SEDIMENT AND PHOSPHORUS LOADS FROM STREAMBANK EROSION AND FAILURE: A SOURCE OF LEGACY PHOSPHORUS IN WATERSHEDS

By

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Bachelor of Science in Biosystems Engineering

Oklahoma State University

Stillwater, OK

2014

Submitted to the Faculty of the
Graduate College of the
Oklahoma State University
in partial fulfillment of
the requirements for
the Degree of
MASTER OF SCIENCE
July, 2015

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ACKNOWLEDGEMENTS

I would like to acknowledge the financial support of the Environmental Protection Agency (EPA) under the grant FY 2012 319(h) Special Project #C9-00F56701 and a 2012-2013 Oklahoma Water Resources Research Institute (OWRRI) grant through the USGS 104(b) grant program. My committee has been full of support and guidance through my entire time at Oklahoma State University. A very special thanks to my advisor, Dr. Garey Fox. Over the past five years he has provided continual guidance and support in all that I did and willing answered my numerous questions. I am honored to call him my mentor and friend. Thank you also to Dr. Chad Penn and Dr. Dan Storm for providing support and instruction. I am deeply grateful to my committee for pushing me and allowing me to grow.

I am very appreciative to Abigail Parnell for always smiling and willing to lend a helping hand with soil processing. Also, thanks to Kate Klavon and Whitney Lisenbee for their assistance in the field. I am also forever thankful for the "Fox Family" – Anish Khanal, Erin Daly, Holly Enlow, Kate Klavon, Whitney Lisenbee, and Yan (Joey) Zhou – for putting up with my craziness, making me laugh, and constantly supporting me. I am so glad to be part of an amazing family. Thank you to all of the BAE grad students for filling the office with joy and friendship. Thanks also to Erica Poe, Katrina Fyfe, and Bryanna Rogers for being my people.

Finally, I would like to thank all of my family, especially my parents, David and Susanne Purvis, who have provided love, wisdom, and support along this journey; my brother, Timothy Purvis, for always providing a listening ear and laughter; and my grandparents: Leroy and Catherine Boyd, for welcoming me into the Cowboy family, and Sammie and Dorothy Purvis, for watching over me. Without all of them I would not be the person I am today.

Name: Rebecca Anne Purvis

Date of Degree: JULY, 2015

Title of Study: SEDIMENT AND PHOSPHORUS LOADS FROM STREAMBANK EROSION AND FAILURE: A SOURCE OF LEGACY PHOSPHORUS IN WATERSHEDS

Major Field: Biosystems and Agricultural Engineering

Abstract:
Streambank erosion may be one pathway for sediment and nutrient loading to streams but insufficient data exists on the magnitude. Riparian protection can significantly decrease streambank erosion in some locations, but estimates of actual sediment load reductions are limited. Objectives of this research include (i) reviewing current knowledge on streambanks as P loading sources and identifying future research needs, (ii) quantifying the amount of streambank erosion throughout a sensitive watershed in eastern Oklahoma, (iii) estimating the benefit of vegetation on reducing streambank erosion, (iv) determining the importance of mass wasting in this system, (v) analyzing the appropriateness of limited monitoring points to determine watershed sediment load, (vi) quantifying the magnitude of and spatial distribution of streambank phosphorus concentrations along a stream system in a watershed with historical poultry litter application, (vii) quantifying the amount of water soluble phosphorus (WSP) and total phosphorus (TP) entering the stream from streambanks, and (viii) comparing streambank P concentrations and loading between two unique streams in the same ecoregion. For Spavinaw Creek, it was estimated that the total soil mass eroded from 2003 to 2013 was 727 $\times 10^6$ kg, average bank retreat was 2.5 m yr^{-1}, and 1.5 x 10^3 kg WSP and 1.4 x 10^5 kg TP loaded. Statistical analysis showed that sites with riparian vegetation had on average three times less bank retreat than unprotected banks. Bank retreat was somewhat positively correlated with stream discharge, suggesting that mass wasting plays a role in streambank erosion within this watershed. Selection of random sites and scaling up to watershed scale greatly underestimated the actual erosion and loading rates. Comparison of P loading between the two systems showed that WSP in one was an order of magnitude higher while TP was on the same order of magnitude. Streambank P loading rates are dependent on the stream system; therefore each stream needs to be individually studied in order to gain a better understanding of the specific loadings from streambanks. Future research is needed on dynamics between different P pools and the integrated streambank erosion processes.

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CHAPTER I

PHOSPHORUS LOADS FROM STREAMBANK EROSION: A SOURCE OF LEGACY PHOSPHORUS IN WATERSHEDS

1.1 Abstract

Nutrients and excess sediment are two of the primary pollutants of surface waters in the United States. Many studies have investigated loadings from upland runoff or sediment transported in the channel, both bed and suspended sediments, but in many cases, limited to no data exist to determine sediment and nutrient loading from streambanks on a watershed scale (Wilson et al., 2008). The objectives of this paper are to review the current knowledge base on streambanks as phosphorus (P) loading sources and to identify future research needs. In many watersheds, long-term loading of nutrients to stream systems has created a legacy source of nutrients that can be mobilized during streambank erosion and failure. Streambank erosion and failure is reported to account for 17 to 92% of the suspended sediment load within a channel and 10 to 93% of P, showing the large impact this process can have on a stream system. Research is needed on the integrated processes of fluvial erosion, mass wasting, and subaerial erosion to better understand the impact of restoration/rehabilitation efforts on reducing sediment and P loading. Currently no mechanistic or empirical approaches are available for modifying erodibility parameters of soil due to subaerial processes or vegetation. Soil research needs to be driven by the goal of soil sustainability, which can be obtained through quantitative principles and measurements of soil

erosion and production, and soil nutrient loss and release. An improved understanding is needed on the dynamic processes between different P pools, and sorption or desorption processes under varying hydraulic and stream chemistry conditions that might be experienced during runoff events. Finally, research is needed on the transport rates of dissolved and sediment-bound P through the entire stream system of a watershed.

1.2 Introduction

Phosphorus (P) is often the limiting nutrient, and its increased concentration is linked with accelerated eutrophication of fresh waters (Sharpley and Rekolainen, 1997). An increase of P in surface water can lead to algae blooms. As these blooms die and decompose, dissolved oxygen in the water decreases, which can negatively impact the aquatic ecosystem (Pierzynski et al., 2000). Advanced eutrophication impacts recreation, industrial water usage, and drinking water quality due to unwanted algae and decreased oxygen levels (Sharpley and Rekolainen, 1997).

Kotak et al. (1993) reported that drinking water supplies throughout the world have experienced large, periodic cyanobacterial blooms. Such blooms contribute to many problems including fish kills, the unpalatability of drinking water, and the formation of trihalomethane during chlorination (Palmstrom et al., 1988; Kotak et al., 1994). Consuming water containing cyanobacteria or water-soluble neuro- and hepatoxins, which are released when the blooms die, can kill livestock and pose a serious health risk to humans (Lawton and Codd, 1991; Martin and Cooke, 1994). Since the 1960s, point sources of water pollution have reduced their loading. However, many water quality issues remain, showing the importance of non-point source loading. Research is beginning to focus on agricultural non-point sources as they generally show higher levels of P than other land uses.

Many studies have investigated loadings from runoff or sediment already in a channel, both from bed and suspended sediments, but in many cases, data does not exist to determine sediment and nutrient loading from streambanks on a watershed scale (Wilson et al., 2008). Streambank erosion and failure may be a significant pathway for P loading to streams, but insufficient data exists on streambank sediment concentrations and P loading for watersheds. Understanding soil conditions and quantifying P loading is necessary for determining the need for and justifying the use of protective measures, such as riparian vegetation (Sekely et al., 2002; Laubel et al., 2003; Kronvang et al., 2012). Triplett et al. (2009) found P loading in the St. Croix River to have increased eight-fold from 1850 to the 1950s and noted that the increase in loading had a close correlation to the sediment load. Zaimes et al. (2008) compared the soil and P losses to surface waters from various land practices and found that land with riparian forest buffers had the lowest contribution.

Natural soil production processes can replace lost nutrients, including P, but are much slower than anthropogenic usage. This deficiency can lead to reduced production (Jones et al., 2013), but also increases the reliance on mining of P (Cordell et al., 2009). However, the United States contains only 2% of global P reserves (USGS, 2015). Amsundson et al. (2015) state that an integrated program to understand nutrient cycling and the potential for recycling is greatly needed to reduce the dependency upon nutrient imports. Gaining an understanding of streambank P concentrations, spatial distribution, and P dynamics within a system is complex and important, but must go hand-in-hand with studying fluvial processes and erosion behavior. Each individual study can show important characteristics, and combining them can help show a more complete picture. P loading estimations have major implications for evaluating the effectiveness of different management practices and the need for water treatment. This research can also be valuable for integrated approaches for P recycling. The objectives of this paper are to review the current knowledge base on P dynamics in soils as it relates to streambanks, summarize current literature on streambanks as P loading sources, and identify future research needs.

1.3 Phosphorus Pools

Streambanks may act as a major nonpoint P sources. Although P is an abundant nutrient in the environment, it is never found in its elemental form. In general, soil P can be split into three pools: solution P, active P, and fixed P. The solution P pool is a small fraction, but is very important as it is the pool where plants uptake P. This is also the only pool in which mobility of P can be measured. Solution P is usually in the orthophosphate form, but may contain small amounts of organic P. This pool can be easily reduced if there is plant growth, but no replenishment (Busman et al., 2009).

The active pool is composed of solid P, which can be easily released into the water solution within the soil. As the P concentration in the solution pool is reduced through plant uptake, the active pool P will replenish it to maintain P availability to plants. Soil fertility is based on the ability of the active P pool to replenish phosphate to the solution pool. This pool is composed of inorganic phosphate sorbed to small soil particles, phosphate which has reacted with elements (e.g., Ca or Al) to form slightly soluble solids, and easily mineralized organic P. Soil particles associated with the active pool can act as a sink or a pool. Therefore, soil eroded into surface water can adsorb excess phosphate or release excess P to the water (Busman et al., 2009).

The fixed P pool is made up of highly insoluble inorganic phosphate compounds and organic compounds that are resistant to microorganism mineralization. Phosphate in this pool has been known to remain for several years without becoming plant available, meaning this pool has little impact on soil fertility. There can be small amounts of fixed P being converted to join the active P pool, but the process is very slow (Busman et al., 2009). Each pool behaves differently, but a deeper understanding of their interaction is needed to better target potential sources of P loading and long-term P pools.

Fertilizer and manure, both large sources of P, are regularly added to land in order to improve fertility and promote plant growth. Phosphate fertilizer is typically pre-treated with acid to make it more soluble, and manure contains soluble, organic, and inorganic phosphates. Initially, moisture from the soil will dissolve the fertilizer or manure particle, which increases the concentration of soluble phosphate in the soil solution. This soluble P is then free to begin moving from the initial particle. Movement will increase the potential for a reaction as the phosphate is likely to pass by and combine with elements including aluminum, calcium, iron, and magnesium to form solid compounds. The soluble phosphate can also adsorb onto soil particles. These two forms of phosphate are then plant available. However, over time other reactions will occur, making these phosphates insoluble and no longer plant available and decreasing soil test P (Busman et al., 2009). The ability for soluble phosphate to sorb onto soil particles is building up the P pool for plants, but also building up a reserve that could be loaded into surface waters through streambank erosion. Once in contact with the water there is an increased potential for the P to desorb from the soil particle and move through the water, where it could be used by algae or bacterial, sorbed to another soil particle, or remain in the water column. There are many fates of P once applied to the soil, and understanding these dynamics is important for estimating potential P pools within streambanks and loading during erosion events.

1.4 Phosphorus Bioavailability

Although there may be large P pools within streambanks, it is not all considered a potential environmental concern. Bioavailable P is that which is immediately available and P that can be transformed to become available through physical, chemical, and biological processes. It has been found that orthophosphate ($H_2PO_4^-$, HPO_4^{2-}, or PO_4^{3-}) is generally the only bioavailable P that will be taken up by bacteria and planktonic algae (Boström et al., 1988). The bioavailability of P is a key component when determining the potentially negative effects of P loading in surface waters. Bioavailability of P is controlled by the time in which the particle is accessible to algae or bacteria and the potential for mobilized sedimentary P to reach the site of algae or bacteria requiring P for production. It is also heavily dependent on the chemical environment: mobilization is greatly affected by environmental factors including pH, redox potential, occurrence of chelators, and degree of dilution (Boström et al., 1988). Another aspect of understanding P bioavailability is in gaining a better understanding of the time perspective in which a P source could remain. Many studies have used the term 'bioavailable P' where their results indicated the minimum P available and did not detect P compounds with slower availability (Nürnberg and Peters, 1984). Results from these studies are beneficial in understanding P dynamics of the system, but are not as helpful in determining management practices, as it does not incorporate the full P available for algae uptake (Boström et al., 1988).

Many studies have carried out different extraction methods in order to estimate the bioavailability of P. However, Boström et al. (1988) points out that many of these routine methods are carried out in a method-specific chemical environment that may not favor certain P mobilization processes. It should also be noted that the algal species plays a large role in which forms of P are considered bioavailable. For example, it was found that pyrophosphate is utilized by *Chlorella* sp. (Galloway and Krauss, 1963) and *Selenastrum* sp. (Fitzgerald, 1970), but not by *Scenedesmus quatriqauda* (Overbeck, 1962). Each of these bioavailability studies has tried to create a method in which to limit the dependent variables, such as Lee et al. (1980) who suggested a batch algal culture test where all conditions for algal growth were optimized and only the P concentration was changed. Although such bioassay tests may show the direct effect of P on algae, the optimized conditions for algae are not always the optimized conditions for P mobilization. An increase in pH causes an increase in P mobilization from sediments (Boström et al., 1988), but test algae has maximum growth at a neutral pH (Golterman, 1977). Many bioassay systems maintain oxic and aerobic respiration conditions, which may prevent mobilization of P from bacterial cells (Shapiro, 1967; Fleischer, 1983) and certain iron compounds in the sediment (Boström et al., 1988). Common assay systems used to estimate bioavailable P in particulate matter include batch systems where algae and particulates are mixed (Lee et al., 1980), two-chambered vessels where algae and particulates are separated by a membrane filter (e.g., Hosomi et al., 1981; Young and DePinto, 1982; Premazzi and Zanon, 1984; Marengo and Premazzi, 1985), general bioassay, and batch systems without algae (Furumai and Ohgaki, 1982; Boström, 1984).

Total dissolved P (TDP) generally encompasses any P compound that passes through a 0.45 μm membrane filter, which means a portion of colloids is included in TDP (Broberg and Persson, 1988). TDP is split into dissolved reactive P (DRP) and dissolved unreactive P (DUP). It is assumed that DRP is equivalent to orthophosphate, making it completely and uniformly biologically utilized by algae and bacteria. DUP is considered to be mostly inert (Lee et al., 1980). However, recent studies have reported that DRP may not be as available as orthophosphate or taken up in the same way (Boström et al., 1988). Cembella et al. (1984) reported that some P esters generally placed in the DUP pool are actually utilized by algae. Therefore, the bioavailable P may not always correspond with the DRP. It was found that there are 17 defined compounds, dominated by phosphatemonoesters, which are successfully utilized by a large number of algal species (Cembella et al., 1984). Some esterified P in large, complex molecules or inorganic condensed phosphates were found to be utilized, but with a lower occurrence (Chu, 1946; Overbeck, 1962). The statement of bioavailable dissolved P being equal to the DRP pool has been found to generally apply for the long-term availability (Walton and Lee, 1972; Lee et al., 1980; Logan, 1982; Sonzogni et al., 1982). However, short-term availability is more dependent on the relationship between the immediately available P pool and DRP (Nürnberg and Peters, 1984). Boström et al. (1988) states that the main drawback to bioassay studies is that each study uses their own particular assay method and there is an urgent need for comparative tests between the methods to determine the limits of their use.

As previously mentioned, environmental conditions play a large role in how P behaves and therefore how much will be bioavailable. Hydraulic processes within the system will determine particulate residence time in the water column before sedimentation that will impact the time available to be taken up from the water column (Williams et al., 1980; Sonzogni et al., 1982). Some bioavailable P may be considered unavailable if it is associated with rapidly settling particles (Sonzogni et al., 1982). There could be the potential for resuspension and future availability, but it is dependent on the particle and hydraulic processes. Desorption of orthophosphates is generally governed by the orthophosphate concentration in the solution. Sonzogni et al. (1982) showed that a lower orthophosphate concentration in lake water compared to tributary water favored desorption when the two waters mixed. However, Staffored-Glase and Barlow (1984) reported an enhanced adsorption when stream water with a higher DRP concentration mixed with lake water. It was explained that this was due to differences in the cation composition of the water mediums and different concentrations of dissolved organic material that were in competition for sorption sites. The differences in these studies emphasize that each system will have varying conditions that will lead to differing behaviors, making it difficult to accurately predict adsorption and desorption.

A very influential environmental condition is the pH. At a high pH the P-binding capacity of iron and aluminum compounds decreases due to ligand exchange reactions (Boström et al., 1982). Lijklema (1977) found that at a high pH recently formed hydrated iron hydroxides have a lower capacity to sorb orthophosphate. This implies that when iron (II) and orthophosphates are released from surface sediments under an anaerobic environment into aerobic lake water with a high pH, only part of the released P will become bound to iron (III) compounds. Therefore, certain external P loads that are associated with iron and aluminum will have different effects in water due to the trophic level (Boström et al., 1988). The ability of sediment to bind P will also play a role in P release. Sediment with a low P-binding capacity will lead to an increased concentration of P remaining in the water column that can contribute to eutrophication (Stauffer, 1985).

It can be difficult to accurately predict and model P behavior within a system as there are a high number of variables including algae species, trophic state, pH, sediment composition, and fluvial processes. However, gaining an understanding of the potential for movement of P is vital for predicting P loading and transport.

