erosion exceeded the particulate P from the upland area, the majority of the dissolved P originated from the upland areas. The dissolved P, which

is easily accessible to aquatic plants, is more important to water quality than the tightly-bound particulate P. In addition, the two point sources contributed a small percent of the total P in the watershed (Figure 5.11).

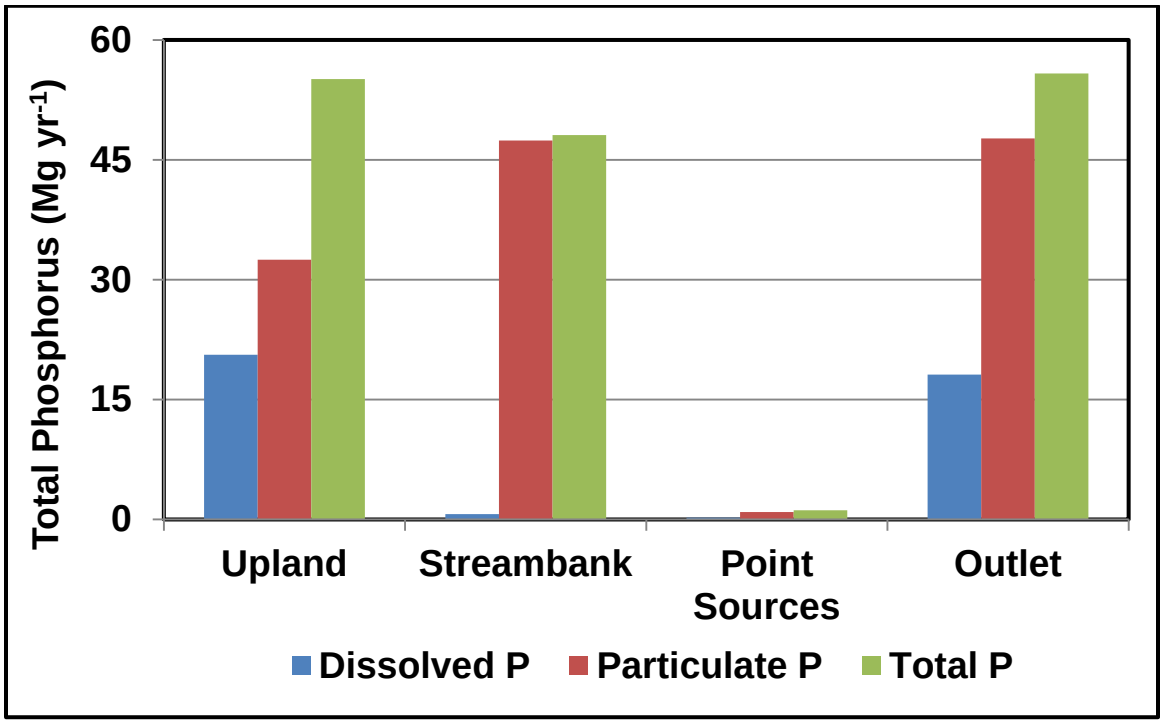

Figure 5.11. Average annual total phosphorus (P) contributions from the Barren Fork Creek watershed upland areas, streambank and point sources compared to the total P load reaching the outlet.

After incorporating streambank-derived P into the SWAT model, the two proposed in-stream P routine variables were calibrated. F_{stor} was calibrated to 0.35 and S_{max} was calibrated to 0.25. If bankfull flow is 1000 m^3 s^{-1}, for example, P will be converted into long-term storage when flow exceeds 350 m^3 s^{-1}. At a flow of 7000 m^3 s^{-1}, 95% of the benthic P is converted to long-term storage. However, S_{max} limits the P converted to long-term storage to 75%.

P calibration improved with streambank erosion compared to without streambank erosion (Table 5.4). The R^2 and NSE improved for both the calibration and validation periods except for the R^2 for the calibration period, which was due to over predicting

streamflow and P load in November 2011. The relative errors for total, dissolved and particulate P were all less than 6% for both the calibration and validation periods (Table 5.5). The inclusion of P from streambank erosion also improved the prediction of particulate and total P during large storm events (Figure 5.12), with most of the peaks comparing favorably with observed loads, except for the large storm in 2009.

Table 5.4. Calibration and validation statistics for SWAT predicted total phosphorus load with and without streambank erosion. NSE is Nash Sutcliff Efficiency.

	Total Phosphorus (kg yr^{-1})	Error (%)	Dissolved Phosphorus (kg^{-1})	Error (%)	Particulate Phosphorus (kg^{-1})	Error (%)
Calibration						
Observed	59,500		16,800		42,700	
Simulated	60,000	0.84	16,700	-0.60	43,200	1.2
Validation						
Observed	55,800		18,100		37,700	
Simulated	57,800	3.6	19,100	5.5	38,700	2.7

Table 5.5. Observed and simulated total, dissolved and particulate phosphorus and their relative errors for the calibration (2009 to 2013) and validation periods (2004 to 2008) with and without streambank erosion.

Statistic	Without Streambank Erosion		With Streambank Erosion	
	Calibration	Validation	Calibration	Validation
R^2	0.82	0.80	0.80	0.95
NSE	0.60	0.77	0.78	0.95

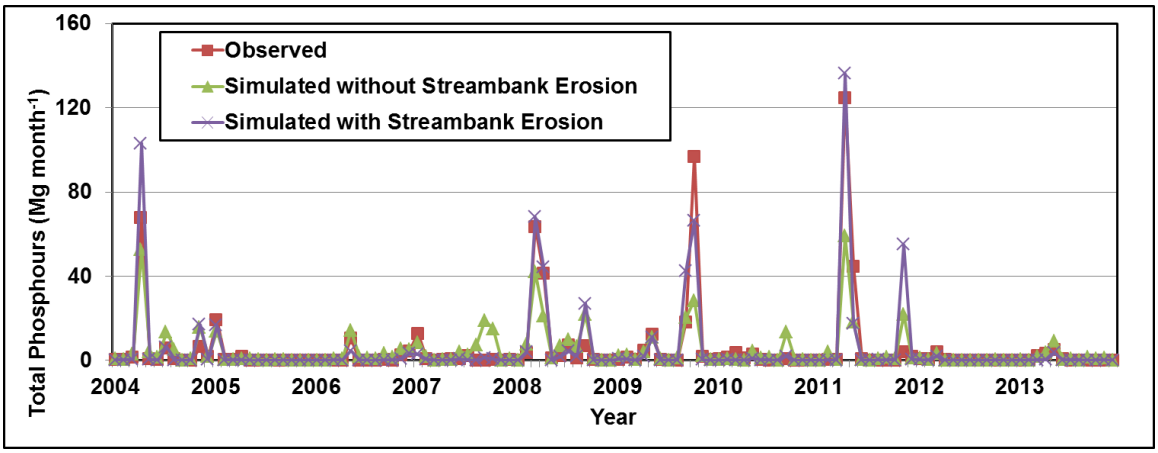

Figure 5.12. Monthly SWAT time series for observed and predicted total phosphorus load from 2004 to 2013 for the Barren Fork Creek watershed with and without streambank erosion.

From 2004 to 2013, approximately 103 Mg yr^{-1} of P entered the Barren Fork Creek from the upland areas, streambank erosion and point sources. Of this total, over 35 Mg yr^{-1} (39 kg yr^{-1} km^{-2}) was converted to long-term storage (Figure 5.13). The P mass balance study by Mittelstet (Chapter 2) estimated the total P stored in the Illinois River watershed at 7.7 to 290 kg yr^{-1} km^{-2} during the period of 1925 to 2015. Based on the results of this study, the total P stored in the Illinois River stream system is probably closer to 7.7 than 290 kg yr^{-1} km^{-2}. During this same time period, 75 kg yr^{-1} km^{-2} of total P left via the watershed outlet to the Illinois River and 1.7 kg yr^{-1} was deposited on the floodplain. A large quantity of P from the benthos was scoured and converted to long-term stored P in 2004. Therefore, the net P added to the benthos was -1.7 kg yr^{-1} km^{-2}. Of the total quantity of P added to the system, approximately 65% left via the outlet and 35% was stored in the stream system and floodplain.