1.5 Streambank Erosion Mechanisms

Streambank erosion is a cyclical process that includes subaerial processes, fluvial erosion, and mass wasting (Simon et al., 2000; Couper and Maddock, 2001; Couper, 2003; Fox and Wilson, 2010). Fluvial erosion is dependent on the applied shear stress, sinuosity, and stream discharge and/or stream power and the ability of the bank's soil to resist the force (Thorne, 1981; Morisawa, 1985). Fluvial erosion is a continuous process when the soil's critical shear stress is exceeded (Daly et al., 2015). When the shear force of the discharge is greater than the resistance of the bank, then particles may be removed and loaded into the channel (Thorne, 1982). Mass movement is due to gravity (Bowie, 1982; Thorne, 1982) and is caused by a decrease in the upper bank's internal strength, resulting in saturation, undermining, or foundation deterioration from seepage (Harmel et al., 1999). Unlike fluvial erosion, mass wasting is episodic. Subaerial erosion is linked with climate and occurs when the soil strength is reduced due to subaerial processes that will then lead to direct erosion or an increased risk for erosion (Daly et al., 2015). These three processes are all dependent on each other. Subaerial erosion can initially weaken a bank, making it more susceptible to erosion. Fluvial erosion can then more easily undercut the bank or scour the bed leading to increased instability, eventually resulting in mass wasting (Fox and Wilson, 2010; Midgley et al., 2012).

One or two of these processes may dominate for some streambanks, but further analysis of the system may be needed to understand the dynamic among the three. In many cases, it is assumed that fluvial erosion is the dominant mechanism controlling erosion. Currently, many erosion models, including the Soil and Water Assessment Tool (SWAT), assume that bank retreat is solely dependent on fluvial erosion (Narasimhan et al., 2007). Understanding the importance and impact of mass wasting and subaerial erosion is necessary to better design and implement streambank stabilization techniques for these rapidly eroding streams (Heeren et al., 2012).

Generally, in the field, channel cross-sections and the implementation of bank pins are the most widely used and accepted methods for measuring bank erosion (Lawler, 1993). Cross-sections allow for the measurement of erosion and deposition, but do require a permanent base point, take more time, and show the changes only at the specified site. Bank pins have accurate measurements (± 3 mm) in alluvial material and allow for quick readings, but may be limited in the amount of erosion that can be detected (Lawler, 1993; Hooke, 1979). Systems with significant episodic retreat events can be difficult to assess with bank pins (Miller et al., 2014). Recently, studies including those by Brice (1982), Odgaard (1987), Beeson and Doyle (1995), Harmel et al. (1999), Heeren et al. (2012), and Miller et al. (2014) have used aerial imagery to estimate bank erosion. Aerial images allow for the long-term analysis of a channel and the rapid evaluation of changes over a large area. Aerial imagery analyses are generally faster than *in situ* methods, but do have an increased error. Harmel et al. (1999) reported that due to the scale of the aerial images, areas having less than 2 m of lateral erosion or deposition over the study period were not detected. Heeren et al. (2012) reported a maximum error of 3 m in bank retreat based on the georeferencing and identification of the banks.

Billions of dollars have been spent on streambank stabilization to help slow bank retreat and reduce sediment loading (Lavendel, 2002; Berhnhardt et al., 2005). Riparian buffers are common conservation practices with established cost-share programs. Vegetation can drastically reduce streambank erosion, but estimates of actual decreases in sediment and sediment-bound nutrients are limited (Beeson and Doyle, 1995; Burkhardt and Todd, 1998; Harmel et al., 1999; Miller et al., 2014). The presence of established vegetation is a preventive measure as the intricate root system helps to stabilize the soil and ultimately reduce the impact of fluvial forces on the streambank (Harmel et al., 1999; Simon et al., 2011). Previous studies report relationships between the presence of vegetation and bank retreat. Beeson and Doyle (1995) reported non-vegetated bends to be five times more likely to have notable erosion. Harmel et al. (1999) found grassed banks to be four times more likely to experience notable erosion compared to tree vegetated banks. Miller et al. (2014) reported banks with historical riparian protection to have three times less bank retreat than those with no protection. A recent study looked at the U.S. erosion rates prior to the introduction of European cultivation techniques and estimated that the average rate was 21 m My^{-1} (My, million years). However, present day estimated rates for the central U.S. can exceed 2000 m My^{-1}. In comparison, parts of the loess plateau in China may see close to 10,000 m My^{-1}. These increased erosion rates are linked with the removal of plant cover and agricultural expansion, which disrupt soil sustainability. Estimated natural rates of soil production range from 50 to 200 m My^{-1} depending on the environment, showing the unsustainability of current land practices (Amundson et al., 2015). Further understanding the effects of riparian protection on sediment loading to streams due to streambank erosion can justify the use and demonstrate the effectiveness of such management practices.

In many watersheds, long-term application of nutrients to stream systems has created a legacy source of nutrients that can be mobilized during streambank erosion and failure. Although vast research efforts have been made evaluating legacy P stores in a channel, most are focused on the bed sediment and water column. However, streambanks can represent a major source of legacy P.

1.6 Legacy Phosphorus

Many questions have arisen about the effectiveness of current conservation practices and whether they are correctly located and implemented and at a scale and intensity to accurately represent the watershed. One main issue growing from these questions is understanding the legacy of management practices, in particular the sinks and pools of P in a watershed (Sharpley et al., 2009). Previous studies have shown that current conservation practices may not be accounting for legacy P adequately (Kleinman et al., 2011; Sharpley et al., 2011). Legacy P is accumulated P which can be remobilized and serve as a continuing source downstream for years to centuries after deposition (McDowell and Sharpley, 2002; Kleinman et al., 2011). Research into legacy P behavior has been increasing to gain a better understanding of the lasting impacts of land practices, the changing of P forms, and how this continuing source can impact watersheds.

One area of focus is surface runoff. P can build up over time, creating a legacy pool, due to P sources being added to the land faster than crops can uptake. These pools have been known to take up to decades to decline if a large reserve has been formed (Cox et al., 1981; Sharpley et al., 2009). In particular, agriculture land can have total P concentrations two- to ten-fold greater than the geologic concentrations found in forests (Syers et al., 2008; Vitousek et al., 2009; Nash and Hannah, 2011; Sattari et al., 2012). However, even when legacy pools are smaller, they have been known to show large contributions due to runoff, particularly in areas that are hydrologically active or hotspots (Gburek and Sharpley, 1998; Gburek et al., 2007).

Studies have also found that legacy P can reach water systems through shallow groundwater contributions (Holman et al., 2008; Domagalski and Johnson, 2011). Generally, groundwater P transfers are considered to be negligible due to high soil P adsorption (Addiscott and Thomas, 2000), but Vadas et al. (2007) found that areas with high concentrations can be a source that needs to be taken into account. Heeren et al. (2010) reported that preferential flow pathways positioned above a shallow groundwater system may only become hydrologically active under high flow events, which is when high contamination loads usually occur. It was also found that flow pathways may become active during recharge between the surface and subsurface, which will affect the nutrient concentration in the groundwater. Recent studies found that subsurface P transport rates were significant when compared to surface runoff rates at low intensity agricultural sites (Mittelstet et al., 2011; Heeren et al, 2012).

Much of the research has been focused on the storage of legacy P already within in a river system, with little attention to potential sources. Stores within a surface water system include deposition of particulate P as fluvial bed sediments (Svendsen and Kronvang, 1993; Ballantine et al., 2009; Rawlins, 2011), sorption of dissolved P onto channel bed sediment (Haggard et al., 2001; 2005; Jarvie et al., 2005; Stutter et al., 2010), sorption onto suspended sediments that may be later deposited on the bed (Owens and Walling, 2002), or water-column P being taken up by plant or microbial biomass (Aldridge et al., 2010; Schade et al., 2011; Drake et al., 2012).

1.7 Streambanks as a Sediment and Legacy P Source

Eroded sediments can be transported downstream and deposited, with excessive deposition leading to the formation of new banks. These banks could be composed of either sediment eroded from a streambank or from the channel bed, both having the potential to transport particulate P. Previous research studies report that streambank erosion and channel scour accounted for 17% to 92% of the suspended sediment load within a channel, demonstrating the large impact this process can have on a stream and receiving reservoir (Table 1.1).

Table 1.1. Previous study estimates on suspended sediment load from streambanks.

Author	Location	Channel	Channel Length (km)	Drainage Area (km^2)	Soil Type	Suspended Sediment Load from Streambanks (%)
USACE (1983)	Western United States (California)	Sacramento River	$7.2x10^2$	$7.1x10^4$	Loam and Silt loam	59
Simon and Hupp (1986)	Southeast United States (Tennessee)	Obion Forked Deer River	NA	$2x10^3$	Loam	81
Odgaard (1987)	Central United States (Iowa)	East Nishnabotna River/Des Moines River	$2.6x10^1$/ $8.5x10^2$	$2.3x10^3$/ $4.1x10^4$	Silt loam	30-40
Ashbridge (1995)	Southwestern England	River Culm	$2.7x10^1$	$2.8x10^2$	Loamy	19
Bull (1997)	Southwestern England	River Severn	$3.5x10^1$	$9x10^0$	Silty clay loam	17
Kronvang et al. (1997)	Denmark	Gelbæk Stream	NA	$1.2x10^2$	Sandly loam and Sandy clay	92
Walling et al. (1999)	Northeast England	River Ouse	$8.4x10^1$	$3.3x10^3$	Loamy and clayey	37
Rondeau et a. (2000)	East-central North America	St. Lawrence River	$4x10^3$	$1x10^6$	Loamy	65
Simon et al. (2002)	Southeast United States (Mississippi)	James Creek	$2x10^1$	$7.4x10^1$	Silt loam	78
Simon and Thomas (2002)	Southeast United States (Mississippi)	Yalobusha River	$2.7x10^2$	$4x10^3$	Clay and loam	90
Simon et al. (2004)	Southeast United States (Alabama)	Shades Creek	$8.8x10^1$	$1.9x10^2$	Loam	71-82
Wilson et al. (2008)	Central United States (Oklahoma)	Fort Cobb Reservoir	NA	$8x10^2$	Sandy loam	46

Streambank erosion is a growing concern for P loading (Table 1.2), as some soils have the capacity to store large amounts of legacy P. Most P in soils is associated with soil particles, meaning that when sediment is loaded into surface water, the P will also be loaded into the water. Depending on the soil P concentration and water P concentration, the sediment may act as a sink or a source of P, meaning P could readily be released into the system (Busman et al., 2009). Studies report links between increased sediment loading and increases in P (Engstrom and Almendinger, 1997; Kroening and Andrews, 1997). Understanding fluvial processes allows for more accurate modeling and predictions of sediment loading from streambank erosion and thus an estimate for nutrient loading from these banks.

Table 1.2. Previous study estimates on phosphorus load from streambanks.

Author	Location	Channel	Channel Length (km)	Drainage Area (km^2)	Soil Type	Phosphorus Load from Streambanks (%)
Boynton et al. (1995)	Northeast United States (Maryland and Virginia)	Chesapeake Bay	$1.1x10^4$ (shoreline)	$1.7x10^4$	Sandy loam and Clay loam	10
Kronvang et al. (1997)	Denmark	Gelbæk Stream	NA	$1.2x10^2$	Sandly loam and Sandy clay	93
Sekely et al. (2002)	Northern United States (Minnesota)	Blue Earth River	$1.7x10^2$	$9x10^3$	Silty clay loam and Loam	7-10
Kronvang et al. (2012)	Denmark	River Odense	$6x10^1$	$4.9x10^2$	Fine clayey sand	17-25
Miller et al. (2014)	South-central United States (Oklahoma)	Barren Fork Creek	$5.6x10^1$	$8x10^2$	Gravelly silt loam	10

1.8 Future Research Needs

1.8.1 Quantifying Streambank P Concentrations

Only limited studies exist in the literature at this time regarding streambank sediment and P loading to streams (Tables 1.1 and 1.2). Additional research is needed to quantify streambank P concentrations, both water soluble phosphorus (WSP) and total phosphorus (TP) concentrations. At the same time, studies should consider differences in soil chemistry between the streambank materials and surrounding upland soils. For example, Miller et al. (2014) reported significantly different soil pH values for streambank sediment compared to upland fields, noting the potential for transported sediment influenced by agricultural fertilization to alter soil chemical properties. Additional data are needed on streambank P loading rates and the fate and transport of P in stream channels.

1.8.2 Soil and Phosphorus Dynamics

Future research needs to look at gaining a deeper understanding of how in-stream and streambank P will behave in a channel. Previous research has focused on understanding P sorption and desorption on soil particles, but needs to be expanded to determine if the behavior will change when the process is taking place in a channel under variable hydrological conditions. Streambank chemistry and nutrient concentrations can be highly dynamic during an event. Understanding what happens to a P once reaching the channel is very important in determining the lasting impact of sediment loading and P loading. Comparing concentrations over time and distance can help in determining the concentrations of P being moved downstream. Programs like LOADEST (Runkel et al., 2004) use water quality data and measured stream discharge from the United States Geological Services (USGS) to predict the P loading over time. However, the question arises as to how to track sediment from banks that have been eroded and how to better understand their fate and transport. Many scenarios arise that make it difficult to get an accurate loading. Once a particle is eroded from a streambank into a channel, it could be moved as suspended sediment through the entire system and deposited at the mouth of the channel. Once deposited, any P on the particle could be released under the correct conditions or the P could remain on the particle for an extended period, creating a source of legacy P. This legacy P could be released within a few days or decades, all depending on the environmental conditions. However, the relationship with these conditions is also not entirely understood.

Another scenario involves the particle being eroded and quickly deposited within the system, leading to the creation of a point bar or eventually a new streambank or floodplain before passing a gauge station. P released from this particle may be sorbed onto sediment along the streambank or bed and increase the P reserve there, become incorporated into the water column, or taken up by biotic life, such as algae, which could then lead to eutrophication as discussed earlier. Not all of these potential pathways will be detected with water quality samples, but are all potential P sources that could affect the system and need to be accounted for and modeled. There could be large pools of P attaching to sediment or being used by microorganisms before a water sampling station that could be neglected and therefore will not be a focus for loading prevention measures.

Research questions include determining the ability of P to sorb or desorb from streambank soils with high concentrations, what happens to a particle with P once it is eroded, and how long will it take a particle to move through a channel system. As previously mentioned, there are many variables that affect the dynamics of sorption and desorption. A growing area of interest is understanding the dynamic between the different P pools and determining which has the greatest impact, either by sorption or desorption. This is commonly studied using isotopic labelling, a technique that tracks the passage of an isotope. Discussions have also suggested future work looking into sorption/desorption dynamics under anoxic conditions and at what depth below a channel or lake particulate P will no longer be a source to overlying water. One major focal point is how each of the discussed variables will behave under different soil mineralogy compositions.

Research has seen similar trends in P behavior across differences in soil mineralogy, but the question arises about changes in magnitude of P transfer. Mineralogy will have a major impact on soil and P chemistry that will control sorption and desorption dynamics and research on the influence will help in determining which soils may be a greater risk for release.

1.8.3 Streambank Erosion Mechanisms and Protection

Studies have shown that streambank erosion can be a major source of sediment loading into a system and potentially a significant source of P. Although research needs to be conducted to determine the impact of sediment and nutrient loading, research also needs to evaluate ways in which streambank erosion can be reduced. Riparian vegetation protects streambanks as it decreases the water velocity and therefore its erosive force, creates a physical barrier between the water and bank, and the root system binds the banks soil (Cooper et al., 1990). Riparian buffers also trap upland sediment, nutrients, and pesticides before reaching streams. Established riparian vegetation lowers flows in associated flood plains during high flows, reducing the detachment capability and flow transport capacity (Hickin, 1984). Miller et al. (2014) reported a three times reduction in bank retreat on banks with riparian vegetation. Harmel et al. (1999) found that grassed banks were two times as likely to experience detectable erosion compared to mixed vegetation banks and four times more likely when compared to forested banks. Beeson and Doyle (1995) found that non-vegetated banks were two times as likely to experience detectable erosion compared to semi-vegetated banks and five times more likely than vegetated banks.

Fluvial processes within a channel are key to its changes, including erosion and deposition. Understanding these processes and their impact within a specific channel is important for determining loading sources. Future research is needed to predict the influence of vegetation on the shear stress applied to streambanks and the impact of roots on the soil erodibility parameters in order to be able to better model streambanks. Previous research has found that the commonly ignored subaerial erosion processes can have a major impact on the resistance of cohesive streambanks leading to fluvial erosion (Couper and Maddock, 2001; Clark and Wynn, 2007; Grabowski et al., 2011). There are still many factors that influence cohesive soil erodibility, including soil texture, structure, unit weight, and water content (Grabowski et al., 2011). One common method for measuring erodibility and estimating erodibility parameters is the Jet Erosion Test (JET), which can be used both in the laboratory and in-situ (Hanson, 1990; Hanson and Cook, 1997; Hanson and Simon, 2001). However, further research is needed on the derivation of erodibility parameters from the JET data and the effects of subaerial processes on the parameters (Daly et al., 2015). Research needs to look at how the technical processes of fluvial erosion, mass wasting, and subaerial erosion interact and affect the bank stability and erodibility. Currently, no mechanistic or empirical approaches are available for modifying erodibility parameters of soil due to vegetation. Research should also consider the integrated functions of a riparian buffer as a filter, stabilization mechanisms, and its ecological and aquatic habitat benefits.

1.8.4 Linking Watershed-Scale and In-Stream Process-Based Models

Process-based models are available for watershed processes (e.g., SWAT, AnnAGNPs) and also for in-stream flow and sediment transport (e.g., CONCEPTS). However, a need still exists for research on the linkage between upland sediment/P transport to streams and then the corresponding in-stream processes. Research is also needed on improving our ability to predict long-term P dynamics in a watershed or upland soils, streambeds, and streambank sediment. As noted earlier, significant effort and funding have been invested recently in stream restoration/rehabilitation, which commonly focuses on streambank stabilization. Typically, these projects are focused on specific reaches of a stream within an entire watershed and most commonly utilize analog or classification based systems as part of the design process. However, questions exist regarding the benefit of this limited restoration on total sediment and nutrient loads within an entire watershed. What is the benefit of an isolated stream restoration/rehabilitation project compared to investment in other management practices? Research is needed to quantify the benefit of restoration on sediment and nutrient load reduction.

1.9 Conclusions

There are a large number of variables in a stream system. Gaining a better understanding of P chemistry, P dynamics with soil and water, quantifying and monitoring erosion, evaluating erosion prevention practices, the importance of fluvial erosion, mass wasting, and subaerial erosion, and the various pathways of P within a channel system will be vital to improving water quality. Databases for sediment and P loading from streambanks need to be expanded beyond the studies reported herein. A primary variable that influences all P mechanisms is the soil mineralogy. Future research on how changes in mineralogy affects P dynamics is needed to allow for better modeling and estimating associated risks. Using techniques such as JETs to determine erodibility parameters can help in estimating and modeling streambank erosion, but additional research advances are still required. Isotopic labeling can assist in determining the dynamics and movement of P between sediment and water. Linking together streambank erosion processes and P dynamics can allow for improved loading models, management practices, and improved water quality.