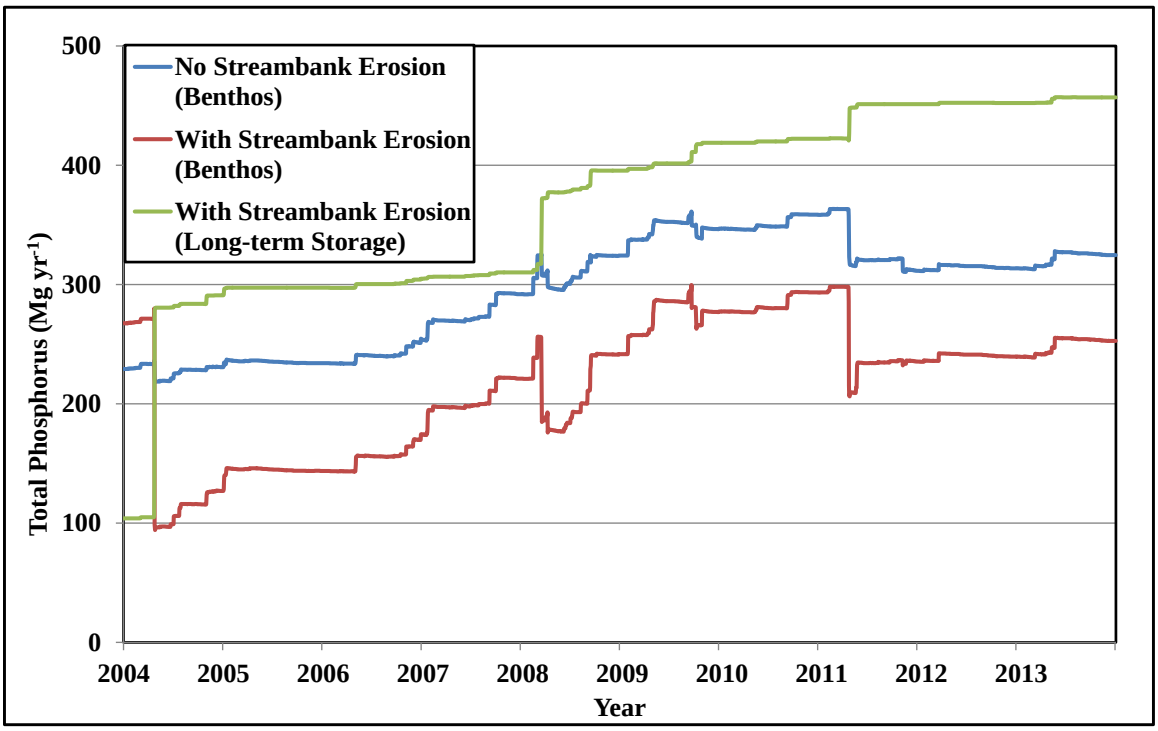

Figure 5.13. Total phosphorus stored in the benthos and long-term storage for SWAT predictions of the Barren Fork Creek from 2004 to 2013 with and without streambank erosion.

5.5 Conclusions

The modified streambank-erosion routine, with the process-based applied shear stress equation and the area-adjustment factor, was applied to the Barren Fork Creek. Uncalibrated, the average reach-weighted predicted streambank erosion from 2004 to 2013 was 40 kg m^{-1} compared to the measured 42 kg m^{-1}. Over 100 Mg of P was added to the Barren Fork Creek annually from 2004 to 2013, of which 47% was from streambank erosion. Due to this influx of streambank P to the system and the current in-stream P routine's limitations, the in-stream P routine was modified by introducing a long-term storage coefficient. This long-term storage coefficient converted particulate P to long-term storage as a function of flow. P calibration with the proposed long-term storage coefficient improved P calibration results, especially for peak flow events. Of the total quantity of P added to the system from 2004 to 2013, approximately 65% left via the watershed outlet and 35% was stored in the stream system and floodplain. This accumulation of P in the stream system, or legacy P, will likely be a source of P for several years or even decades.

The modified SWAT streambank-erosion routine produced reasonable estimates of streambank erosion. Incorporating particulate P from the streambank erosion can improve SWAT predicted P loads. Streambank erosion can be a significant contributor of P at a watershed scale and thus should be considered when addressing water quality in watershed management plans. For watersheds around the world with dynamic and eroding streambanks with elevated P, the modified streambank erosion and in-stream P routines can be used to improve modeling results and provide watershed managers a

better understanding of the significance of both streambank erosion and streambank P in the watershed.

CHAPTER VI

CONCLUSIONS

6.1 Conclusions

The overall objectives of this research were to (1) develop a method to quantify and identify the location of legacy P using a mass balance approach and uncertainty analysis, (2) use the SWAT model to determine land use and conservation practices needed to meet numeric water-quality standards, (3) improve and test the proposed streambank-erosion routine for the SWAT model and (4) apply the improved streambank-erosion routine to determine the significance of P derived from streambank erosion and its influence on P modeling results. The following are conclusions based on the results of this dissertation.

- A method, which is easily transferred to other watersheds, was developed to quantify and identify legacy P using a mass balance and uncertainty analysis.
- The mass balance approach and uncertainty analysis were successfully applied to the Eucha-Spavinaw and Illinois River watersheds.
- The P accumulation from 1925 to 2015 was 3.8 to 5.8 kg ha^{-1} yr^{-1} in the Illinois River watershed and 3.6 to 5.0 kg ha^{-1} yr^{-1} in the Eucha-Spavinaw watershed.

- Watershed-based plans must consider legacy P sources, such as STP, streambanks and floodplains, when proposing or developing conservation practices.
- Incorporating uncertainty analysis gives watershed managers more confidence in P mass balance results.
- From 1925 to 2015, only 14 to 19% of all P imported to the Illinois River and Eucha-Spavinaw watersheds was removed via reservoir discharge, poultry litter exports and food exports.
- To decrease uncertainty in the mass balance approach, data collection efforts should focus on quantifying P contributions from the largest imports and exports.
- Using the SWAT model, a method was developed to determine if a reservoir water-quality standard was exceeded.
- The SWAT model was successfully used to predict the necessary land-use changes and conservation practices required to meet numeric water-quality standards for the Flint Creek, Barren Fork Creek and Illinois River subwatersheds.
- Meeting the numeric water-quality standard was primarily related to land use, STP and poultry-house density.
- SWAT was able to simulate streambank erosion accurately for composite streambanks in the Barren Fork Creek watershed.
- The empirical shear-stress equation, currently used in the streambank-erosion routine for the SWAT model, was replaced by a process-based shear stress

equation and improved streambank-erosion predictions for both literature- and field-based simulations.

- Although the process-based applied shear stress equation was the most influential modification, incorporating the top width, streambank depth and area-adjustment factor improved predictions and more accurately represented the measured irregular cross-sections.
- Accurate modeling results can be obtained by incorporating the literature-based stream system parameters bed slope, sinuosity and radius of curvature.
- US SWAT users should use the regional bankfull equations to estimate bankfull width and depth and not the US regression equations.
- To improve prediction accuracy, bed slope should be measured using aerial images and topography maps. These measured values can be used to derive an equation for bed slope, which can replace the DEM-estimated bed slope values that SWAT calculates by default.
- Radius of curvature and sinuosity should be incorporated into the SWAT model as they improve SWAT modeling predictions. While sinuosity must be measured using aerial images, the equation used by the streambank-erosion routine to estimate radius of curvature yielded reasonable estimates for the Barren Fork Creek watershed.
- The un-calibrated SWAT model predicted streambank erosion with a relative error of 34% for the Barren Fork Creek watershed.

- With minimal modifications to the channel cover factor, the streambank-erosion routine predictions agreed favorably with observed data for the Barren Fork Creek watershed.
- Incorporating streambank P into the SWAT model improved model predictions, especially during large flow events for the Barren Fork Creek watershed.
- P from streambanks contributed 47% of the annual P load in the Barren Fork Creek watershed.
- The in-stream P routine was successfully modified to account for long-term stored P in the stream system.

6.2 Recommendations for Future Research

Though we can use STP and US Geological Survey flow and P data to estimate P retained in soils and reservoirs, future work needs to better quantify P stored in the stream system. Quantifying P stored in the floodplains, streambanks and stream sediment is the missing piece of the P mass balance. Multiple soil samples need to be taken in each of these areas of storage and the spatial distribution of the STP analyzed. Long-term studies need to be conducted to get better estimates of the scouring and deposition of both sediment and P during large storm events. The retention time of these sources and their transport through the stream system needs to be better understood in order to provide watershed managers and policy makers with the best location and most cost-efficient conservation practices to implement. This will aid in both P modeling and to identify potential conservation practices.

Levels of P in atmospheric deposition vary significantly around the world. Future work needs to better quantify P from atmospheric deposition at the watershed scale.

Levels of P in atmospheric deposition need to be analyzed both spatially, as well as temporally. The origin of the deposition also needs to be analyzed to determine the net atmospheric deposition and if it is a significant source of P in the watershed.

Elevated STP is an issue worldwide. P in both soils and oceans is increasing worldwide and it takes decades, if not centuries, for elevated STP to decline to manageable levels. Future work needs to study possible management and stabilization practices to reduce runoff from field sites with elevated STP. Efficient methods need to be developed to remove and transport P from high-nutrient soils to P-deficient soils.

There is uncertainty in both critical shear stress (τ_c) and erodibility coefficient (k_d). Further testing of the ability to predict τ_c using the silt and clay content is needed as well as exploring other τ_c - k_d relationships, such as the relationship proposed by Simon et al. (2011). The type of vegetation and root density along streambanks varies significantly. More research is needed to quantify how root density from different types of riparian vegetation and soil types impact τ_c and k_d.

Future research projects need to test the modified streambank erosion and in-stream P routines in other watersheds. While each of these modified routines improved modeling efforts in the Illinois River and Eucha-Spavinaw watersheds, they need to be applied to watersheds that vary from the Ozark Highland ecoregion gravel-bed streams. The most important improvements that need to be made to the SWAT streambank-erosion routine are the incorporation of multiple bank layers and modification of channel dimensions throughout the simulation.

CHAPTER VII

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APPENDICES

BARREN FORK CREEK CROSS SECTIONS

A total of 28 cross-sections were surveyed on the Barren Fork Creek using a laser level, measuring tape and survey rod: eight at cross-over points, nine at meanders and eleven at straight cross sections (Figure A.1). This data was then used to derive regression equations or averages for each of the streambank parameters used in the SWAT model.

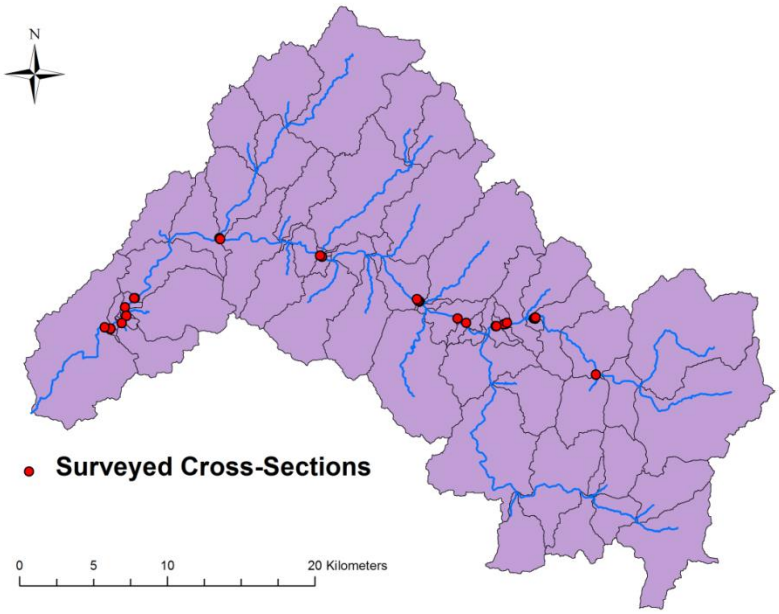

Figure A.1. Locations of the 28 cross sections surveyed on the Barren Fork Creek.

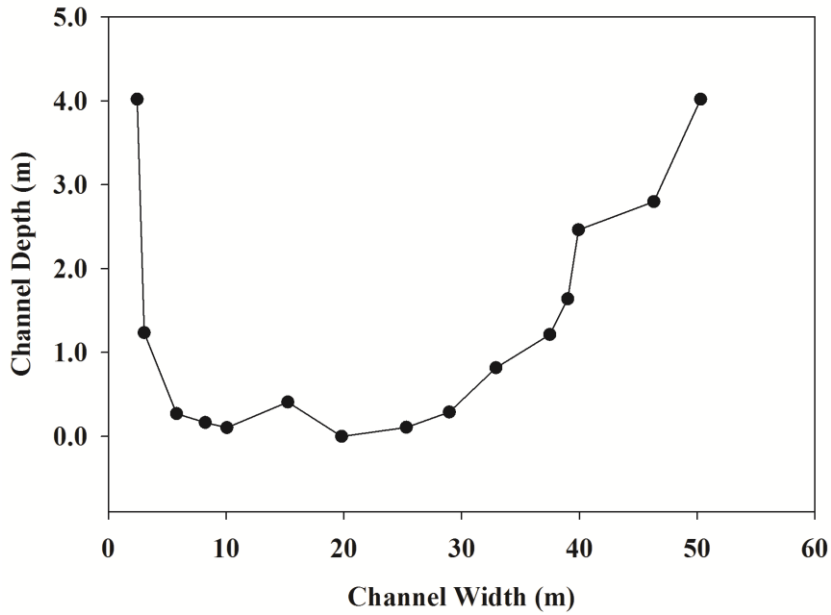

Figure A.2. Cross-sectional survey located on a straight reach at the U.S. Geological Survey gage station near Dutch Mills, Arkansas (365480 N, 3971663 E) on the Barren Fork Creek.

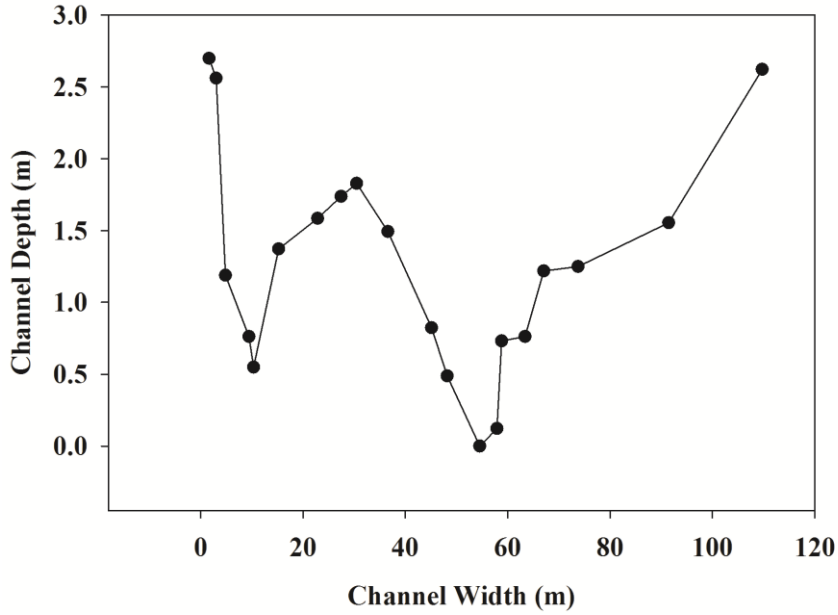

Figure A.3. Cross-sectional survey located on a straight reach at 361417 N, 3975506 E on the Barren Fork Creek.

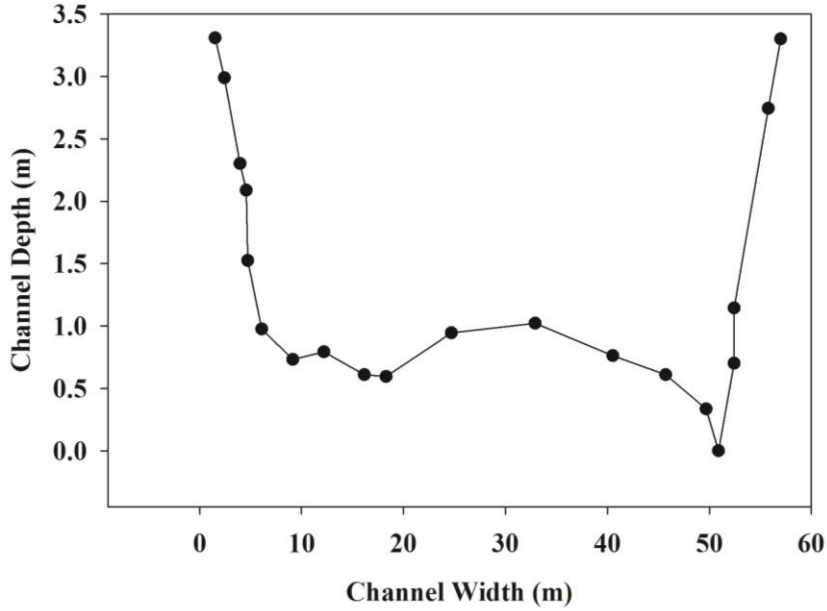

Figure A.4. Cross-sectional survey located at a cross-over at 361364 N, 3975435 E on the Barren Fork Creek.