1.10 Acknowledgements

This material is based upon work supported by the Environmental Protection Agency (EPA) under the grant FY 2012 319(h) Special Project #C9-00F56701 and a 2012-2013 Oklahoma Water Resources Research Institute (OWRRI) grant through the USGS 104(b) grant program.

The authors acknowledge Joshua Cross and Amanda Fox for reviewing earlier versions of this manuscript.

CHAPTER II

STREAMBANK SEDIMENT LOADING RATES AT THE WATERSHED SCALE: AN ARCGIS APPROACH

2.1 Abstract

Streambank erosion may be one pathway for sediment and nutrient loading to streams, but insufficient data exists on the magnitude of this source. Riparian protection can significantly decrease streambank erosion in some locations, but estimates of actual sediment load reductions are limited. Objectives of this research included (i) quantifying the amount of streambank erosion and failure throughout a sensitive watershed in eastern Oklahoma, (ii) estimating the benefit of riparian management practices on reducing streambank erosion and failure, (iii) determining the importance of mass wasting in this system, and (iv) analyzing the appropriateness of selecting a few monitoring points to determine an entire stream sediment load from streambanks. The research focused on Spavinaw Creek within the Eucha-Spavinaw watershed in eastern Oklahoma, where composite streambanks consist of a small cohesive topsoil layer underlain by noncohesive gravel. Erosion from 2003-2013 was derived using aerial photography and processed in ArcMap to quantify eroded area. ArcMap was also utilized in determining the bank retreat rate at various locations in relation to the riparian vegetation buffer width. Statistical analysis showed that sites with riparian vegetation had on average three times less bank retreat than unprotected banks. The total soil mass eroded from 2003 to 2013 was estimated at 7.27 x10^7 kg yr^{-1} and the average bank retreat was 2.5 m yr^{-1}. Many current erosion models assume that fluvial erosion is the dominant

stream erosion process. Bank retreat was positively correlated with stream discharge and/or stream power, but with considerable variability, suggesting that mass wasting may play a role in streambank erosion within this watershed. Finally, watershed monitoring programs only characterize erosion at a few sites and may scale results to the entire watershed. Selection of random sites and scaling to the watershed scale greatly underestimated the actual erosion and loading rates.

2.2 Introduction

Streambank erosion is a cyclical process that includes subaerial processes, fluvial erosion, and mass wasting (Couper and Maddock, 2001; Couper, 2003; Simon et al., 2000; Fox and Wilson, 2010). One of these processes may dominate for some streambanks, but further analysis of the system is needed to determine which one. In many cases, it is assumed that fluvial erosion is the dominant mechanism controlling erosion. Fluvial erosion is dependent on the applied shear stress, sinuosity, and stream discharge and/or stream power. Currently, many erosion models, including the Soil and Water Assessment Tool (SWAT), assume that bank retreat is solely dependent on fluvial erosion (Narasimhan et al., 2007). Composite streambanks, such as those found in the Eucha-Spavinaw watershed would commonly be assumed to erode due to fluvial erosion (Midgley et al., 2012; Miller et al., 2014). However, there is currently little data showing how important mass wasting is in the system. Understanding its importance and impact is necessary to better design and implement streambank stabilization techniques for these rapidly eroding streams (Heeren et al., 2012).

In many cases, insufficient data exists to determine sediment loading from streambanks on a watershed level (Wilson et al., 2008), but previous studies have found that streambanks can have large contributions to suspended sediment. For example, streambanks in the United Kingdom found that streambank erosion contributed 17% of the suspended sediment load in the

River Severn (Bull, 1997), 19% for the River Culm (Ashbridge, 1995), and 37% for the River Ouse (Walling et al., 1999). A study in Denmark estimated that 92% of the suspended sediment load was from a combination of streambank erosion and channel scour (Kronvang et al., 1997). The United States Army Corps of Engineers (1983) estimated that 59% of the sediment flux into the Sacramento River was from streambank erosion. Rondeau et al. (2000) estimated that streambank erosion and channel scour along parts of the St. Lawrence River contributed 65% of the suspended sediment load. Streambank erosion from the East Nishnabota and Des Moines Rivers in Iowa contributed 30-40% of the load (Odgaard, 1987). Simon et al. (2002) estimated that in James Creek, 78% of the total sediment load was from streambank erosion. It was reported in Shades Creek streambank contributions ranged from 71-82% (Simon et al., 2004). Yalobusha River streambanks were estimated to contribute 90% to the suspended sediment load (Simon and Thomas, 2002). Simon and Hupp (1986) reported that 81% of suspended load in Obion Forked Deer River were from streambanks. Eroded surface soil was estimated to account for 46% of suspended sediment in the Fort Cobb Reservoir (Wilson et al., 2008). Overall these studies found that streambank erosion and channel scour accounted for 17% to 92% of the suspended sediment load within a channel, showing the large impact this process can have on a system

Also, in many watersheds long-term application of nutrients to stream systems has created a legacy source of nutrients that can be mobilized during streambank erosion and failure. Billions of dollars have been spent in the United States on streambank stabilization to help slow bank retreat and reduce sediment loading (Lavendel, 2002; Berhnhardt et al., 2005). Riparian buffers are common conservation practices with established cost-share programs. Vegetation can drastically reduce streambank erosion, but estimates of actual decreases in sediment and sediment-bound nutrients are limited (Beeson and Doyle, 1995; Burkhardt and Todd, 1998; Harmel et al., 1999; Miller et al., 2014). The presence of established vegetation is a preventive measure as the intricate root system helps to stabilize the soil and ultimately reduce the impact of fluvial forces on the streambank (Harmel et al., 1999; Simon et al., 2011). Previous studies have found relationships between the presence of vegetation and bank retreat. Beeson and Doyle (1995) reported non-vegetated bends to be five times more likely to have notable erosion. Harmel et al. (1999) found grassed banks to be four times more likely to experience notable erosion compared to tree vegetated banks. Miller et al. (2014) reported banks with historical riparian protection to have three times less bank retreat than those with no protection. Further understanding the effects of riparian protection on sediment loading to streams due to streambank erosion can justify the use and demonstrate the effectiveness of such management practices.

Current monitoring procedures for streambank erosion and failure typically select a few sites along a stream to monitor for erosion rates (Rosgen, 2001). These selected sites may be those known to having large erosion events, but they may also be limited to those that are easily accessible. In many cases these few sites are scaled up to represent the streambank retreat rates and soil loading rates for a whole watershed, such as in Simon et al. (2009) and Miller et al. (2014). The scaled up representation is then used to help determine how many erosion reduction practices need to be implemented. However, the scaling up of these few sites may not provide an accurate representation of the erosion behavior that the stream may actually be experiencing. More data on the accuracy of using a few sites for predicting erosion of an entire watershed is needed to determine if this procedure is effective.

Therefore, the research objectives were four-fold: (i) quantify the amount of streambank erosion and failure throughout a sensitive watershed in eastern Oklahoma, (ii) estimate the benefit of riparian management practices on reducing streambank erosion and failure, (iii) determine the importance of mass wasting in this system, and (iv) analyze the appropriateness of selecting a few monitoring points to determine an entire stream sediment from streambanks.

2.3 Materials and Methods

2.3.1 Eucha-Spavinaw Watershed Description

The Eucha-Spavinaw watershed spans the Oklahoma-Arkansas border, with 60% in Oklahoma. Lake Eucha and Lake Spavinaw are reservoirs that supply water for one million people (OCC, 2007). The Eucha-Spavinaw watershed has many reaches listed on the EPA 303(d) impaired waters list for nutrient related impairments, which has led to eutrophication of Lake Eucha and compromised drinking water and increased treatment costs (OCC, 2007). This research focused on Spavinaw Creek, a fourth order stream originating in northwestern Arkansas, adjacent to land with long-term poultry litter application, which flows west into Oklahoma. Spavinaw Creek continues further west from Lake Eucha into Lake Spavinaw (Figure 2.1) and contributes 57% of total runoff within the watershed (OCC, 2007). Streams in this watershed are characterized as rapidly eroding and consisting of cherty soils and gravel streambeds (Heeren et al., 2012). The topsoil is typically a silt loam material. Based on a study in 2007, at least 48% of streambanks in this watershed are classified as unstable due to underlying unconsolidated gravel being undercut by fluvial forces (OCC, 2007).

Storm et al. (2002) reported land use for the watershed in the following categories: row crop (2.6%), forested (51.3%), hayed pastures (13.3%), well managed pastures (23.1%), poorly managed pastures (6.5%), brushy rangeland (0.1%), urban (1.3%), and water (1.7%). A large portion of the pastures are used for poultry production. The Oklahoma Department of Agriculture, Food, and Forestry (ODAFF) and the Arkansas Soil and Water Conservation Commission (ASWCC) reported that the watershed has an industry large enough to annually support 77 million birds, leading to an annual production of 73,000 metric tons of litter, which contains over 1,300 metric tons of phosphorus (Everett, 2004).

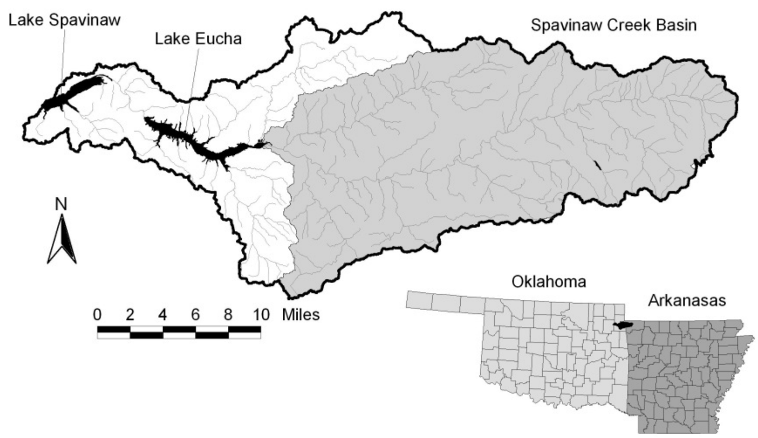


Figure 2.1. Eucha-Spavinaw watershed in northeastern Oklahoma and northwestern Arkansas.

2.3.2 Spavinaw Creek Discharge

One major factor for streambank erosion is stream discharge and the resulting stream power. Spavinaw Creek daily flow data over a 10-yr period (2003-2013) was collected from United States Geological Services (USGS) gauge 07191220 near Sycamore, OK (Figure 2.2). Stream discharge was then used to calculate the stream power:

$$\Omega = \rho_w g Q S \quad (1)$$

where Ω is the stream power (W), ρ_w is the water density (kg m^{-3}), g is gravitational acceleration (m s^{-2}), Q is the stream discharge (m^3 s^{-1}), and S is the channel bed slope. Note that stream power is directly proportional to the stream discharge (Figure 2.2).

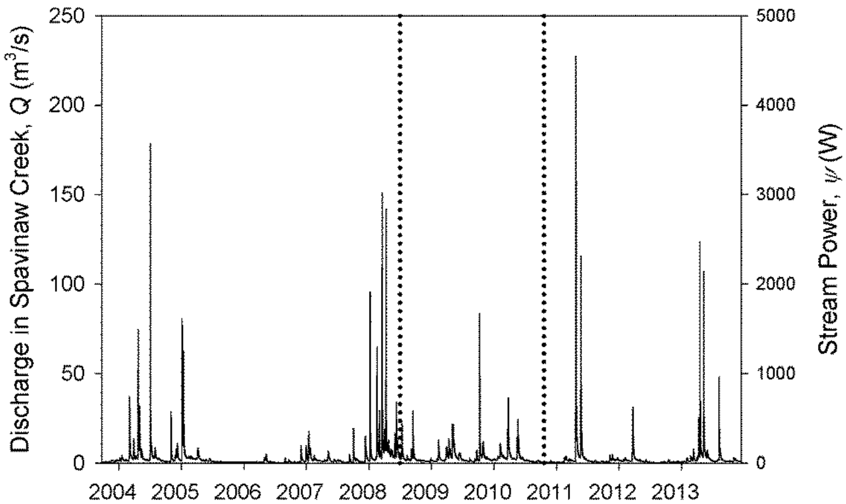


Figure 2.2. Daily discharge (USGS) and stream power for Spavinaw Creek from 2003-2013.

The Weibull (1939) relationship (Haan et al., 1991) was used to estimate the discharge associated with the following storm event return periods: 1, 1.5, 2, 5, 10, and greater than 10 years. The number of storm events less than each return period and total stream power per study period were calculated (Table 2.1) for comparison to measured erosion rates during the same time periods as the available aerial imagery, as discussed below.

Table 2.1. Number of storm events per time period and cumulative stream power.

Period	Number of Storm Events less than each Return Period						Power (W)
	<1 yr	<1.5 yr	<2 yr	<5 yr	<10 yr	>10 yr	
2003-2008	749	72	12	2	3	1	85,569
2008-2010	563	36	2	1	0	0	43,545
2010-2013	377	22	11	3	1	1	44,340
2003-2013	1,689	130	25	6	4	2	173,454

2.3.3 Aerial Imagery Analysis

Generally, in the field, channel cross-sections and the implementation of bank pins are the most widely used and accepted methods for measuring bank erosion (Lawler, 1993). Cross-section surveys allow for the measurement of erosion and deposition, but do require a permanent base point, take more time, and only show the changes at the specified site. Bank pins have accurate measurements (± 3 mm) in alluvial material and allow for quick readings, but may be limited in the amount of erosion that can be detected (Hooke, 1979; Lawler, 1993). Systems with significant episodic retreat events can be difficult to assess with bank pins (Miller et al., 2014). Recently, studies including those by Brice (1982), Odgaard (1987), Beeson and Doyle (1995), Harmel et al. (1999), Micheli and Kirchner (2002), Heeren et al. (2012), and Miller et al. (2014) have used aerial imagery to estimate bank erosion. Aerial images allow for the long-term analysis of a channel and the rapid evaluation of changes over a large area. Aerial imagery analyses are generally faster than *in situ* methods, but do have an increased error. Due to the episodic nature of this watershed, it was decided to use aerial imagery for the erosion analysis.

Aerial images of Delaware County, OK for 2003 (September 23), 2008 (July 2), 2010 (October 21), and 2013 (October 15) were obtained from the National Agricultural Imagery Program (NAIP), each with 1 m horizontal resolution. Each image was georeferenced in ArcMap 10 (ESRI, 2014) and then used to estimate bank erosion. The first step in quantifying erosion was to trace both sides of the streambank on each image (Figure 2.3). The traced streambank line followed the line of the established streambank, neglecting the formation of gravel bars. Left and right bank polylines were created and georeferenced in ArcMap 10 by manually tracing the streambanks of Spavinaw Creek for the four images.

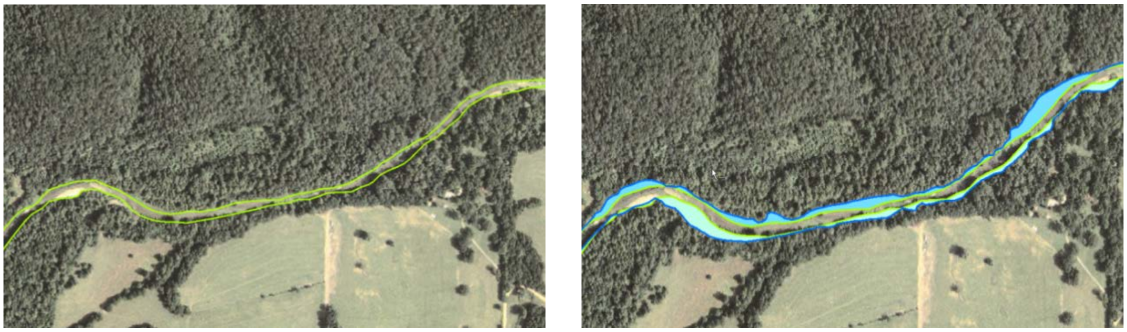


Figure 2.3. Left and right bank (green lines) for Spavinaw Creek on the 2003 NAIP aerial image (left), and creation of polygons (shaded blue area) to determine erosion from 2003-2008 along Spavinaw Creek (right) using ArcGIS.

The ‘join’ function was used in ArcMap to combine the banks from the beginning and end of the specific study period. For example, in order to determine erosion between 2003 and 2008, the 2003 bank line was compared to the 2008 bank line to identify erosion areas. After connecting the bank lines, the new feature was converted into a polygon (Figure 2.3). The purpose of this step was to find the area between the lines as polylines cannot have an area. Through the attribute table of the polygon, the area of each segment along the streambank was calculated. All polygons were analyzed individually to determine if it was erosion or the deposition of a gravel bar. Each polygon that represented erosion was noted, giving a total area of erosion along the streambank. It was noted that the eroded bank retreat using aerial imagery is not as accurate as *in situ* procedures, with an estimated error on the order of 1 m in retreat based on the georeferencing and identification of banks (Heeren et al., 2012).

2.3.4 Calculating Eroded Area

The total eroded area (m^2) during each period was determined. Then creek-averaged lateral retreat was calculated as the eroded area divided by the creek length. Total sediment loading (SL, kg or kg yr^{-1}) into the stream was obtained using the following formula:

$$SL = (CR)(D_{ts})(\rho_b) \tag{2}$$

where CR is the creek-averaged lateral retreat (m), D_{ts} is the average depth of the topsoil derived based on measurements at five sites along Spavinaw Creek (m), and ρ_b is the average bulk density of the soil (kg m^{-3}). The D_{ts} and ρ_b were derived from measurements at five sites along the stream. This method was carried out for three time periods corresponding to the aerial imagery dates: 2003-2008, 2008-2010, and 2010-2013.

2.3.5 Riparian Vegetation

One key component of streambank erosion and failure has been suggested to be the absence of established riparian vegetation. Vegetation analysis was conducted using the processed NAIP images at randomly selected points (n=100) along the creek. At each point, the width of the eroded polygon area was collected and the near-stream buffer width was estimated. Near-stream buffer was considered to be established trees along the edge of the streambank. Sand or grasses had a buffer width of zero. The eroded width, or retreat length (RL), was divided by the number of years in the specific time period to determine RL per year. The yearly RL was plotted versus the buffer width to analyze for trends for the three time periods: 2003-2008, 2008-2010, and 2010-2013.