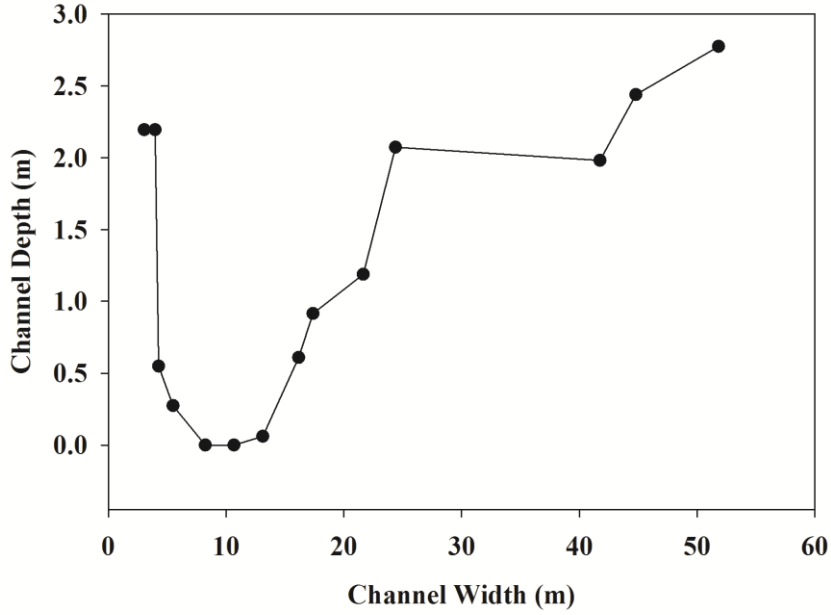

Figure A.5. Cross-sectional survey located on a meander at 361272 N, 3975458 E on the Barren Fork Creek.

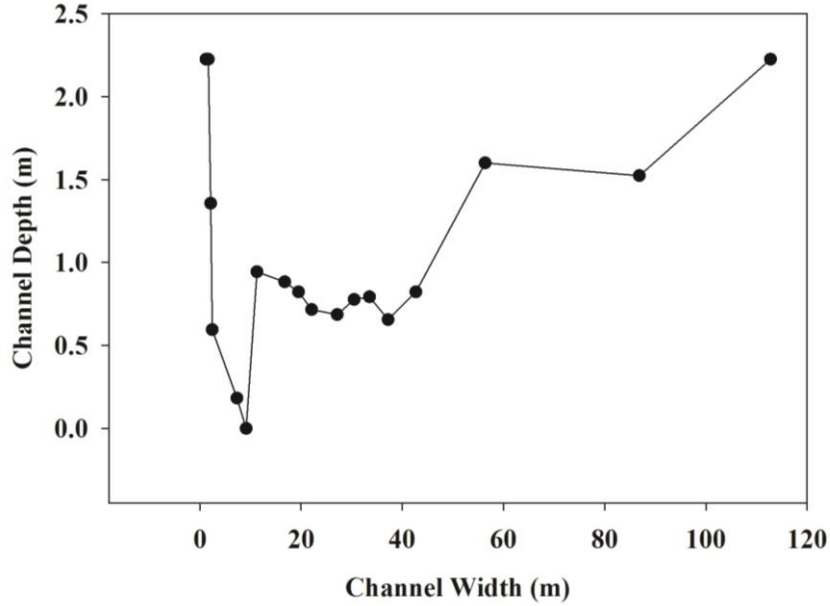

Figure A.6. Cross-sectional survey located at a cross-over at 359447 N, 3975165 E on the Barren Fork Creek.

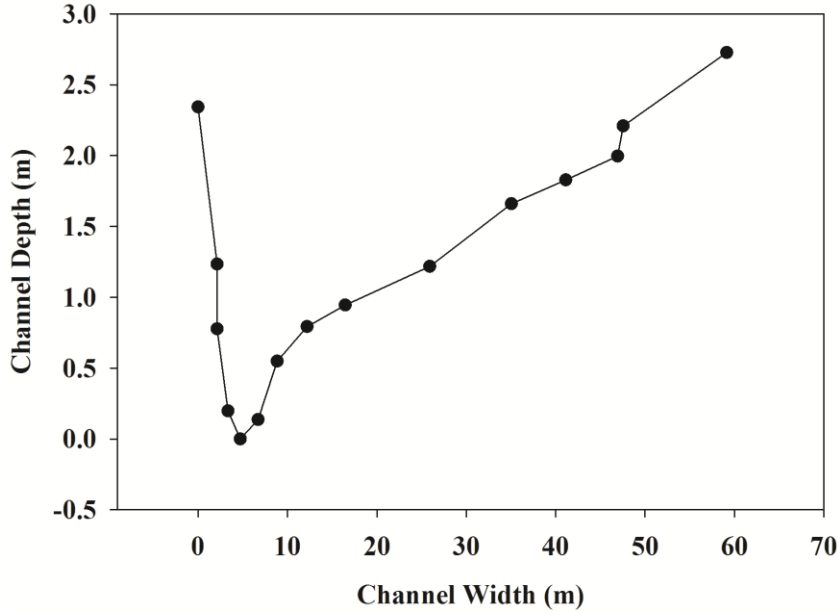

Figure A.7. Cross-sectional survey located on a meander at 3594405 N, 3975097 E on the Barren Fork Creek.

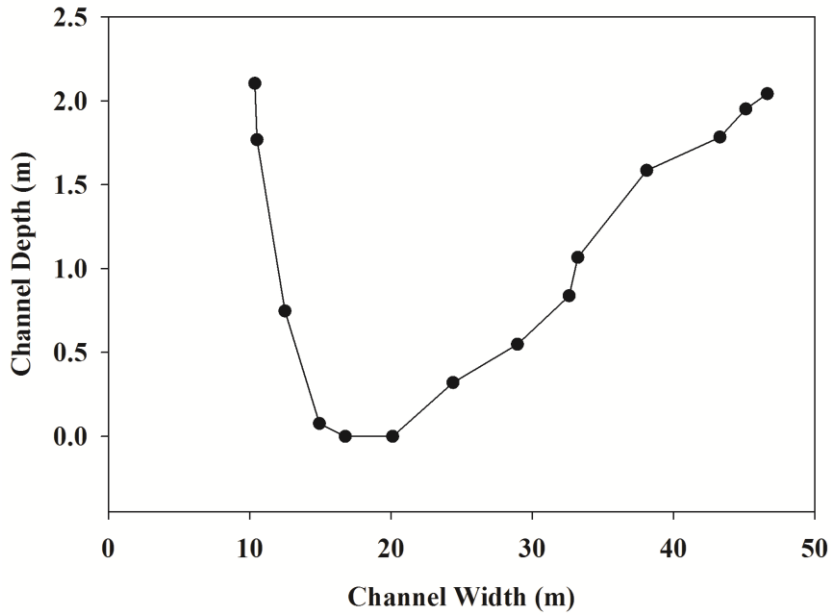

Figure A.8. Cross-sectional survey located on a straight reach at 359273 N, 3975070 E on the Barren Fork Creek.

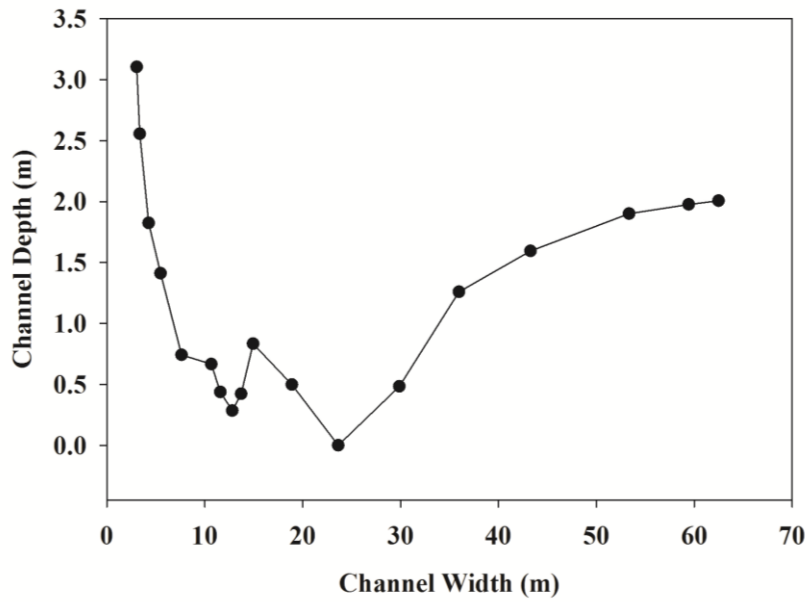

Figure A.9. Cross-sectional survey located at a cross-over at 358773 N, 3974947 E on the Barren Fork Creek.

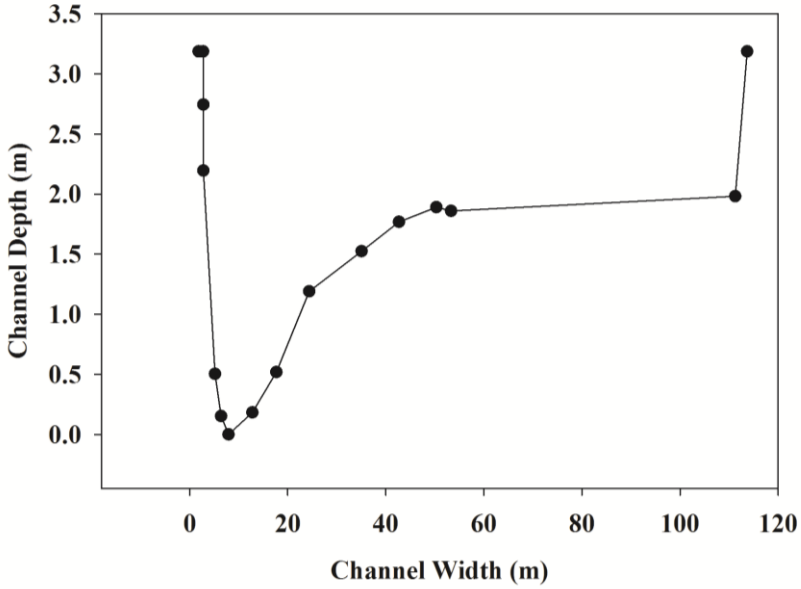

Figure A.10. Cross-sectional survey located on a meander at 358705 N, 3974940 E on the Barren Fork Creek.

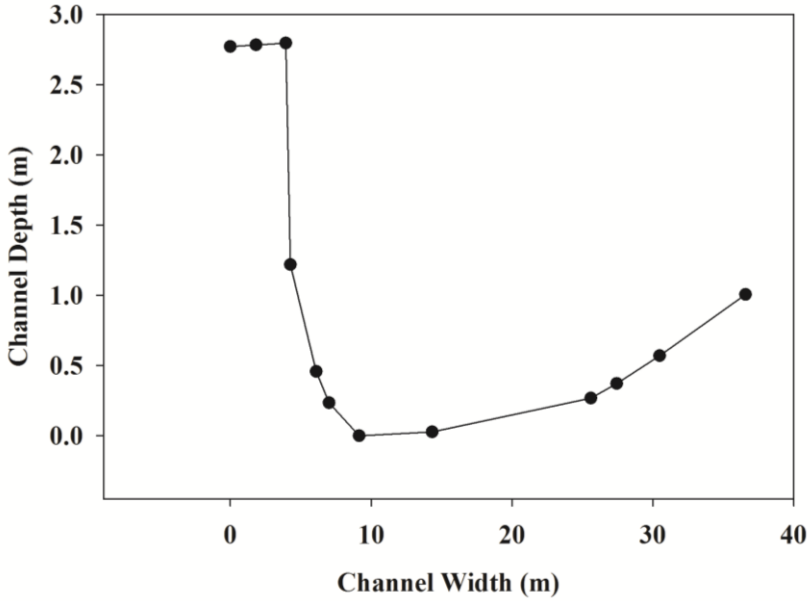

Figure A.11. Cross-sectional survey located on a meander at 356712 N, 3975175 E on the Barren Fork Creek.

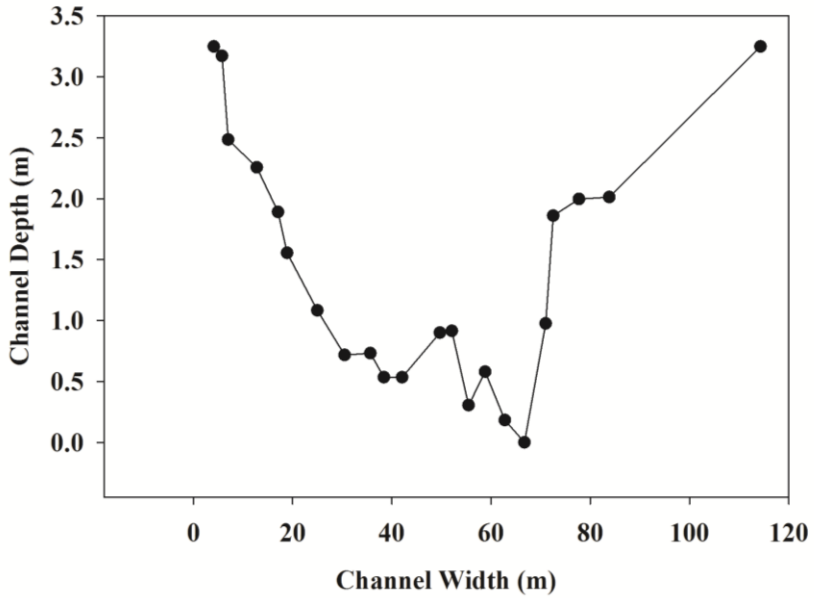

Figure A.12. Cross-sectional survey located at a cross-over at 353555 N, 3976619 E on the Barren Fork Creek.

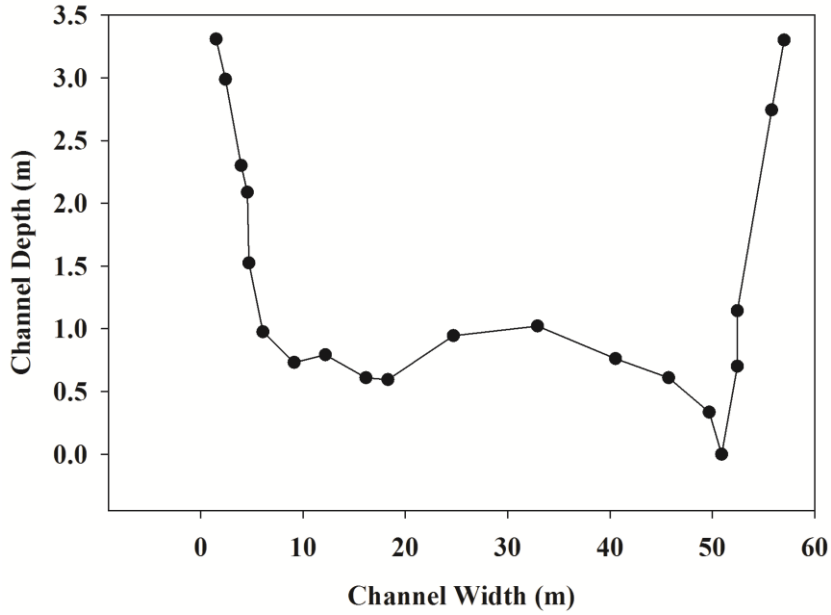

Figure A.13. Cross-sectional survey located on a straight reach at 353469 N, 3976687 E on the Barren Fork Creek.

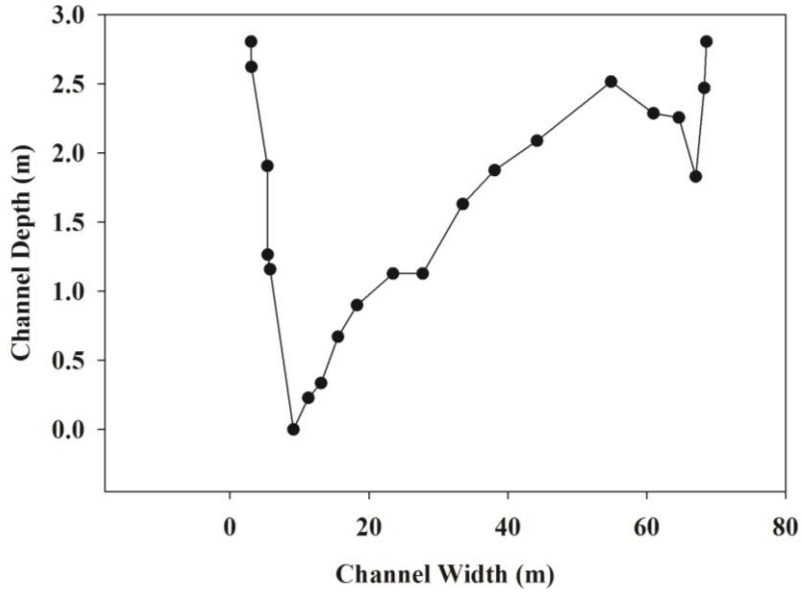

Figure A.14. Cross-sectional survey located on a meander at 353356 N, 3976777 E on the Barren Fork Creek.