2.3.6 Influence of Meandering

Another aspect that may influence erosion rates is sinuosity or meandering. Streambanks along the outside of a meander bend experience higher shear stress (Leopold and Langbein, 1960), which can lead to increased erosion. The aerial images were used again, but focused on measuring bank retreat at meanders. First, a circle was visually fit to each bend along the previously traced streambank line in ArcGIS. This circle was originally placed within the meander and enlarged until it touched both sides of the bend (Figure 2.4). The radius of the circle was then obtained from the object properties as the site's radius of curvature (ROC). The 'measure' tool in ArcMap was then utilized to estimate the bank retreat length from one study period bank to the later bank. All measurements for erosion were taken at the approximate centerline of the bend. Retreat length and corresponding ROC were compared with erosion rate.

Figure 2.4. Measuring the meander radius of curvature and retreat length using ArcGIS.

2.3.7 Random Site Sampling Analysis

Current stream monitoring procedures usually select a few sites to continuously monitor in order to determine erosion rates and stream behavior for the entire length of a stream (Rosgen, 2001). However, scaling from a couple of sites to a watershed may not be represent the actual bank retreat. For each time period, a total of 1, 2, 3, 5, and 10 random sites were selected along the creek. In order to select the sites, the total number of erosion polygons was noted and a random number generator was used to select which polygon(s) would serve as the study site(s). At each selected site the eroded area and bank length was estimated. From this eroded area the following data were calculated: yearly eroded area, average bank retreat, and the total and yearly eroded soil mass and volume. The ratio of sampled streambank length to total streambank length was calculated and used to scale up the eroded area from the random sites to represent the amount of erosion expected along the entire stream.

2.4 Results and Discussion

2.4.1 Overall Erosion

Analysis of the NAIP imagery from 2003, 2008, 2010, and 2013 showed varying bank retreat and soil loading rates during each study period. For each time frame, the average bank retreat rate, eroded soil per year per meter of bank, eroded soil mass per year, and total eroded soil mass were calculated based on the erosion polygons (Table 2.2). The 2008-2010 period had the lowest retreat and loading, and 2010-2013 had the highest. In order to better understand the trends, the stream discharge (Figure 2.2) was analyzed. Erosion and retreat during each analysis period was highly correlated (α=0.05) with the discharge rates. In the 2008-2010 period, there were primarily 1-yr storm events (Table 2.1), resulting in low stream power acting on the streambanks. Under high flow events, the streambanks can become easily saturated and less stable. As previously mentioned, the alluvial floodplain watershed has an underlying gravel layer. High stream power erodes this gravel toe over time, causing the overlying soil to become geotechnically unstable and collapse into the stream. The high bank retreat and soil loading in 2010-2013 corresponded to large storm events and the associated stream power.

Table 2.2. Bank retreat, and volume and mass of eroded soil during the three study periods for Spavinaw Creek.

Period	Average Bank Retreat (m yr^{-1})	Eroded soil per yr per m of bank (kg x 10^3 m^{-1} yr^{-1})	Eroded soil mass per year (kg x 10^6 yr^{-1})	Total eroded soil mass (kg x 10^6)
2003-2008	2.4	3.3	66	328
2008-2010	0.7	1.0	21	41
2010-2013	4.3	5.9	119	358
2003-2013	2.5	3.4	68	727

The behavior between the overall stream power and average bank retreat is important to better understand the erosion mechanisms. A positive relationship between stream power and bank retreat was not present during the 2010-2013 period (Figure 2.5). This period did have the most bank retreat and soil loadings, but it had less cumulative power compared to the 2003-2008 period. This period had a few large flow events within a few days that may have caused the large-scale erosion events (Figure 2.2). Fluvial erosion may play a dominate role in this system, as shown from analysis of the 2003-2008 and 2008-2010 time periods; however other mechanisms, such as mass wasting, may also have a major impact on streambank erosion along Spavinaw Creek.

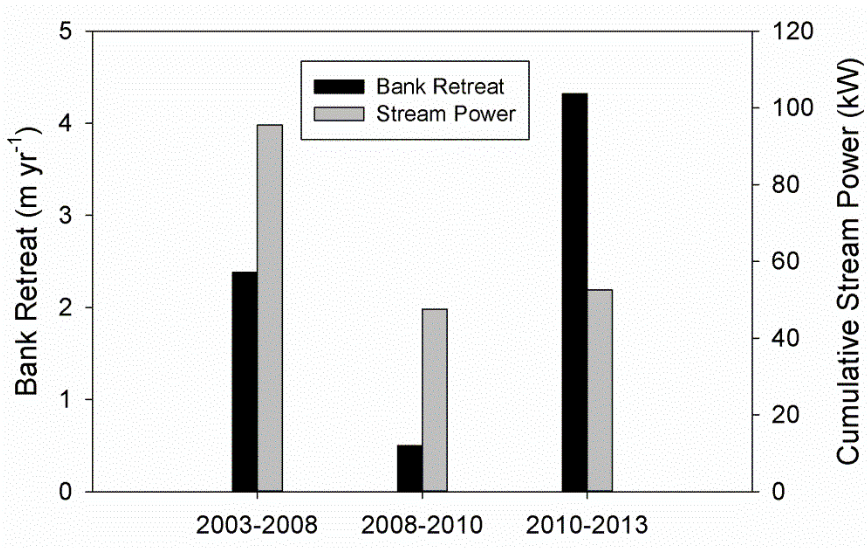


Figure 2.5. Relationship between average bank retreat and cumulative power for the three study periods.

2.4.2 Influence of Riparian Vegetation

A statistical analysis using SigmaPlot (v12.5) (SPSS, 2011) was performed to test significant differences between the annual and total retreat on vegetated banks versus those without vegetation. Box and whisker plots were created for each time period for the average annual retreat and the total bank retreat with and without a buffer (Figure 2.6). P-values were calculated for average retreat and annual retreat during the three time periods using the Mann-Whitney Rank Sum Test. This nonparametric test was used since the annual retreat rate and total retreat data were not normally distributed. There was a statistically significant difference ($p<0.001$) in bank retreat between vegetated and unprotected banks for 2003-2008 and 2010-2013, but not for 2008-2010 ($p>0.050$). Referring back to the stream discharge (Figure 2.2), this time period had the least number of large storm events, and therefore the least individual and total stream power. Therefore, this insignificance may be linked to the lower applied shear stress and/or less mass wasting. A two-way ANOVA (Appendix B) was performed in Minitab v16 (Golden Software, 2002) to analyze the influence of two independent variables. For this study the independent variables compared were the presence of a buffer and the time period. It was found that the presence or absence of a buffer ($p=0.000$), the time period ($p=0.000$), and the interaction between the two ($p=0.004$) were significant in the estimated bank retreat. A Tukey's Multiple Comparison Test (Appendix B) was performed and showed that the 2010-2013 period was statistically different to the other periods. This may be attributed to the difference seen for the trend been bank retreat and stream power during this time period.

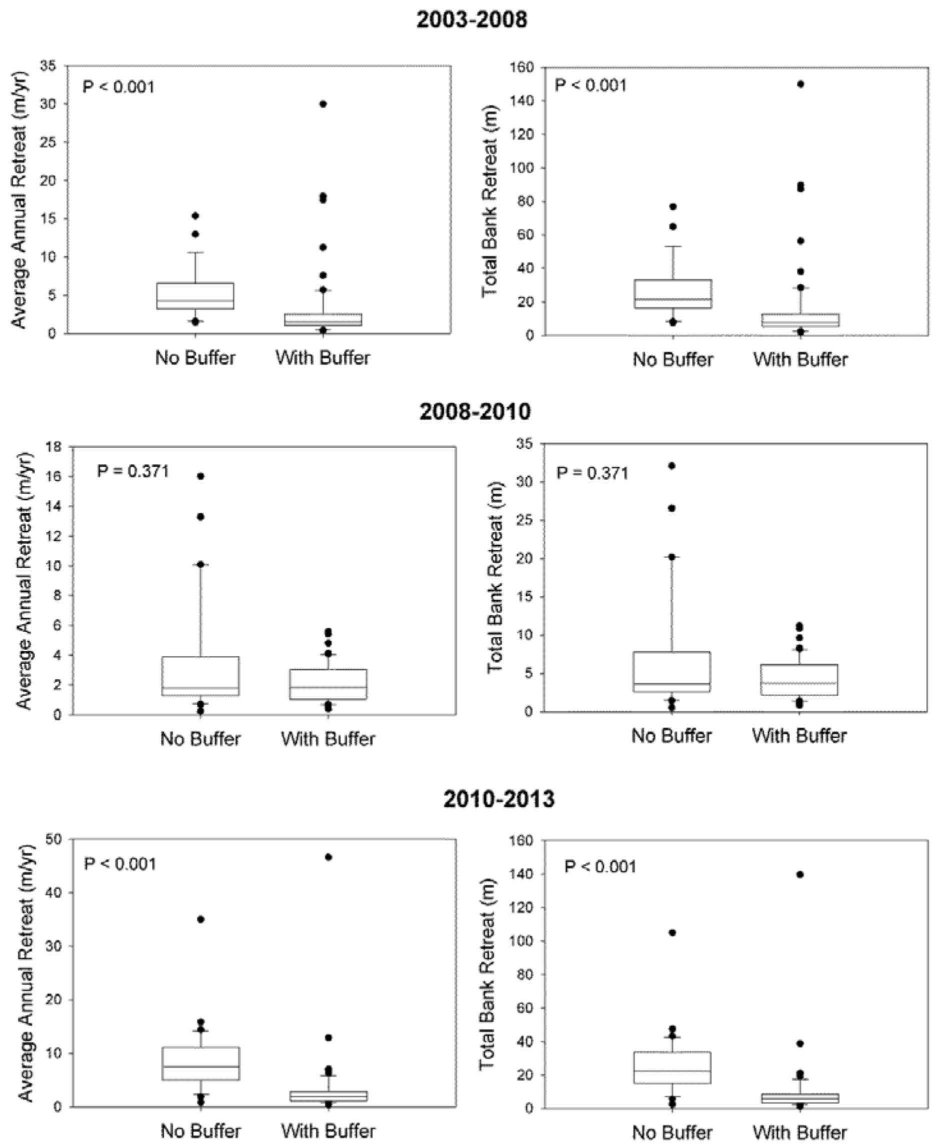


Figure 2.6. Box and whisker plots of average and total retreat lengths for streambanks with and without a vegetated cover during 2003-2013.

The vegetated banks during the time periods 2003-2008 and 2010-2013 had approximately three times less bank retreat compared to those with no vegetation. This three times reduction in bank retreat with riparian vegetation is similar to a study conducted by Miller et al. (2014) for similar composite streambanks in the Illinois River Watershed in eastern Oklahoma. These reductions are also comparable to those found in Beeson and Doyle (1995), Harmel et al. (1999), and Simon et al. (2009).

Corroborating Daly et al. (2015), these data and the results of the two-way ANOVA demonstrate the need to consider the monitoring period when assessing bank retreat rates. For example, based on monitoring data from 2008-2010, no clear difference would be observed due to the presence or absence of a riparian buffer; both conditions had approximately 2 m yr^{-1} retreat rates. However, this retreat rate is comparable to that seen on streambanks with a riparian buffer during 2003-2008 and 2010-2013, while streambanks without riparian vegetation experienced greater than 4 m yr^{-1} of bank retreat in 2003-2008 and more than 7 m yr^{-1} of bank retreat in 2010-2013. Therefore, the presence of established riparian vegetation can have three times less bank retreat along composite banks characteristic of the Eucha-Spavinaw Watershed, but may also be dependent on the study period as stream power and mass wasting vary temporally.

2.4.3 Influence of Meandering

As mentioned earlier, streambanks along a meander tend to experience higher power and shear stress compared to the inside of a bend. A subset of meanders present during each time period were analyzed for their ROC, and the bank retreat and ROC were negatively correlated (α=0.05) (Figure 2.7). A small radius of curvature was representative of a tighter bend in the creek, which experienced higher shear stress along the banks as the energy could not be dissipated over a longer stretch of bank. Therefore, a meander with a lower ROC should experience a higher bank retreat. Analysis of meanders along Spavinaw Creek over the 10-yr period agreed with this hypothesis showing an exponentially declining relationship between ROC and bank retreat. However, considerable variability was still present in these data, once again suggesting there are multiple processes occurring that contribute to the overall bank retreat.

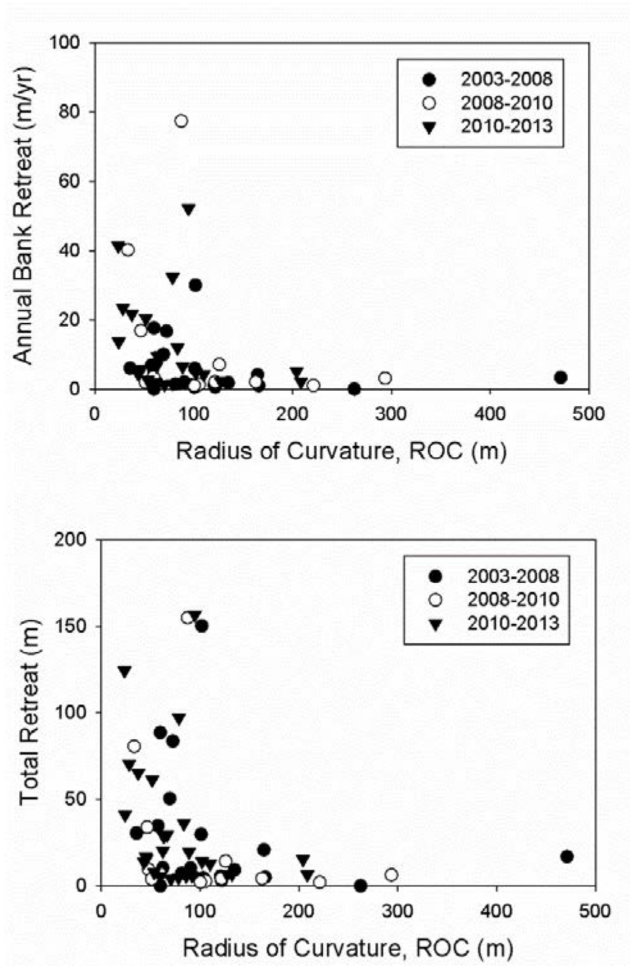


Figure 2.7. Radius of curvature and corresponding annual and total bank retreat for creek meanders during 2003-2013.

2.4.4 Influence of Sampling Location

Scaled up estimates from a few sampling or monitoring points to a watershed were compared to the estimated total load from the ArcMap analysis for different numbers of random sites for all three periods. Almost all instances of random sampling monitoring provided estimates that predicted 80% or less of the total measured erosion (Figure 2.8). Only one of fifteen trials overestimated total erosion, but it predicted almost two times more soil loading than was measured.

Currently, many stream monitoring plans consist of only measuring erosion at a few sites along the entire stream, and then those measurements are used to estimate sediment loading for the entire stream system (Miller et al., 2014). From this research, only monitoring a few sites can lead to large underestimation of actual erosion loads. If stream restoration and erosion mitigation is being implemented on a per site basis only, then individual monitoring will be representative. However, if determining loads for a creek or watershed to aid in the implementation of reduction measures, it is clear that a small number of sites was not effective in representing true behavior. Other methods should be considered, such as more monitoring sites or using aerial imagery and discharge data to determine historical trends.

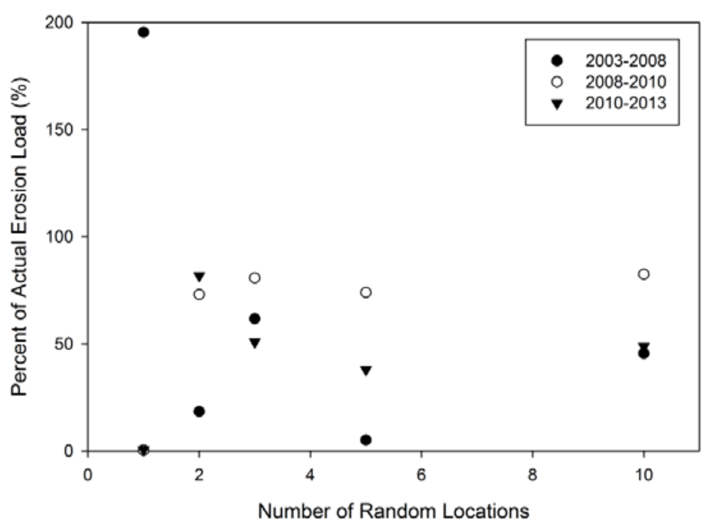


Figure 2.8. Percent of total yearly eroded soil mass per m per year from 2003-2013 as a function of the number of randomly selected monitoring locations.

2.5 Conclusions

In the Spavinaw Creek watershed, total streambank erosion was estimated from 2003 to 2013 was 727 x 10^6 kg and the average bank retreat was 2.5 m yr^{-1}. Numerous processes come into play that control streambank erosion and failure. The relationship between stream power and bank retreat showed that fluvial erosion does play an important role within the Eucha-Spavinaw watershed, but also that mass wasting may contribute to large erosion events. The presence of riparian vegetation along a streambank reduced the annual bank retreat by approximately three times compared to an unprotected bank. This observation is comparable to other stream systems, such as the Barren Fork Creek in the Illinois River Watershed. Monitoring only a few sites to determine erosion rates would be effective if determining the behavior at a singular location, but was usually not representative when scaled up for an entire creek. The use of aerial imagery and historic flow data has shown to be an acceptable method to estimate erosion along an entire creek, but may be limited by the ability to accurately identify the bank locations, making its use more appropriate in larger order streams. Current numerical models that focus solely on fluvial erosion may not accurately predict streambank erosion. This calls for the use of process-based models that incorporate fluvial erosion, mass wasting, and subaerial erosion. The ability to better understand the behavior and effects of different variables, such as buffer width and radius of curvature, within a system provides improved stream management and the design of effective conservation practices.

2.6 Acknowledgements

The author acknowledges the financial assistance of a FY 2012 319(h) Special Project #C9-00F56701.

CHAPTER III

ESTIMATING STREAMBANK PHOSPHORUS LOADS: HOW MUCH PHOSPHORUS DO STREAMBANKS CONTRIBUTE AT THE WATERSHED SCALE?

3.1 Abstract

Nutrient and sediment loading from streambanks are a growing concern within many watersheds. However, there are only a few studies on streambank phosphorus (P) concentrations and spatial distributions in watersheds. The objectives of this research included (i) quantifying the magnitude of and spatial distribution of streambank phosphorus concentrations along a stream system in a watershed with historical poultry litter application, (ii) quantifying the amount of water soluble phosphorus (WSP) and total phosphorus (TP) entering the stream from streambanks, and (iii) comparing streambank P concentrations and loading between two unique streams in the same ecoregion. Standard soil sampling methods were used at five sites along Spavinaw Creek in eastern Oklahoma, and samples were processed to measure pH, electrical conductivity (EC), WSP, and TP. There was no clear longitudinal trend in WSP, TP, pH, and EC. Using estimated sediment loading (727 x 10^6 kg) from aerial images, it was estimated from 2003-2013 there was 1.5 x 10^3 kg WSP and 1.4 x 10^5 kg TP loaded into Spavinaw Creek from streambanks in Oklahoma. LOADEST, a nutrient load estimator created by the United States Geological Services (USGS), was used to estimate in-stream phosphorus loads. In-stream estimates were an order of magnitude larger for WSP and comparable for TP. LOADEST estimates loads from multiple

sources, both point and nonpoint, not only streambank erosion. A previous study performed a similar analysis along Barren Fork Creek (BFC) in the Illinois River watershed. Both Spavinaw Creek and BFC flow through the Ozark ecoregion and have cherty topsoil with an underlying gravel layer. WSP loading in BFC was an order of magnitude higher than in Spavinaw, while TP loadings were on the same order of magnitude. Streambank P loading rates are dependent on the stream and watershed; therefore each stream needs to be individually studied in order to gain a better understanding of the specific loadings from streambanks.

3.2 Introduction

Nutrient and excess sediment are two of the primary pollutants of surface waters in the United States. An increase of phosphorus in surface water can lead to algae blooms. As these blooms die and decompose, dissolved oxygen in the water decreases, which can negatively impact the aquatic ecosystem (Pierzynski et al., 2000). For example, in recent years the Tulsa Metropolitan Utility Authority has seen an increase in Spavinaw Lake water treatment costs. Their consumers have also complained about the taste and odor (Tortorelli, 2006). These changes have been linked with an increase in nutrient levels within the lake (Tulsa Metropolitan Utility Authority, 2001).

Storm et al. (2002) found that the total mass of nutrient sources in the catchment, in decreasing order, were land application of poultry litter, rural municipal wastewater treatment plants, agricultural row crops, and naturally occurring sources. However, in many cases, there are insufficient data to determine sediment and nutrient loading from streambanks on a watershed scale. Streambank erosion and failure may be a significant pathway for P loading to streams but insufficient data exists on streambank sediment concentrations and P loading for this and many other watersheds. The objectives of this research included quantifying the magnitude of and spatial distribution of streambank phosphorus concentrations along a stream system in a watershed with historical litter application and to then use these concentrations to quantify the amount of water soluble phosphorus (WSP) and total phosphorus (TP) entering the stream from streambanks.

Understanding soil conditions and quantifying phosphorus loading is necessary for determining the need for and justifying the use of protective measures, such as riparian vegetation (Sekely et al., 2002; Laubel et al., 2003; Kronvang et al., 2012). Triplett et al. (2009) found P loading in the St. Croix River to have increased eight-fold from 1850 to the 1950s and noted that the increase in loading had a close correlation to the sediment. Zaimes et al. (2008) compared the soil and P losses to surface waters from various land practices and found that land with riparian forest buffers had the lowest contribution. However, a question arises as to whether one study on streambank P loading in one area can be applied to other similar stream systems. Are P loads to streams in the same ecoregion and with similar streambank characteristics comparable and applicable to others in the region?

Gaining an understanding of streambank P concentrations and spatial distribution within a system is complex and important, but must go hand-in-hand with studying fluvial processes and erosion behavior. Each individual study can show important characteristics and combining them will help show a more complete picture. For example, Miller et al. (2014) quantified erosion and sediment loading at ten sites along the Barren Fork Creek and reported P concentrations within these ten streambanks. By relating the two the TP loading to the surface water was determined. Phosphorus loading estimations have major implications for evaluating the effectiveness of different management practices. Therefore, one of the objectives of this study was to compare streambank P concentrations and loading between two unique streams in the same ecoregion.

3.3 Materials and Methods

3.3.1 Soil Sampling and Processing

Soil samples were taken at five points along Spavinaw Creek to quantify phosphorous concentrations within the streambank (Figure 3.1). Following Miller et al. (2014), at each site samples were taken at three transects per bank at three vertical depths: near the bank surface (typically 12 cm below ground surface), middle of the topsoil depth (typically 30 cm below ground surface), and at the streambank toe (typically 70 cm below ground surface). At each depth a core was taken horizontally into the streambank. The core was then divided into five lateral depths: 0-5 cm, 5-10 cm, 10-20 cm, 20-50 cm, and 50-100 cm. Site characteristics can be found in Table 3.1.

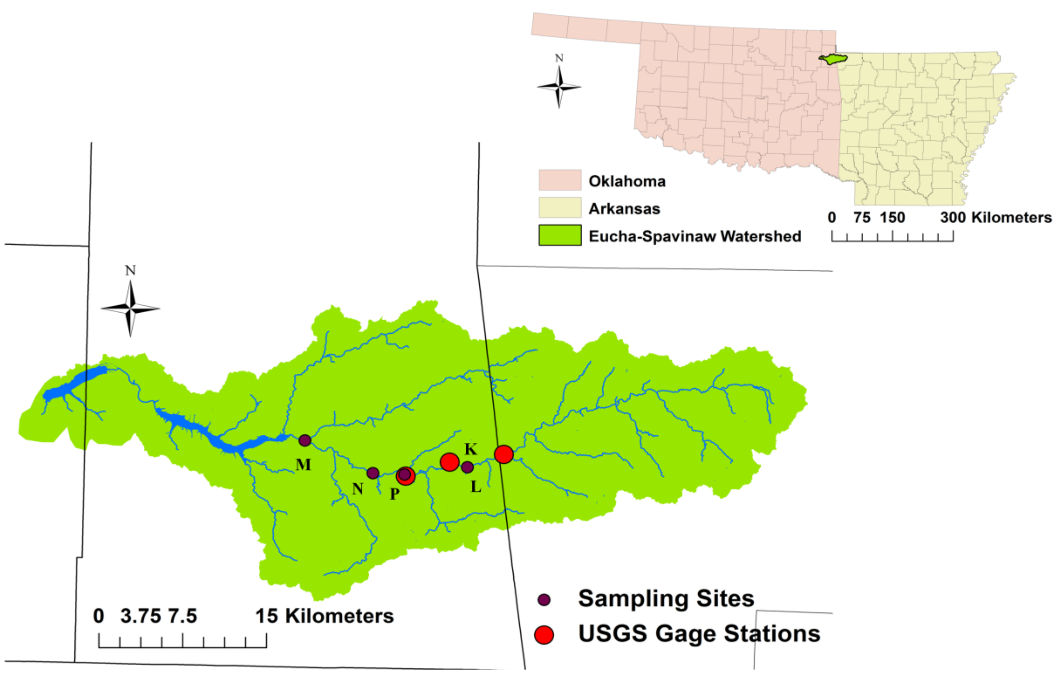


Figure 3.1. Eucha-Spavinaw watershed showing soil sampling and USGS gauge station locations.

Table 3.1. Sample characteristics for the five soil sampling points along Spavinaw Creek.

Site	Sample Date	Soil Type	Distance from Confluence (km)	Topsoil Depth (cm)	Bulk Density (kg/m^3)	Land Use
K	June 2013	Gravelly silt loam	20	70	1450	Pasture/Riparian Woodland
L	June 2013	Gravelly silt loam	19	70	1430	Pasture/Riparian Woodland
M	June 2013	Gravelly silt loam	1	60	1500	Riparian Woodland
N	November 2014	Gravelly silt loam	9	25	1450	Pasture/Riparian Woodland
P	November 2014	Gravelly silt loam	12	30	1450	Pasture/Riparian Woodland

Samples were characterized for WSP, TP, degree of phosphorous saturation (DPS), and general soil properties (e.g., pH and EC). Soil pH was measured by adding 5 g of soil and 15 mL of deionized water (DI water) to a vial in order to obtain the 1:3 (soil:water) ratio required for the test. Samples were shaken for 60 s to ensure that the soil was thoroughly wet. Then they were left for 20 min to equilibrate and then shaken for another 60 s. After another 20 min period of equilibration the calibrated pH probe was placed in the mixture. After the probe stabilized, the reading was taken to the nearest 0.01. Between each sample the probe was rinsed with DI water. The same sample preparation procedure was used for electric conductivity (EC), using a calibrated EC meter (Smith and Doran, 1996). The EC is associated with the salt content of fertilizer application, where an increase in associated fertilizer salts increases soil EC (Miller et al., 2014).

WSP is an index of the amount of phosphorous associated with the soil that will enter a dissolved state should the soil be eroded into the stream (Pote et al., 1996; Fuhrman et al., 2005). WSP was measured by adding 1 g of soil and 10 mL of DI water in a 50 mL centrifuge tube to obtain the 1:10 (soil:water) ratio required for the test. Samples were placed on a shaker table for one hour on the low setting. Then, samples were centrifuged for 5 min at 2000 rpm. Following this samples were put through a vacuum filtration system. The collected extract was sent to the

Oklahoma State University Soil, Water, and Forage Analytical Laboratory for an inductively coupled plasma mass spectrometry (ICP-MS) analysis on P.

TP is a measure of all forms of P in the soil. TP was measured using the US EPA method 3050b (US EPA, 1996), in which 2 g of soil subsample is heated in a solution of HCl and nitric acids. The solution is then filtered through Whatman #40 filter paper. Extracts were sent for ICP-MS analysis for P.

DPS (%) is an index of the potential for soils to sorb or desorb P. DPS expresses the P concentration relative to the concentrations of the aluminum (Al) and iron (Fe) oxides that serve to bind P in soil. The P sorption capacity is controlled by the presence of amorphous Al and Fe oxides (van der Zee and van Riemsdijk, 1988; Lookman et al., 1995; Guo and Yost, 1999). Previous studies have found the potential for soils to release P to runoff or by leaching to surface water by erosion can be indicated through the DPS value (Sharpley, 1995; Leinweber et al., 1999; Pautler and Sims, 2000; Maguire and Sims, 2002; Sims et al., 2002). An increase of soil DPS shows an increased desorption of P to water. DPS was calculated using the following equation:

$$DPS = \left[\frac{P_{ox}}{Fe_{ox}+Al_{ox}}\right]100\% \tag{1}$$

where P_{ox} is the oxalate-extractable P (mol kg^{-1} soil) , Al_{ox} is the oxalate-extractable Al (mol kg^{-1} soil), and Fe_{ox} is the oxalate-extractable Fe (mol kg^{-1}). These values are determined by adding 1 g of soil and 40 mL of ammonium oxalate extracting solution in a 50 mL centrifuge tube. Samples were then placed on a shaker table in the dark for two hours on the low setting. Then, samples were then centrifuged for 13 min at 2000 rpm. Following this extracts were filtered through Whatman #42 filter paper. Extracts were sent for ICP-MS analysis for P, Al, and Fe as outlined by Schoumans (2000).

The mean and standard deviation were calculated for pH, EC, WSP, and TP for each of the five sampling sites. Box plots were created showing the variation between sites relative to their distance from the Spavinaw-Beaty confluence. The two-dimensional spatial distribution of WSP and TP were plotted using the depth below ground surface and depth into the streambank as the spatial coordinates. Data from each of the three transects were combined to create one plot per sample site. Contour plots were created using the Kriging gridding method in Surfer 8 (Golden Software, 2002). These plots were used to investigate intra- and inter-site variability.

Particle size analysis (ASTM 422, 2002), carried out on composited samples from each site, found that the streambanks are composed of silt loam. Size class fractions ranged from 25-38% sand, 50-70% silt, and 5-11% clay. These findings are consistent with the United States Department of Agriculture Natural Resources Conservation Service (USDA-NRCS, 2013) Web Soil Survey, which showed that soil along the Spavinaw Creek streambanks was a Clarksville series, which is a gravelly silt loam.

3.3.2 Estimating Streambank P Loading

Aerial images of Delaware County, OK for 2003 (September 23), 2008 (July 2), 2010 (October 21), and 2013 (October 15) were obtained from the National Agricultural Imagery Program (NAIP), each with 1 m horizontal resolution. Erosion from 2003-2013 was derived using aerial photography and processed in ArcMap to quantify eroded area. The total area (m^2) of streambank erosion during each period was calculated. The creek-averaged lateral retreat (*CR*, m) was calculated as the eroded area divided by the creek length. Total sediment loading (*SL*, kg or kg yr^{-1}) into the stream was calculated:

$$SL = (CR)(D_{ts})(\rho_b) \qquad (2)$$

where *CR* is the creek-averaged lateral retreat (m), D_{ts} is the average depth of the topsoil derived based on measurements at five sites along Spavinaw Creek (m), and ρ_b is the average bulk density of the soil also derived from measurements at five sites (kg m^{-3}). This method was carried out for three time periods: 2003-2008, 2008-2010, and 2010-2013. It was estimated that the total soil mass eroded from 2003 to 2013 was 727 $x10^6$ kg and the average bank retreat was 2.5 m yr^{-1}.

One key component of streambank erosion and failure has been suggested to be the presence of established riparian vegetation. Vegetation analysis used the processed NAIP images. Random points (*n*=100) were selected along the creek. At each point, the width of the eroded polygon area was collected and the near-stream buffer width was estimated. Sand or grasses had a buffer width of zero. The eroded width, or retreat length (RL), was divided by the number of years in the specific time period to determine RL per year. It was estimated that banks with riparian vegetation had approximately three times less bank retreat than unprotected banks from 2003-2013.

The WSP and TP loads were calculated by multiplying the total mass of eroded topsoil (*SL*, kg or kg/yr) and the average *WSP* (WSP_{avg}, mg WSP/kg soil) or *TP* (TP_{avg}, mg TP/kg soil):

$$Total\ WSP = WSP_{avg} \times SL \quad (3)$$

$$Total\ TP = TP_{avg} \times SL \quad (4)$$

A Monte Carlo simulation was performed to estimate the variation in estimated WSP and TP loads by accounting for parameter uncertainty. A Monte Carlo simulation is a statistical method that uses sample means to estimate population means (Dunn and Shultis, 2012). Each independent variable has a range of potential scenarios that are not represented in a single estimated value. Evaluating the data involves determining the statistical distribution model, creating probability distributions on the input parameter, and obtaining a final output distribution. The Monte Carlo method uses multiple simulations, each with different estimated data to draw a statistical conclusion (Sabbagh and Fox, 1999). A Monte Carlo simulation shows where the measured sample mean falls within many scenarios, quantifying the uncertainty. Four independent variables were analyzed: WSP or TP in streambank soil, soil bulk density, top soil depth, and eroded area. For each variable, the measured data were analyzed in Minitab 16 (Minitab, 2009), where a goodness-of-fit test was performed to determine which distribution best fit the observed data. Based on the null hypothesis, H_0: the model adequately described the data, a model was considered an acceptable fit if the p-value was greater than 0.05 (95% confidence) and had the lowest Anderson-Darling (AD) statistic compared to the other distributions. The parameters of the probability density function (PDF) and cumulative density function (CDF) for quantifying parameter uncertainty were obtained for the best fit distribution.

According to Driels and Shin (2004), a total of 1000 simulations are required to achieve a 95% confidence level. For each variable 1000 random numbers between 0 and 1 were generated. The random number was set equal to the CDF, or F(X), in the corresponding distribution equation, which was used to calculate the estimated independent variable (WSP, TP, EA, D_{ts}, ρ_b) for each individual simulation. The 1000 Monte Carlo simulation estimates for each variable were used to calculate the potential WSP loading using equation (3). The same procedure was carried out for TP loading using equation (4).

3.3.3 Estimating In-stream P Loads

LOAD ESTimator (LOADEST) (Runkel et al., 2004) is a FORTRAN program created by the USGS that is used to estimate constituent loading in surface water. LOADEST requires the discharge and constituent concentration over time in order to derive a regression model for estimating potential loading. Historical discharge and P in the unfiltered water, which is split into dissolved P (DP) and total P, was obtained for the 10 yr study period from the Water Quality Portal (USGS, EPA, and NWQMC) and input into LOADEST. Data were collected from the USGS gauge on Spavinaw Creek near Cherokee City, AR (USGS 07191179), located near the

Arkansas-Oklahoma border, the gauge on Spavinaw Creek near Colcord, OK (USGS 071912213), halfway along Spavinaw Creek in Oklahoma, and the gauge on Spavinaw Creek on Sycamore (USGS 07191220), which is between the other two gauges (Figure 3.1). TP and DP loads from the three gauges and their distance from the confluence were used to extrapolate and estimate the concentrations at the confluence. The difference between the confluence estimate and the estimate at the Arkansas-Oklahoma border providesd an estimate in the loading contributed by all sources in Oklahoma. Estimated in-stream DP and TP loads were compared to the estimated ranges from the in-streambank WSP and TP Monte Carlo simulations.

3.4 Results and Discussion

3.4.1 Environmentally Sensitive Phosphorous Concentrations

As previously stated, excess P can pose an environmental threat to the quality of the water and streambanks can act as a potential source. Soil thresholds have been suggested that represent an increased potential for releasing P into water. Suggested thresholds are soils with DPS values greater than 25% (Breeuwsma et al., 1995) and/or greater than 8.2 mg WSP kg^{-1} soil (Sims, 1993). These thresholds were obtained for specific soil types, focused on upland soils, and will vary with soil chemistry. Therefore, they are implemented for a general comparison, but may change with each study. Analysis of measured WSP and calculated DPS shows that only 5% of sampled streambank points are considered to currently be a potential risk with WSP over 8.2 mg WSP kg^{-1}, but DPS lower than 25% (Figure 3.2). The top right quadrant (region above both thresholds) is considered the region of environmentally sensitive soil P concentrations that will pose an immediate threat to water quality. There was no clear trend in this data, which may be due to differences in variables including soil mineralogy, sampling depth, and pH. Miller et al. (2014) reported that in the Barren Fork 14% of samples were above the WSP threshold, 25%

above the DPS threshold, and 13% were above both of the thresholds. These streambanks were also characterized by gravelly silt loam, showing that soil type was not the only factor in determining the potential for P sorption and release. For example, soil pH, cation concentration, soil organic matter, and temperature could all affect the P concentrations.