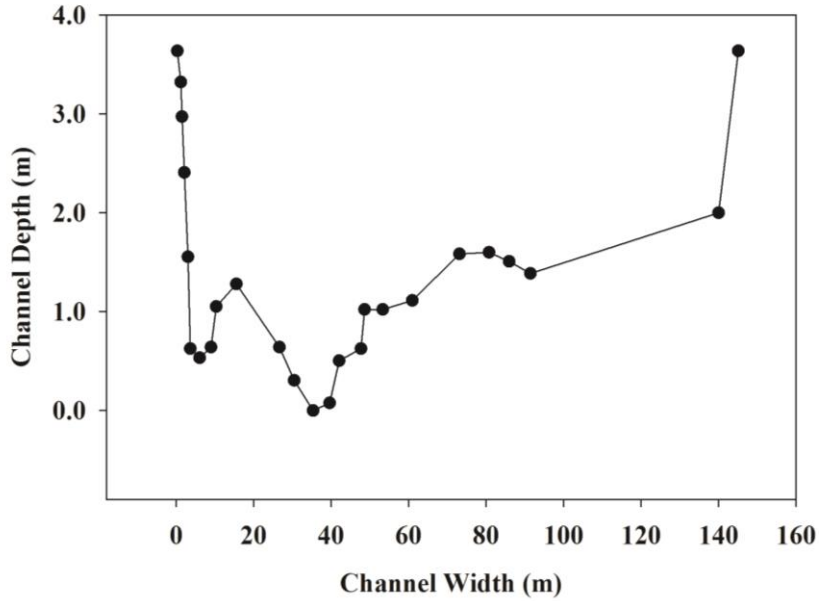

Figure A.15. Cross-sectional survey located on a cross-over at 346927 N, 3979630 E on the Barren Fork Creek.

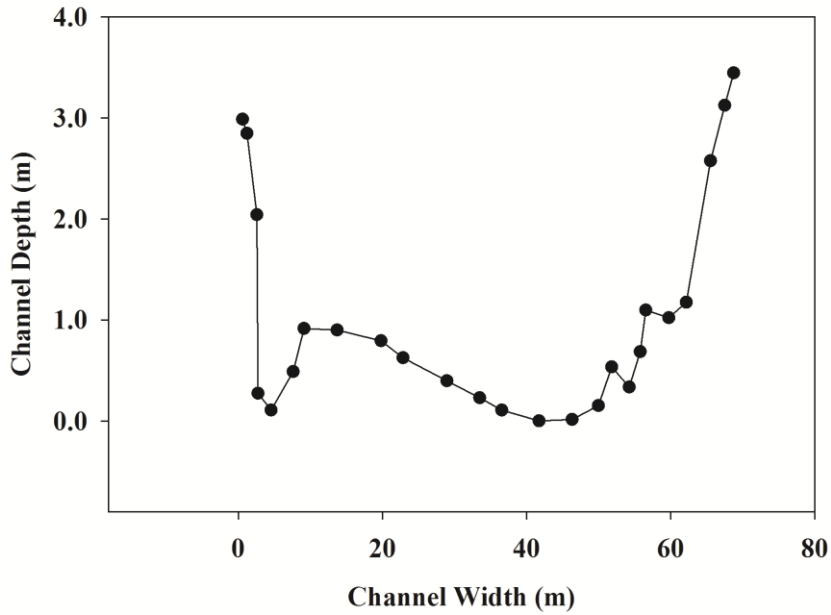

Figure A.16. Cross-sectional survey located on a straight reach at 346884 N, 3979651 E on the Barren Fork Creek.

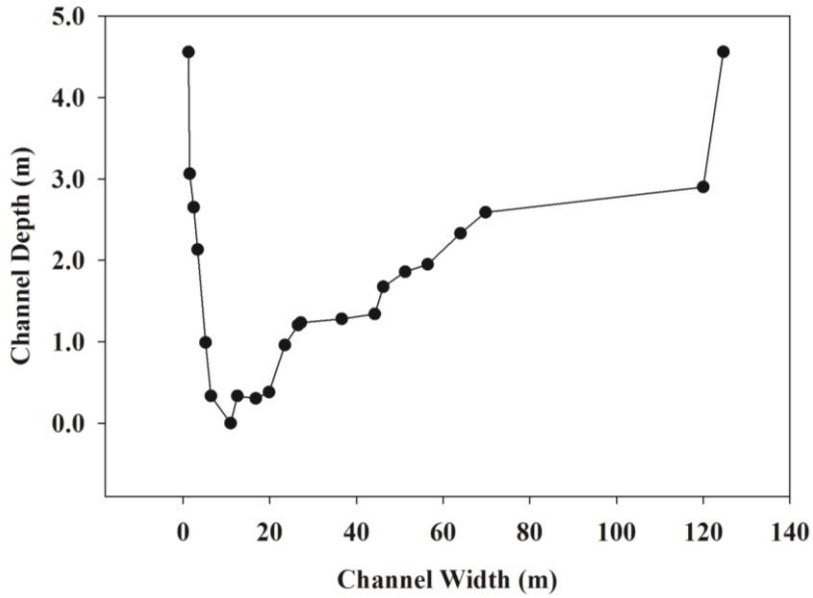

Figure A.17. Cross-sectional survey located on a meander at 346815 N, 3979706 E on the Barren Fork Creek.

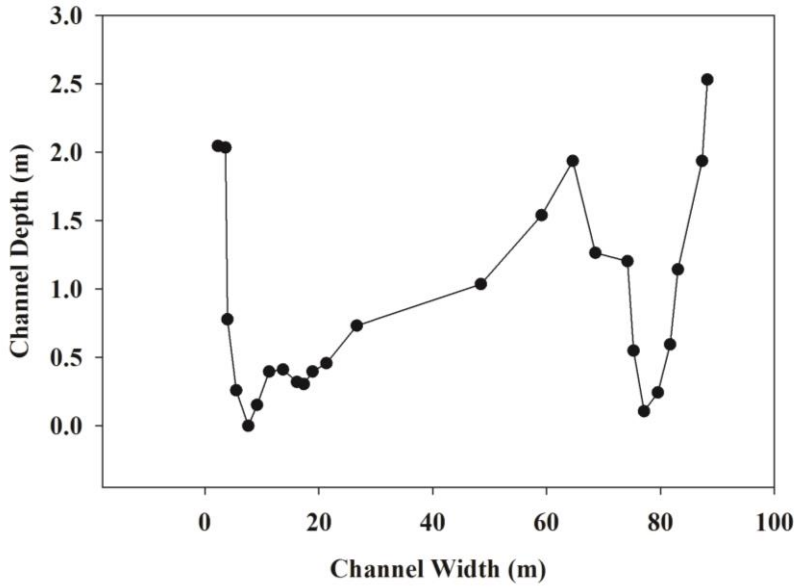

Figure A.18. Cross-sectional survey located on a meander at 340047 N, 3980843 E on the Barren Fork Creek.

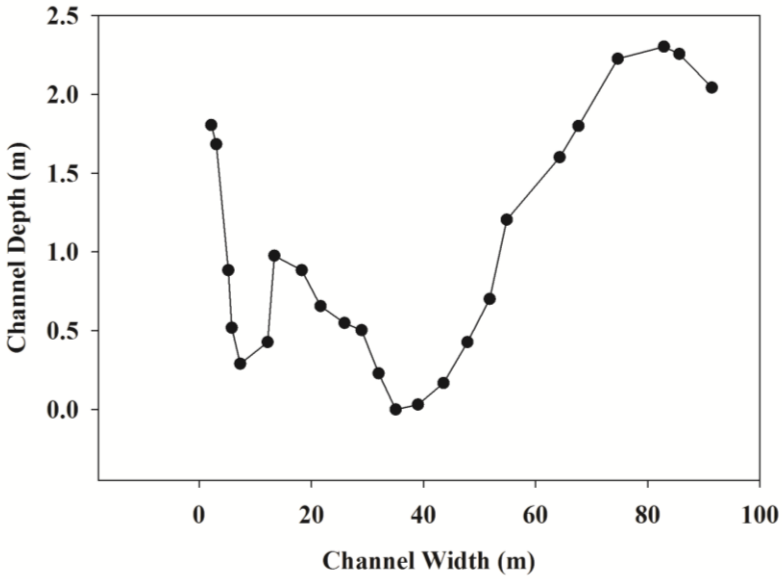

Figure A.19. Cross-sectional survey located on a cross-over at 340029 N, 3980855 E on the Barren Fork Creek.