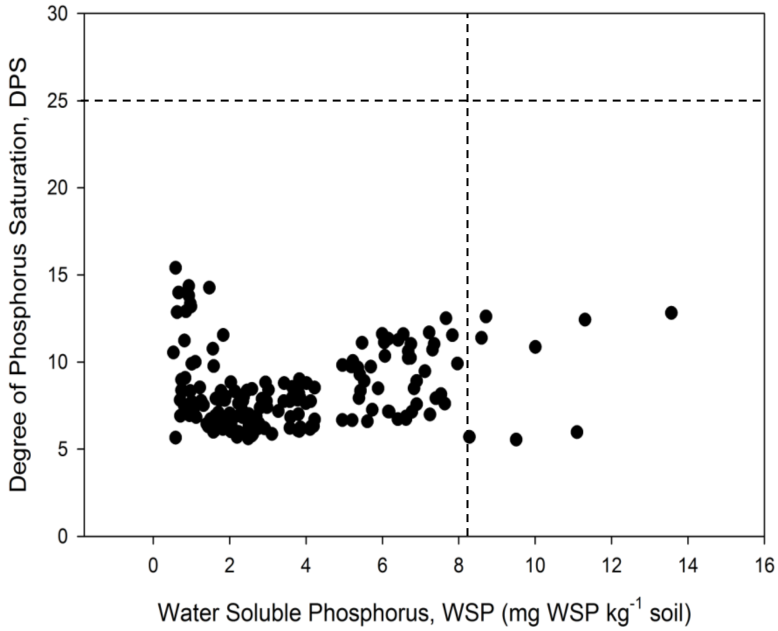


Figure 3.2. Environmental sensitivity of streambank soil samples from all five sampling sites along Spavinaw Creek.

3.4.2 WSP and TP Streambank Spatial Distributions

Contour analysis of the WSP concentrations (Figure 3.3) suggested no consistent visual pattern in WSP concentrations between the five sites. Site K had WSP levels less than 2.6 mg kg^{-1} throughout the cross-section, while site L, just downstream, had much higher levels, especially near the surface. Site M had higher concentrations of WSP just in a strip below the surface, possibly due to WSP leaching from the surface further upstream and moving through the lower topsoil layers or from high flow events where WSP could leave the creek and sorb onto soil particles. Another potential cause could be the historical movement of sediment. Sediment of different characteristics could have been deposited as a bank was forming, creating a layer of different texture that has the potential to better sorb P. Site N had a hot spot of WSP where the topsoil met the underlying alluvial gravel layer. This was most likely caused by the saturation of the gravel layer, allowing for transfer of WSP within the creek onto the bottom layer of topsoil through subsurface preferential flow paths (Heeren et al., 2011; Miller et al., 2014). This hotspot could also be due to the movement of sediment with legacy P that has been deposited over time and created a bank high in WSP. Site P had no particular trends or peaks.

TP was also spatially analyzed (Figure 3.4). Site L and M had similar trends for TP as they did with WSP, with the higher concentrations at the top and just below the surface, respectively. These high concentrations may be due to anthropogenic activities as high concentrations were seen near the surface, or could be the deposition of sediment over time that was high in TP. Site K had higher TP values around 40 cm below the ground surface, but comparison to the WSP plot showed that another form of P, such as plant available, was dominating. Site N had high TP towards the streambank face, which generally followed the WSP behavior. Site P had a peak of TP at the interface between the topsoil and underlying gravel, possibly due to P in the creek sorbing onto this exposed topsoil layer. Generally TP will not leach and be mobile like WSP, so streambank concentrations were typically hypothesized to be a result of sediment deposition or anthropogenic activities.

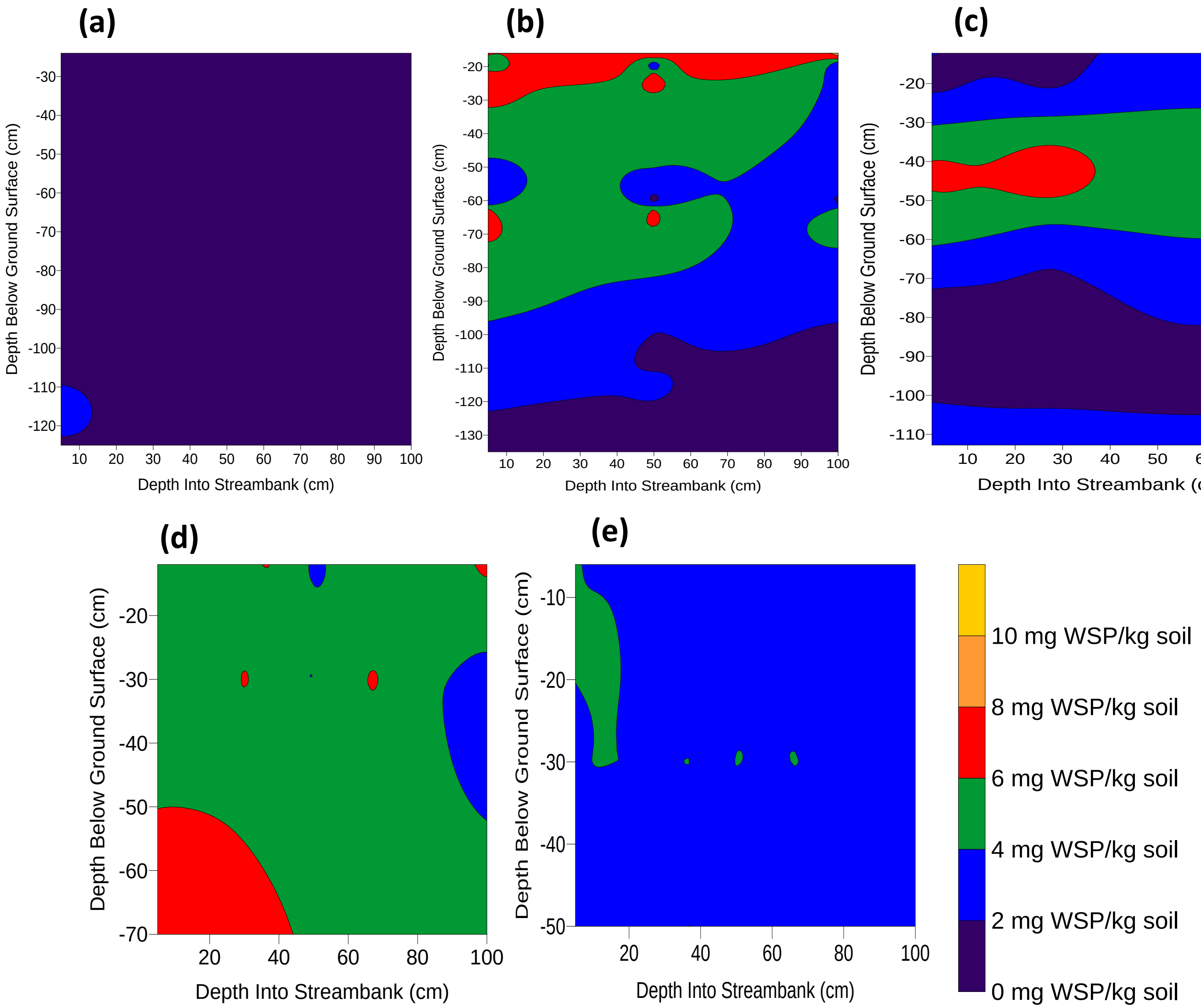


Figure 3.3. Water soluble P spatial distributions for five sampled sites: (a) Site K, (b) Site L, (c) Site M, (d) Site N, and (e) Site P.

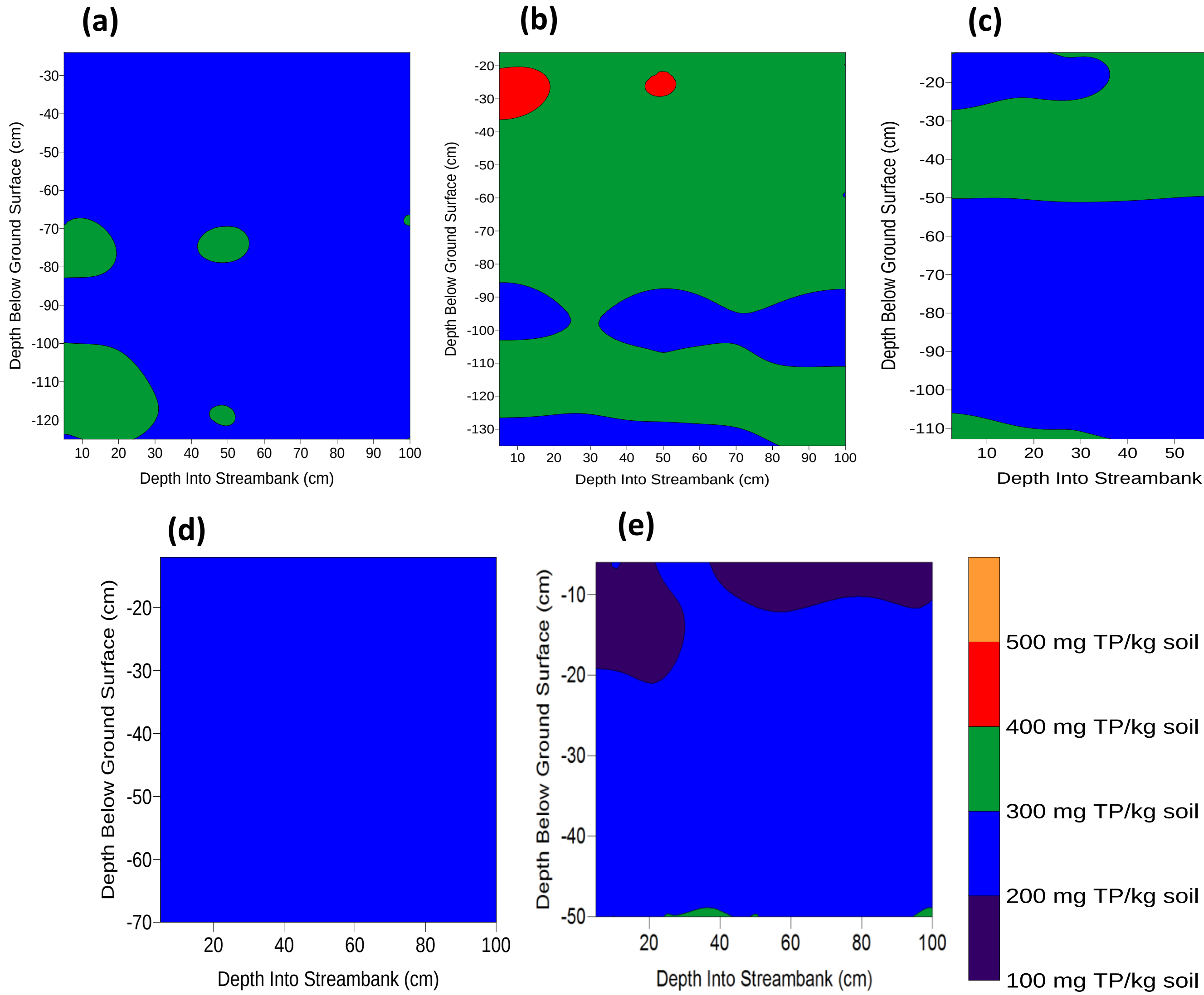


Figure 3.4. Total P spatial distributions for five sampled sites: (a) Site K, (b) Site L, (c) Site M, (d) Site N, and (e) Site P.

The contour plots suggested no clear visual relationship between WSP and TP concentrations, which was supported by comparing all of the streambank WSP and TP data together (Figure 3.5). Therefore, areas with high TP did not always have high WSP, showing that other forms of P, including other inorganic pools and organic pools, may be dominating the streambank P concentrations.

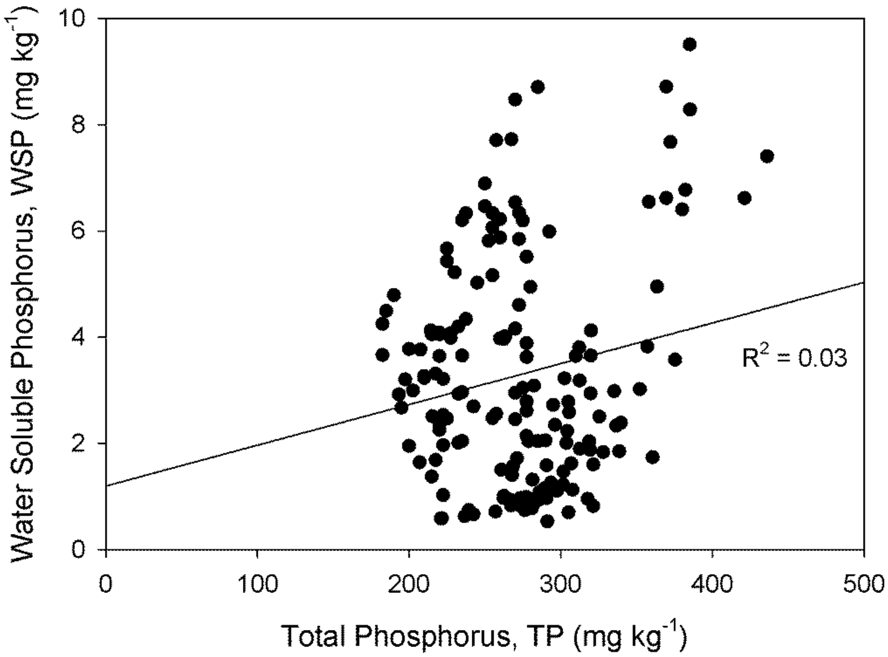


Figure 3.5. Comparison of streambank water soluble P and total P concentrations.

3.4.3. Longitudinal Distributions of Streambank Chemistry

From the spatial analysis of the streambank cross-sections, each site displayed a different visual trend in WSP and TP concentrations. Was there a trend present along the length of Spavinaw Creek? The average and standard deviation of pH, EC, WSP, and TP were assessed from each site and plotted using SigmaPlot v12.5 (SPSS, 2011) based on their distance from the confluence with Beaty Creek.

No general trend in pH was observed relative to the distance from the confluence (p=0.685) (Figure 3.6). The magnitude of the pH change over the 20 km length of the stream was approximately 7.3 to 7.0. However, median values for soil pH in this region is 5.5 (Zhang, 2001). The addition of phosphorus-rich litter raises the pH of the soil. Eroded sediments from upstream, upland sources may be transported downstream and forming these banks in Oklahoma. No longitudinal trend was observed for average EC (p=0.821) (Figure 3.6). No general trend was observed for average WSP (p=0.57) or average TP (p=0.57) across the five sites (Figure 3.6). Spatial differences may be more linked to differences in land use and management such as fertilizer type and application.

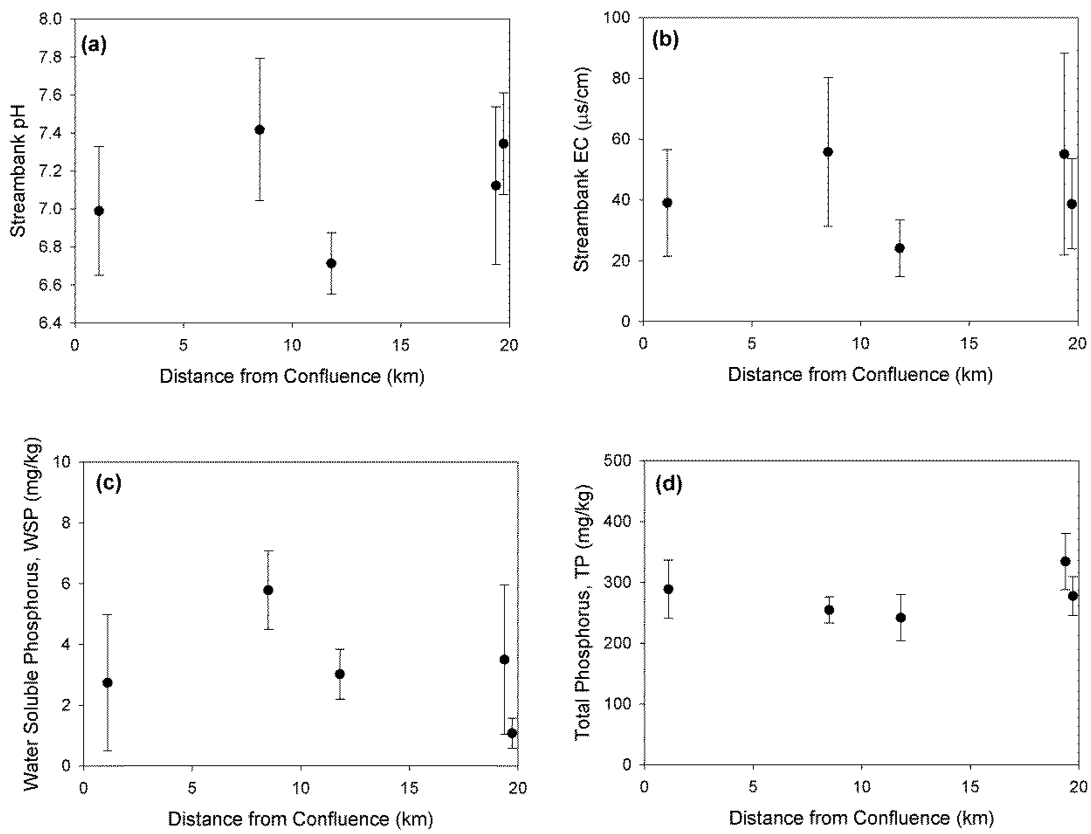


Figure 3.6. Longitudinal spatial distribution of (a) average pH, (b) average EC, (c) average water soluble P, and (d) total P at Sites M, N, P, K, and L along Spavinaw Creek.

3.4.4 Sediment and Phosphorous Loading

Measured WSP and TP concentrations were multiplied by the eroded soil mass to quantify the WSP and TP loading. From 2003-2013, Spavinaw Creek received 2.4 x 10^3 kg of WSP (2.4 x 10^2 kg WSP yr^{-1}) and 1.5 x 10^5 kg of TP (1.5 x 10^4 TP yr^{-1}) from streambanks (Table 3.2). This loading estimate was representative only of predicted streambank loading, not other potential P sources.

Table 3.2. Estimated streambank sediment and phosphorus loads along Spavinaw Creek from 2003-2013.