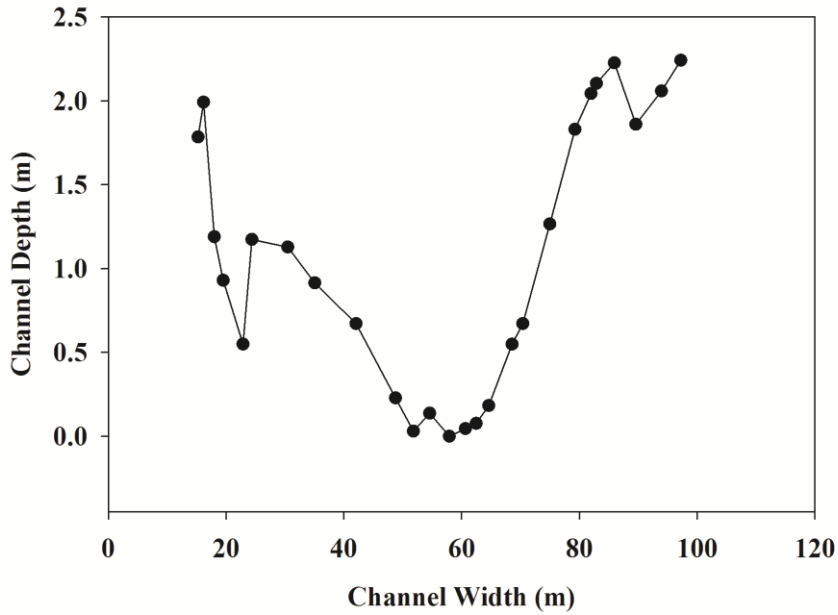

Figure A.20. Cross-sectional survey located on a straight reach at 339979 N, 3980899 E on the Barren Fork Creek.

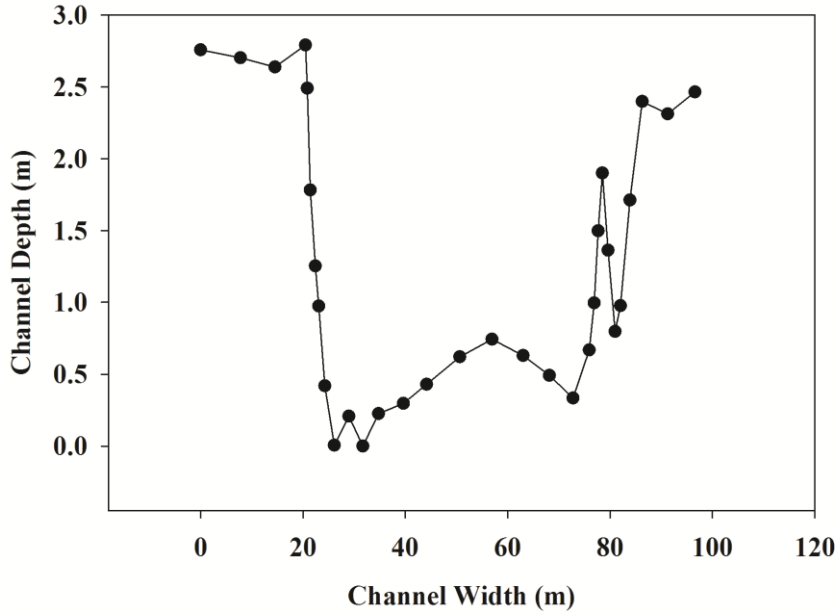

Figure A.21. Cross-sectional survey located on a straight reach at 333579 N, 3976229 E on the Barren Fork Creek.

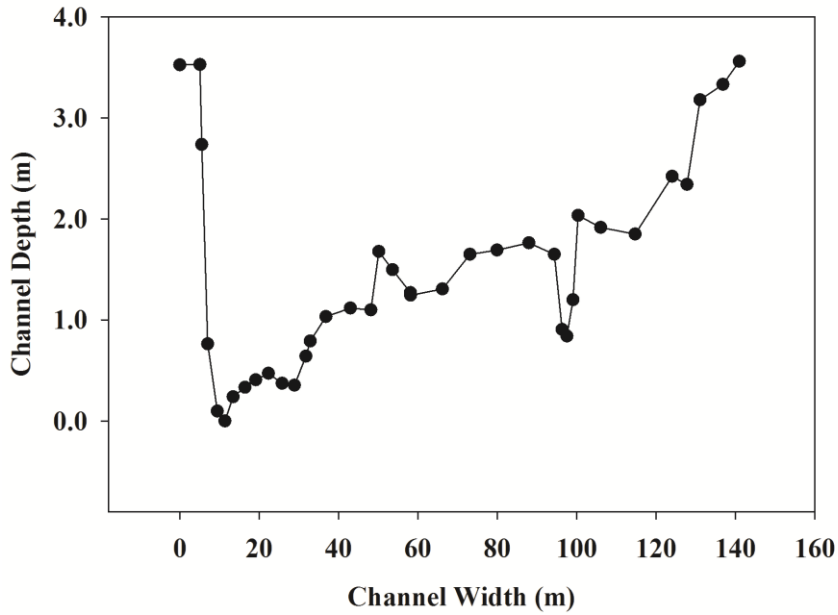

Figure A.22. Cross-sectional survey located on a straight reach at 333451 N, 3975536 E on the Barren Fork Creek.

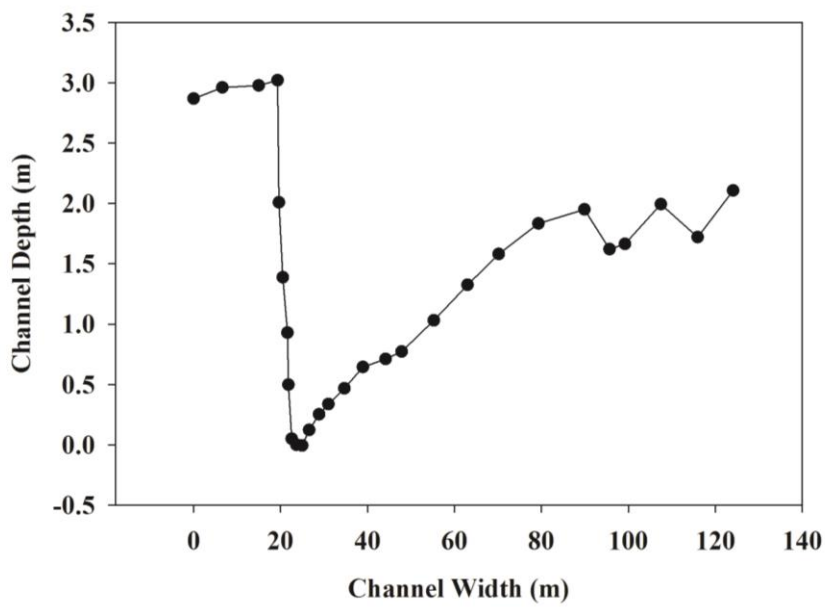

Figure A.23 Cross-sectional survey located on a meander at 333413 N, 3975106 E on the Barren Fork Creek.

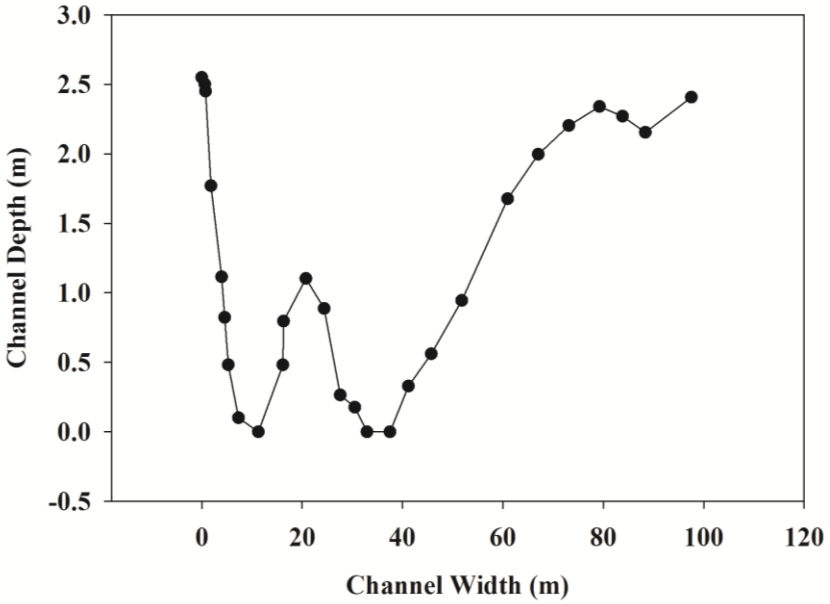

Figure A.24. Cross-sectional survey located on a straight reach at 332633 N, 3974785 E on the Barren Fork Creek.