Study Period	Average Bank Retreat (m yr^{-1})	Eroded soil per yr per m of bank (kg m^{-1} yr^{-1})	Eroded soil mass per year (kg yr^{-1})	Total eroded soil mass (kg)	Total WSP Loading mass (kg)	Total TP Loading mass (kg)
2003-2008	2.4	3.2 x 10^3	6.6 x 10^7	3.3 x 10^8	1.1 x 10^3	6.7 x 10^4
2008-2010	0.7	1.0 x 10^3	2.1 x 10^7	4.1 x 10^7	1.4 x 10^2	8.4 x 10^3
2010-2013	4.3	5.9 x 10^3	11.9 x 10^7	3.6 x 10^8	1.2 x10^3	7.3 x 10^4
2003-2013	2.5	3.4 x 10^3	6.8 x 10^7	7.3 x 10^8	2.4 x 10^3	1.5 x 10^5

A three parameter Weibull distribution was the best fit distribution for WSP, TP, bulk density, and eroded area:

$$f(X) = \frac{\alpha}{\beta}\left(\frac{X-\gamma}{\beta}\right)^{\alpha-1} exp^{-\left(\frac{X-\gamma}{\beta}\right)^{\alpha}} \quad (5)$$

where *X* is the measured variable (*WSP*, *TP*, ρ_b, or *EA*), α is the shape, β is the scale, and γ is the threshold. Topsoil depth was best fit by a normal distribution:

$$f(X) = \frac{1}{\sqrt{2\pi\sigma^2}} exp^{-\frac{(X-\mu)^2}{2\sigma^2}} \quad (6)$$

where *X* is the measured variable (D_{ts}), μ is the location, and σ is the scale. These parameters were obtained from Minitab 16 (Minitab, 2009) from the original measured data (Table 3.3).

Table 3.3. Minitab distributions and Monte Carlo inputs for water soluble P, total P, bulk density, eroded area, and topsoil depth based on Spavinaw Creek measurements.

Variable	Distribution	Data Set	Inputs for Monte Carlo*	p value	AD value
WSP	3 parameter Weibull	Original	α = 1.25; β = 2.97; γ = 0.52	0.071	0.706
		Simulated	α = 1.21; β = 2.86; γ = 0.53	0.340	0.425
TP	3 parameter Weibull	Original	α = 2.08; β = 111.4; γ = 174.6	0.112	0.612
		Simulated	α = 2.08; β = 112.0; γ = 174.9	0.108	0.616
Bulk Density	3 parameter Weibull	Original	α = 2.51; β = 62.4; γ = 1401	0.095	0.610
		Simulated	α = 2.46; β = 61.7; γ = 1401	0.313	0.419
Eroded Area	3 parameter Weibull	Original	α = 0.48; β = 303,019; γ = 201	0.044	0.788
		Simulated	α = 0.49; β = 292,857; γ = 202	0.384	0.402
Topsoil Depth	Normal	Original	μ = 49.6; σ = 21.9	0.068	0.660
		Simulated	μ = 49.5; σ = 22.2	0.015	0.960

*α = shape; β = scale; γ = threshold; μ = location; σ = scale.

These distribution parameters were then used to create the CDF necessary to run the Monte Carlo simulation. The three parameter weibull distribution CDF and the normal distribution CDF are given by the following equations, respectively:

$$F(X) = 1 - exp^{-\left(\frac{X-\gamma}{\beta}\right)^{\alpha}} \tag{7}$$

$$F(X) = \frac{1}{2}\left[1 + erf\left(\frac{X-\mu}{\sigma\sqrt{2}}\right)\right] \tag{8}$$

The average WSP loading from the Monte Carlo analysis was 1.5 x 10^3 kg over the 10 yr period, which was equivalent to the 1.5 x 10^3 kg calculated from averages (Table 3.3). The PDF of the WSP load estimates from the Monte Carlo simulation showed that approximately 80% of loads were less than the estimate of 1.5 x 10^3 kg of WSP. The average TP loading from the Monte Carlo analysis was 1.4 x 10^5 kg from 2003-2013, which is slightly larger than the 1.2 x 10^5 kg estimated from averages (Table 3.4). The PDF of the Monte Carlo loading suggested that 78% of the TP loads were less than the average estimate, showing that this average estimate is also on the high end of the range in potential loading rates.

Table 3.4. Water soluble P and Total P averages and estimates from Monte Carlo analysis.

WSP	Calculated based on Averages	Calculated from Monte Carlo Analysis
Average WSP (mg WSP kg^{-1} soil)	3.5	3.2
Average Topsoil Depth (cm)	47	50
Bulk Density (kg m^{-3})	1,450	1,456
Eroded Area (m^2)	6.3 x 10^5	5.9 x 10^5
Average WSP Loading (kg)	1.5 x 10^3	1.5 x 10^3
TP	**Calculated based on Averages**	**Calculated from Monte Carlo Analysis**
Average TP (mg TP kg^{-1} soil)	273	274
Average Topsoil Depth (cm)	47	50
Bulk Density (kg m^{-3})	1,450	1,456
Eroded Area (m^2)	6.3 x 10^5	6.9 x 10^5
Average TP Loading (kg)	1.2 x 10^5	1.4 x 10^5

3.4.5 LOADEST Comparison

The LOADEST performance relative to observed in-stream P concentrations from all three gauges were comparable, based on the Nash-Sutcliffe Model Efficiency Value (NSE), with the LOADEST DP estimate better than TP (Table 3.5). Although the coefficient of determination (R^2), NSE, and plots (Figure 3.7) indicated a positive correlation (α=0.05) between the observed and LOADEST estimates, there was high variability.

Table 3.5. LOADEST model fit statistics for the USGS gauges at Colcord, OK, Sycamore, OK, and Cherokee City, AR.

Station	Fit Statistic	TP (mg L^{-1})	DP (mg L^{-1})
Spavinaw Creek at Colcord, OK	R^2	0.57	0.65
	NSE	0.56	0.64
Spavinaw Creek at Sycamore, OK	R^2	0.46	0.70
	NSE	0.45	0.70
Spavinaw Creek at Cherokee City, AR	R^2	0.55	0.68
	NSE	0.55	0.68

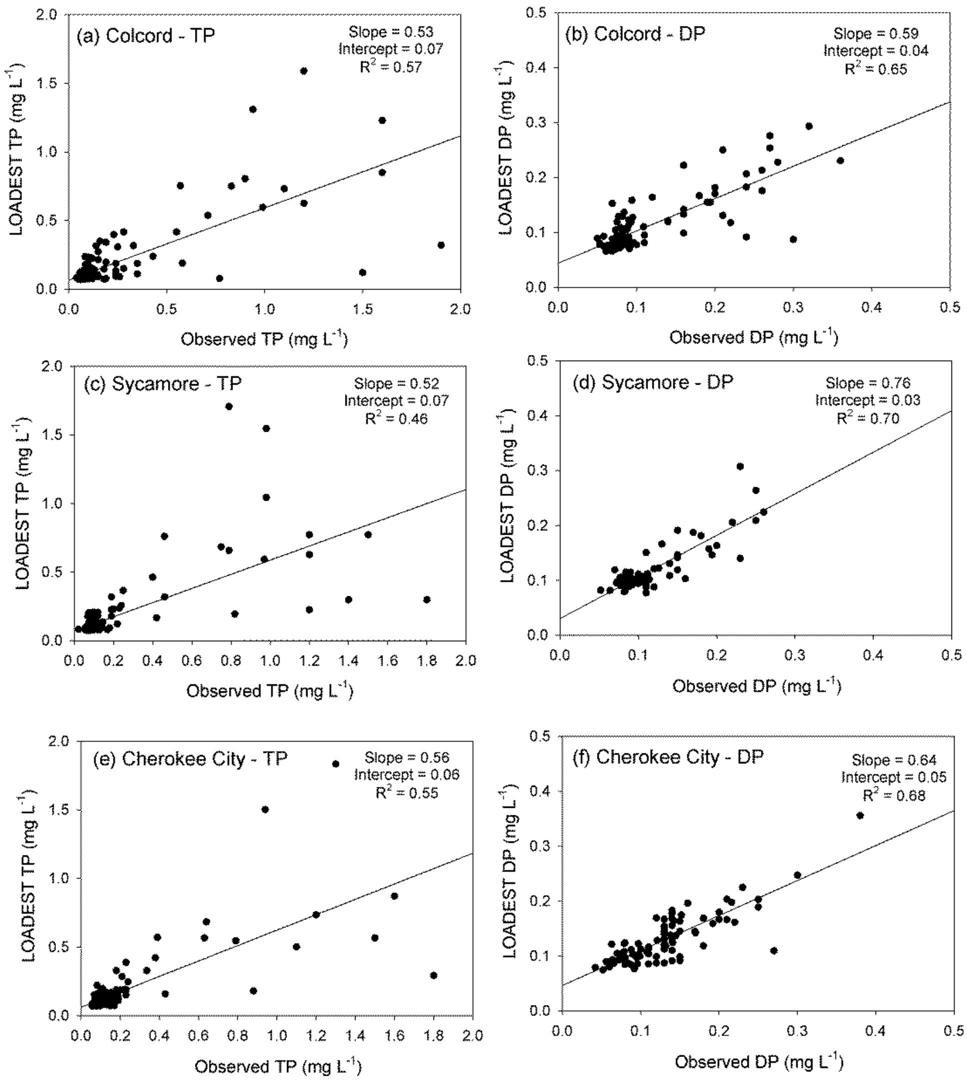


Figure 3.7. Plots of observed and LOADEST predicted concentrations for total P at (a) Colcord, (c) Sycamore, and (e) Cherokee City, and dissolved P at (b) Colcord, (d) Sycamore, and (f) Cherokee City.

The TP and DP loads from the three gauge stations (Table 3.6) were plotted relative to their distance from the confluence with Beaty (Figure 3.8). Extending the best-fit line on graphs estimated that the loads reaching the confluence were 5.9×10^4 kg TP yr^{-1} and 2.4×10^4 kg DP yr^{-1}. The difference between these estimates and those from LOADEST at the Cherokee City gauge were 3.9×10^4 kg TP yr^{-1} and 1.4×10^4 kg DP yr^{-1}, which represent the P loading into Spavinaw Creek from Oklahoma. Streambanks represented a much lower contribution of dissolved P per year than LOADEST predicted DP based on in-stream water quality data.

Streambank WSP was only 1% of the dissolved P estimated by LOADEST. TP estimates from streambank samples correlated to 31% of the LOADEST estimate. LOADEST accounted for phosphorus reaching the surface water from many sources: overgrazing, cattle/pasture, point sources, litter, crops, urban, baseflow, elevated STP, hay to forest, and other non-point sources (Storm and Mittelstet, 2015). This large difference between the WSP and DP estimates can be attributed to the many other sources of P within the watershed. The LOADEST estimates were plotted on the CDF for the Monte Carlo estimated loads (Figure 3.9). The LOADEST DP estimate corresponded to 98% of the streambank WSP load distribution, and the LOADEST TP corresponded to 57% of the TP load distribution (Figure 3.9).

Table 3.6. Dissolved P, Water soluble P, and Total P estimates from soil samples and LOADEST estimates at the three gauge stations and Oklahoma contribution.

Method	Dissolved P (kg yr^{-1})	WSP (kg yr^{-1})	Total P (kg yr^{-1})
Streambanks	-	1.4×10^2	1.2×10^4
Colcord LOADEST	1.7×10^4	-	3.9×10^4
Sycamore LOADEST	1.2×10^4	-	2.6×10^4
Cherokee City LOADEST	9.7×10^3	-	2.0×10^4
Confluence Estimate	2.4×10^4	-	5.9×10^4
Oklahoma Contribution	1.4×10^4	-	3.9×10^4

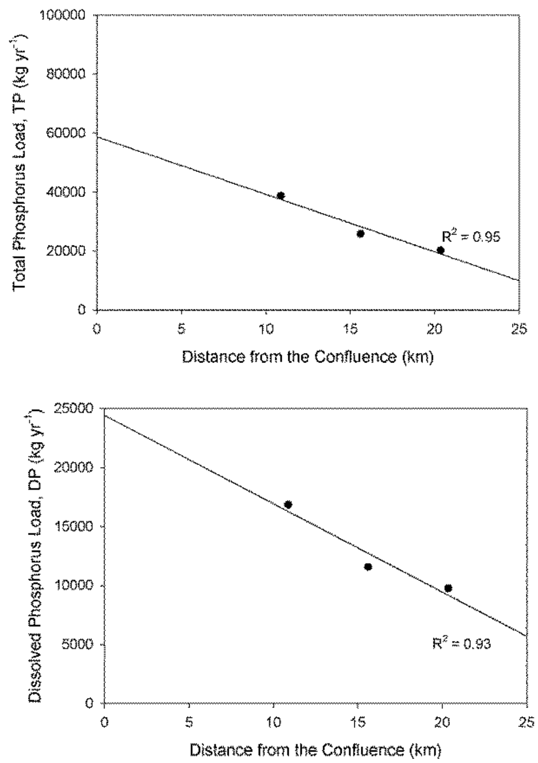


Figure 3.8. LOADEST estimates relative to the distance from the Spavinaw-Beaty confluence and extrapolation to estimate load reaching confluence.

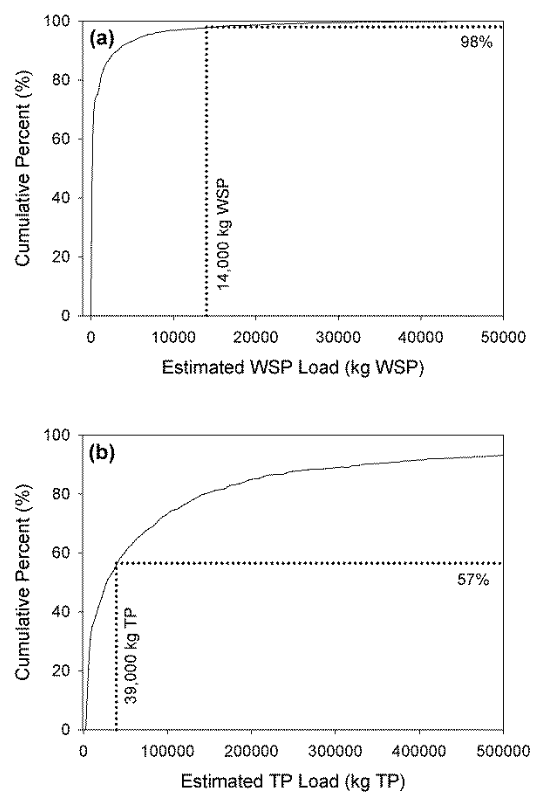


Figure 3.9. Cumulative Density Function plots of the Monte Carlo estimated for (a) Water soluble P and (b)Total P loads and the corresponding load estimate from the LOADEST model.

3.4.6 Comparison to Other Ecoregion-Specific Watersheds

Miller et al. (2014) carried out a study along Barren Fork Creek (BFC) and estimating sediment and P loading from streambanks with and without riparian vegetation from 2003-2010. Barren Fork Creek is a fourth-order stream within the Illinois River Watershed that originates in northwest Arkansas and joins with the Illinois River at Lake Tenkiller in eastern Oklahoma. The Illinois River Watershed, like the Eucha-Spavinaw watershed, has many areas listed on the EPA's 303(d) list for nutrient related impairments. BFC flows through the Ozark ecoregion and is characterized by cherty topsoil with underlying gravel layers (Fuchs et al., 2009; Fox et al., 2011).

The study looked at three sites considered to be historically unprotected by riparian vegetation (HUP) and seven sites that were historically protected (HP). Similar techniques as those discussed previously in this paper were implemented in order to determine streambank erosion and WSP and TP concentrations were used. Average topsoil depth was 1 m. It was found that HUP sites had on average 49 m of bank retreat, while HP sites had 18 m during the 7-year study period (2003-2010).

Phosphorus testing and analysis showed that of their 253 samples, 14% had a WSP concentration greater than the threshold (8.2 mg WSP kg^{-1} soil), 25% had a DPS value greater than the threshold (25% DPS), and 13% were above both thresholds. In total, the ten study sites contributed 2.2×10^2 kg WSP (3.1 kg WSP yr^{-1}) and 1.7×10^3 kg TP (2.5×10^2 kg TP yr^{-1}) from 2003-2010. A helicopter survey was carried out and found that of the 55 km reach of BFC in Oklahoma, 11 to 55% (36% average) of 2 km reaches were considered unstable and eroding. Annual WSP and TP load estimates from the ten sites were scaled up to represent all of the reaches considered to be unstable. This led to an estimate of 1.2×10^3 kg WSP yr^{-1} and 9.3×10^4

kg TP yr^{-1} for all of BFC during the seven years LOADEST predicted 1.38 x 10^{4} kg DP yr^{-1} of and 4.6 x 10^{4} kg TP yr^{-1}.

Both Barren Fork Creek and Spavinaw Creek flow through the Ozark ecoregion, have cherty topsoil with underlying gravel layers, and surrounding land has historic litter application. However, there are very clear differences between the results along the two creeks (Table 3.7). The average WSP estimated from streambanks in Spavinaw was only 1% of the DP estimated from LOADEST; in Barren Fork it was 8.7%. LOADEST estimates of DP for both creeks are much higher than those predicted from streambanks, which is due to LOADEST taking into account point and non-point sources as discussed previously. Average TP estimates from streambanks and LOADEST are on the same order of magnitude for both creeks. BFC receives almost an order of magnitude higher WSP and TP from streambanks compared to Spavinaw Creek.

Table 3.7. Comparison of Dissolved P, Water soluble P, and Total P estimates at Spavianw Creek and Barren Fork Creek.

Location	Method	Dissolved P (kg yr^{-1} ha^{-1})	WSP (kg yr^{-1} ha^{-1})	Total P (kg yr^{-1} ha^{-1})
Spavinaw	Streambanks	-	1.5 x 10^{-3}	1.3 x 10^{-1}
	LOADEST	1.5 x 10^{-2}	-	4.1 x 10^{-1}
Barren Fork	Streambanks	-	1.5 x 10^{-2}	1.2 x 10^{0}
	LOADEST	1.7 x 10^{-1}	-	5.7 x 10^{-1}

Although Barren Fork Creek and Spavinaw Creek flow through the same ecoregion and have similar land use and streambank characteristics, they are an order of magnitude different in terms of their WSP loading. Therefore, estimates from one creek, whether it be BFC or Spavinaw Creek, cannot accurately be used to predict P loading along another system. Each creek has unique differences in surrounding land management, discharge, topsoil depth, and other variables that affect sediment and P loading. When estimating loading rates, each creek should be assessed individually. These individual studies can then be linked together to gain a better understanding of the system as a whole.