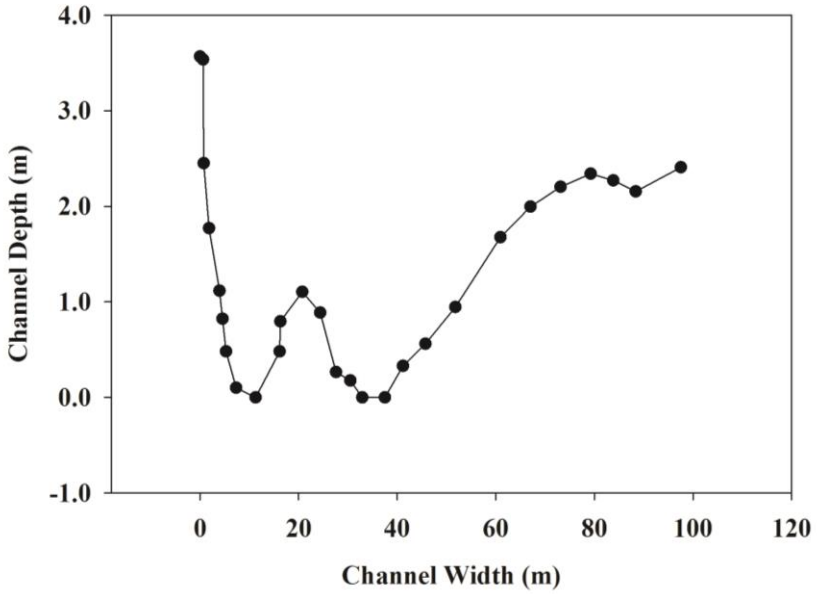

Figure A.25. Cross-sectional survey located on a cross-over at 332596 N, 3974712 E on the Barren Fork Creek.

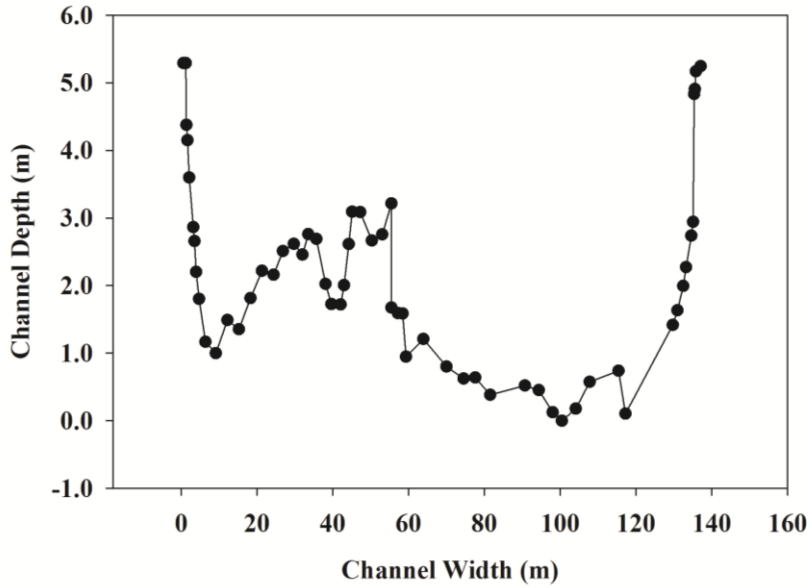

Figure A.26. Cross-sectional survey located on a straight reach at the U.S. Geological Survey gage station near Eldon, Oklahoma (334227 N, 3976830 E) on the Barren Fork Creek.

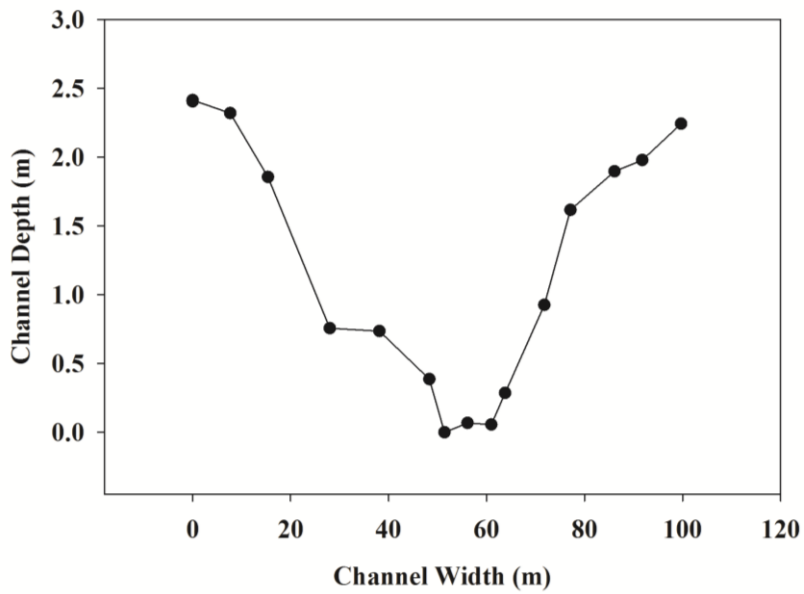

Figure A.27. Cross-sectional survey located on a cross-over at 332644 N, 3974899 E on the Barren Fork Creek.

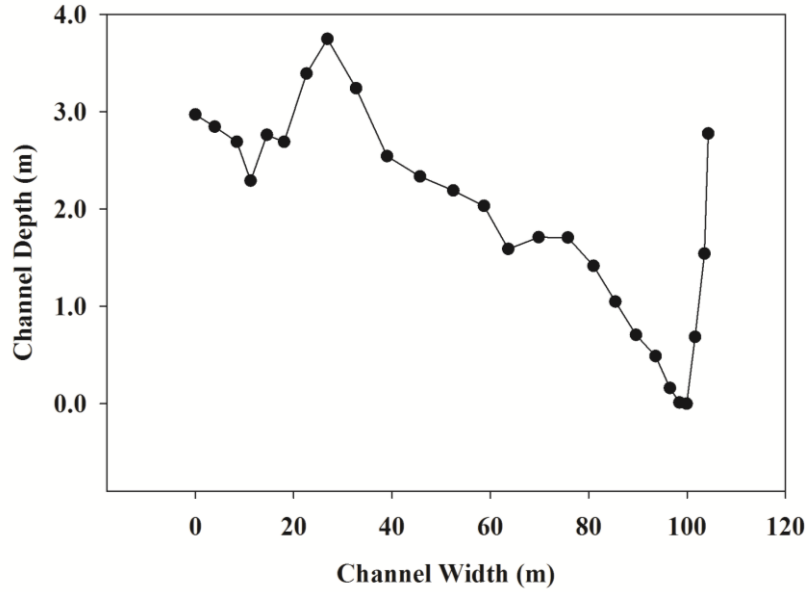

Figure A.28. Cross-sectional survey located on a meander at 332274 N, 3974867 E on the Barren Fork Creek.

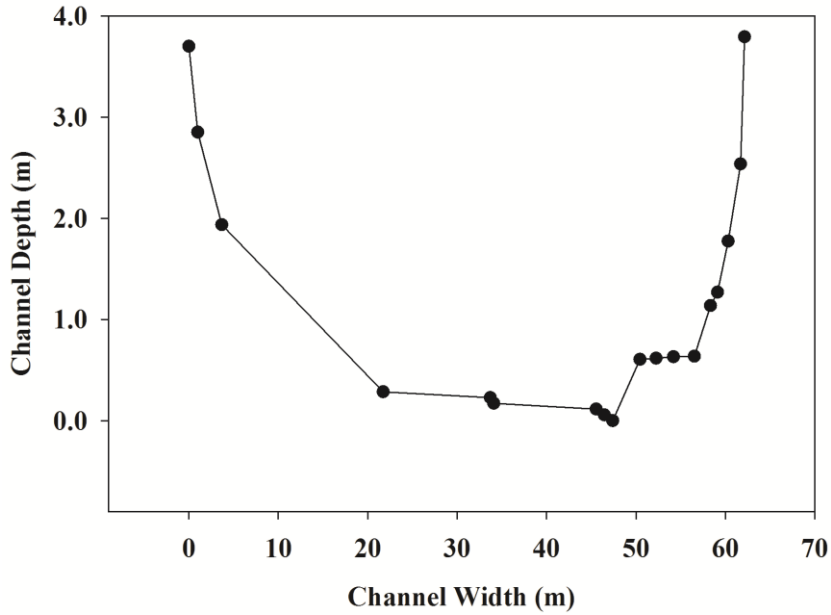

Figure A.29. Cross-sectional survey located on a straight reach at 331669 N, 3973131 E on the Barren Fork Creek.

VITA

Aaron Ray Mittelstet

Candidate for the Degree of

Doctor of Philosophy

Thesis: QUANTIFYING PHOSPHORUS LOADS AND STREAMBANK EROSION IN THE OZARK HIGHLAND ECOREGION USING THE SWAT MODEL

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