3.4.7. Influence of Vegetation on Phosphorus Loading

Longitudinal analysis of WSP and TP found that concentrations remained relatively constant along Spavinaw Creek, showing that there is no significant difference in the concentrations at locations with or without riparian vegetation. However, the presence of established vegetation along streambanks was estimated to reduce bank retreat by three times. Therefore, the potential reduction in P loading with riparian protection was more related to the presence of vegetation limiting erosion than affecting the concentration of P. Based on the aerial imagery, approximately one-third of the streambanks are unprotected. In order to determine the P load reduction due to the addition of riparian vegetation, the loading under the current conditions must first be calculated. The current conditions assumed an average WSP concentration of 3.5 mg WSP kg^{-1} soil, two-thirds of the streambanks were vegetated and bank retreat of 2.5 m yr^{-1}, while the remaining one-third was non-vegetated and had three times more bank retreat (7.5 m yr^{-1}). This calculation estimated the original P load to be 14.6 mg WSP kg^{-1} soil m yr^{-1}. The new P loading assumed a situation in which all of the streambanks were vegetated, had a retreat of 2.5 m yr^{-1}, and a WSP concentration of 3.5 mg WSP kg^{-1} soil. The new loading was estimated as 8.8 mg WSP kg^{-1} soil m yr^{-1}. Based on these estimates there would be a two-fifths reduction in P loading if established riparian vegetation was implemented on the non-vegetated reaches.

3.5 Conclusions

In Spavinaw Creek watershed it was estimated that from 2003 to 2013 the WSP load was 2.4×10^3 kg and the TP load was 1.5×10^5 kg. Streambank P load estimates were comparable to LOADEST in-stream estimates for TP, but LOADEST predicted an order of magnitude higher DP loads than the WSP load estimated from banks. Sources other than streambank P were contributing to the DP observed in the stream. The implementation of vegetation along unprotected banks could potentially result in a two-fifths reduction in P loads. Spavinaw Creek streambank P load estimates were compared to those reported for Barren Fork Creek, which was in the same ecoregion and had similar streambank characteristics. Spavinaw Creek had an order of magnitude lower WSP loading per year. Therefore, although both creeks were similar, there were other differences such as land use, litter application, and hydrology that made it difficult to use one study to predict the behavior of another creek. In order to gain an understanding of nutrient loading into surface water, individual streams systems should be studied to account for this variability.

3.6 Acknowledgements

This material is based upon work supported by the Environmental Protection Agency (EPA) under the grant FY 2012 319(h) Special Project #C9-00F56701 and a 2012-2013 Oklahoma Water Resources Research Institute (OWRRI) grant through the USGS 104(b) grant program. The authors acknowledge Kate Klavon and Whitney Lisenbee for assisting with soil sampling, and Abigail Parnell for assisting with soil processing.

REFERENCES

Addiscott, T. M., and D. Thomas. 2000. Tillage, mineralization and leaching: Phosphate. *Soil and Tillage Research Journal* 53: 255-273. doi: 10.1016/S0167-1987(99)00110-5.

Aldridge, K. T., J. D. Brookes, and G. G. Ganf. 2010. Changes in abiotic and biotic phosphorus uptake across a gradient of stream condition. *River Research and Applications* 26: 636-649. doi: 10.1002/rra.1276.

Amundson, R., A. A. Berhe, J. W. Hopmans, C. Olson, A. E. Sztein, and D. L. Sparks. 2015. Soil and human security in the 21st century. *Science* 348(6235): 1261071(1)-1261071(5). doi: 10.1126/science.1261071.

Ashbridge, D., 1995. Processes of river bank erosion and their contribution to the suspended sediment load of the River Culm, Devon. In *Sediment and water quality in river catchments*, 229-245. Foster et al.. Chichester, United Kingdom: John Wiley & Sons.

ASTM International. 2002. Standard Test Method for Particle-Size Analysis of Soils, Method 422-63. *http://www.astm.org*. Accessed 9/1/2014.

Ballantine, D. J., D. E. Walling, A. L. Collins, and G. J. L. Leeks. 2009. The content and storage of phosphorus in fine-grained channel bed sediment in contrasting lowland agricultural catchments in the UK. *Geoderma* 151: 141-149. doi: 10.1016/j.geoderma.2009.02.021.

Beeson, C. E., and P. F. Doyle. 1995. Comparison of bank erosion at vegetated and non-vegetated channel bends. *Water Resources Bulletin* 31(6): 983-990.

Bernhardt, E. S., M. A. Palmer, J. D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G. M. Kondolf, P. S. Lake, R. Lave, J. L. Meyer, T. K. O'Donnell, L. Pagano, B. Powell, and E. Sudduth. 2005. Synthesizing U.S. river restoration efforts. *Science* 308: 636-637, doi:10.1126/science.1109769.

Boström, B., M. Jansson and C. Forsberg. 1982. Phosphorus release from lake sediments. *Archiv für Hydrobiologie–Beiheft Ergebnisse der Limnologie* 18: 5-59.

Boström, B., 1984. Potential mobility of phosphorus in different types of lake sediment. *Internationale Revue der gesamten Hydrobiologie und Hydrographie* 69: 457-474.

Boström, B., G. Persson, and B. Broberg. 1988. Bioavailability of different phosphorus forms in freshwater systems. *Hydrobiologia* 170: 133-155.

Bowie, A. J., 1982. Investigations of vegetation for stabilizing eroding streambanks. *Transactions of the ASAE* 25(6): 1601-1606, 1611.

Boynton, W. R., J. H. Garber, R. Summers, and W. M. Kemp. 1995. Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries* 18 (1B): 285-314.

Breeuwsma, A. J., G. A. Reijerink, and O. F. Schoumans. 1995. Impact of manure on accumulation and leaching of phosphate in areas of intensive livestock farming. In *Animal Waste the Land-Water Interface*, K. Steele, ed.. CRC Press Inc..

Brice, J. C., 1982. Stream channel stability assessment. Federal Highway Administration, Office of Research and Development, FHWA/RD-82/021.Washington, D.C.

Broberg, O., and G. Persson. 1988. Particulate and dissolved phosphorus forms in freshwater: composition and analysis. *Hydrobiologia* 170: 61-90.

Bull, L. J., 1997. Magnitude and variation in the contribution of bank erosion to the suspended load of the River Severn, UK. *Earth Surface Processes and Landforms* 22: 1109-1123.

Burkhardt, J. C., and B. L. Todd. 1998. Riparian forest effect on lateral stream channel migration in the glacial till plains. *Journal of the American Water Resources Association* 34(1): 179-184.

Busman, L., J. Lamb, G. Randall, G. Rehm, and M. Schmitt. 2009. Nutrient Management: The nature of phosphorus in soils. University of Minnesota Extension. WW-06795-GO. http://www.extension.umn.edu.

Cembella, A. D., J. A. Naval, and P. J. Harrison. 1984. The utilization of inorganic and organic phosphorus compounds as nutrients by eukaryotic microalgae: A multidisciplinary perspective: Part 1. *Critical Reviews in Microbiology* 10: 317-391.

Chu, S. P., 1946. The utilization of organic phosphorus by phytoplankton. *Journal of the Marine Biological Association of the United Kingdom* 26: 285-295.

Clark, L. A., and T. M. Wynn. 2007. Methods for determining streambank critical shear stress and soil erodibility: Implications for erosion rate predictions. *Transactions of the ASABE* 50(1): 95-106.

Cooper, D. J., K. M. Mutz, and B. Van Haveren. 1990. Draft intermountain riparian lands evaluation methodology. Prepared for USEPA, Office of Policy Analysis and Office of Wetlands Protection, Washington, D.C.

Cordell, D., J. -O. Drangert, and W. White. 2009. The story of phosphorus: Global food security and food for thought. *Global Environmental Change* 19: 292-305.

Couper, P. R., 2003. Effects of silt-clay content on the susceptibility of river banks to subaerial erosion. *Geomorphology* 56(1-2): 95-108.

Couper, P. R., and I. P. Maddock. 2001. Subaerial river bank erosion processes and their interaction with other bank erosion mechanisms on the River Arrow, Warwickshire, UK. *Earth Surface Processes and Landforms* 26(6): 631-646.

Cox, F. R., E. J. Kamprath, and R. E. McCollum. 1981. A descriptive model of soil test nutrient levels following fertilization. *Soil Science Society of America Journal* 45: 529-532. doi: 10.2136/sssaj1981.03615995004500030018x.

Daly, E. R., 2014. Quantifying fluvial resistance of streambanks using JET erosion tests. PhD diss. Stillwater, Okla.: Oklahoma State University, Department of Biosystems and Agricultural Engineering.

Daly, E. R., R. B. Miller, and G. A. Fox. 2015. Modeling streambank erosion and failure along protected and unprotected composite streambanks. *Advances in Water Resources*. http://dx.doi.org/10.1016/j.advwatres.2015.01.004.

Domagalski, J. L., and H. M. Johnson. 2011. Subsurface transport or orthophosphate in five agricultural watersheds, USA. *Journal of Hydrology* 409: 157-171. doi: 10.1016/j.jhydrol.2011.08.014.

Drake, W. M., J. T. Scott, M. Evans-White, B. Haggard, A. Sharpley, and C. W. Rogers. 2012. The effect of periphyton stoichiometry and light on biological phosphorus immobilization and release in streams. *Limnology* 13: 97-106. doi: 10.1007/s10201-011-0359-z.

Driels, M. R., and Y. S. Shin. 2004. Determining the number of iterations for Monte Carlo simulations of weapon effectiveness. No. NPS-MAE-04-005. Naval Postgraduate School Department of Mechanical and Astronautical Engineering. Monterey, Cali.

Dunn, W. L., and J. K. Shultis. 2012. Introduction. In *Exploring Monte Carlo Methods*, 1-20. Elsevier.

Engstrom, D. R., and J. E. Almendinger. 1997. Historical changes in sediment and phosphorus loading to the Upper Mississippi River: Mass-balance reconstructions from the sediments of Lake Pepin. Final Research Report prepared for the Metropolitan Council Environmental Servicesm St. Croix Watershed Research Station. Mairine-on-St. Croix, MN.: Science Museum of Minnesota.

ESRI. 2014. *ArcGIS Desktop*. Ver. 10.2. Redlands, CA: Environmental Systems Research Institute.

Everett, J., 2004. Eucha-Spavinaw Watershed Management Team. Presented at the Secretary of Agriculture’s Poultry Waste Task Force meeting April 20, 2004 in Jay, Oklahoma.

Fitzgerald, G. P., 1970. Evaluation of the availability of sources of nitrogen and phosphorus for algae. *Journal of Phycology* 6: 239-247.

Fleischer, S., 1983. Microbial P release during enhanced glycolysis. *Naturwissenschaften* 70: 415.

Fox, G. A., and G. V. Wilson. 2010. The role of subsurface flow in hillslope and streambank erosion: A review. *Soil Science Society of America Journal* 74(3): 717-733.

Fox, G. A., D. M. Heeren, R. B. Miller, A. R. Mittelstet, and D. E. Storm. 2011. Flow and transport experiments for a streambank seep originating from a preferential flow pathway. *Journal of Hydrology* 403(3-4): 360-366.

Fuchs, J. W., G. A. Fox, D. E. Storm, C. J. Penn, G. O. Brown. 2009. Subsurface transport of phosphorus in riparian floodplains: influence of preferential flow paths. *Journal of Environmental Quality* 38 (2): 473-484.

Fuhrman, J. K., H. Zhang, J. L. Schroeder, and R. L. Davis. 2005. Water soluble phosphorus as affected by soil to extractants ratios, extraction time and electrolyte. *Communications in Soil Science and Plant Analysis* 36: 925-935.

Furumai, H., and S. Oghaki. 1982. Fractional composition of phosphorus in sediments related to release. *Water Science and Technology* 14: 215-226.

Galloway, R. A., and R. W. Krauss. 1963. Utilization of various phosphorus compounds by chlorella. In *Studies on microalgae and photosynthetic bacterial*, 569-575. *Plant Cell Physiology Special Issue*.

Gburek, W. J., and A. N. Sharpley. 1998. Hydrologic controls on phosphorus loss from upland agricultural watersheds. *Journal of Environmental Quality* 27: 267-277. doi: 10.2134/jeq1998.00472425002700020005x.

Gburek, W. J., A. N. Sharpley, and D. B. Beegle. 2007. Incorporation of variable-source-area hydrology in the Phosphorus Index: A Paradigm for improving relevancy of watershed research. In *Proceedings of the 2nd Interagency Conference on Research in the Watersheds*, 151-160. D.L. Fowler, ed. Otto, NC. 16-18 May 2006. Asheville, NC.: US Forest Service.

Golden Software. 2002. *Surfer*. Ver. 8. Golden, Colo.: Golden Software, Inc.

Golterman, H. L., 1977. Sediments as a source for algal growth. In *Interactions between sediments and freshwater*, 286-293. H.L. Golterman, ed. The Hague: Dr. W. Junk.

Grabowski, R. C., I. G. Droppo, and G. Wharton. 2011. Erodibility of cohesive sediment: The importance of sediment properties. *Earth-Science Reviews* 105(1): 101-120. doi: 10.1016/j.earscirev.2011.01.008.

Guo, F., and R. S. Yost. 1999. Quantifying the available soil phosphorus pool with the acid ammonium oxalate method. *Soil Science Society of America Journal* 63: 651-656.

Haan, C. T., B. J. Barfield, and J. C. Hayes. 1991. Hydrologic Frequency Analysis. In *Design Hydrology and Sedimentary for Small Catchments*, 5-36. San Diego, CA.: Academic Press, Inc.

Haggard, B. E., D. E. Storm, and E. H. Stanley. 2001. Effect of a point source input on stream nutrient retention. *Journal of the American Water Resources Association* 37: 1291-1299. doi: 10.1111/j.1752-1688.2001.tb03639.x.

Haggard, B. E., E. H. Stanley, and D. E. Storm. 2005. Nutrient retention in a point-source-enriched stream. *Journal of the North American Benthological Society* 24: 29-47.

Hanson, G. J., 1990. Surface erodibility of earthen channels at high stresses: II. Developing in situ testing device. *Transactions of the ASAE* 33(1): 132-137.

Hanson, G. J., and K. R. Cook. 1997. Development of excess shear stress parameters for circular jet testing. ASAE Paper No. 972227. St. Joseph, Mich.: ASAE.

Hanson, G. J., and A. Simon. 2001. Erodibility of cohesive streambeds in the loess area of the Midwestern USA. *Hydrological Processes* 15(1): 23-28.

Harmel, R. D., C. T. Haan, and R. C. Duntnell. 1999. Bank erosion and riparian vegetation influences; upper Illinois River, Oklahoma. *Transactions of ASAE* 42 (5): 1321-1329.

Heeren, D. M., R. B. Miller, G. A. Fox, D. E. Storm, T. Halihan, and C. J. Penn. 2010. Preferential flow effects on subsurface contaminant transport in alluvial floodplains. *Transactions of the ASABE* 53(1): 127-136.

Heeren, D. M., G. A. Fox, R. B. Miller, D. E. Storm, A. R. Mittelstet, A. K. Fox, C. J. Penn, and T. Halihan. 2011. Stage-dependent transient storage of phosphorus in alluvial floodplains. *Hydrological Processes* 25(20): 3230-3243, doi: 10.1002/hyp.8054.

Heeren, D. M., A. R. Mittelstet, G. A. Fox, D. E. Storm, A. T. Al-Madhhachi, T. L. Midgley, A. F. Stringer, K. B. Stunkel, and R. B. Tejral. 2012. Using rapid geomorphic assessments to assess streambank stability in Oklahoma Ozark streams. *Transactions of ASABE* 55(3): 957-968.

Hickin, E. J., 1984. Vegetation and river channel dynamics. *Canadian Geographer* 28(2): 111-126.

Holman, I. P., M. J. Whelan, N. J. K. Howden, P. H. Bellamy, N. J. Willby, M. Rivas-Casado, and P. McConvey. 2008. Phosphorus in groundwater: An overlooked contributor to eutrophication? *Hydrological Processes* 22: 5121-5127. doi: 10.1002/hyp.7198.

Hooke, J. M., 1979. An analysis of the processes of river bank erosion. *Journal of Hydrology* 42: 39-62.

Hosomi, M., M. Okada, and R. Sudo. 1981. Release of phosphorus from sediments. *Verhandlungen des Internationalen Verein Limnologie* 21: 628-633.

Jarvie, H. P., M. D. Jurgens, R. J. Williams, C. Neal, J. J. L. Davies, C. Barrett, and J. White. 2005. Role of river bed sediments as sources and sinks of phosphorus across two major eutrophic UK river basins: The Hampshire Avon and Herefordshire Wye. *Journal of Hydrology* 304: 51-74. doi: 10.1016/j.jhydrol.2004.10.002.

Jones, D. L, P. Cross, P. J. A. Withers, T. H. DeLuca, D. A. Robinson, R. S. Quilliam, I. M. Harris, D. R. Chadwick, and G. Edwards-Jones. 2013. Review: Nutrient stripping: the global disparity between food security and soil nutrient stocks. *Journal of Applied Ecology* 50(4): 851-862. doi: 10.111/1365-2664.12089.

Kleinman, P. J. A., A. N. Sharpley, A. R. Buda, R. W. McDowell, and A. L. Allen. 2011. Soil controls of phosphorus in runoff: Management barriers and opportunities. *Canadian Journal of Soil Science* 91: 329-338. doi: 10.4141/cjss09106.

Kotak, B. G., S. L. Kenefick, D. L. Fritz, C. G. Rousseaux, E. E. Prepas, and S. E. Hrudey. 1993. Occurrence and toxicological evaluation of cyanobacterial toxins in Alberta lakes and farm dugouts. *Water Research* 27: 495-506.

Kotak, B. G., E. E. Prepas, and S. E. Hrudey. 1994. Blue green algal toxins in drinking water supplies: research in Alberta. *Lake Line* 14: 37-40.

Kroening, S. E., and W. J. Andrews. 1997. Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin: Nitrogen and phosphorus in streams, streambed sediment, and groundwater, 1971-94. U.S. Geological Survey, Water-Resources Investigations Report 97-4107. Mounds View, MN.

Kronvang, B., R. Grant, and A. L. Laubel. 1997. Sediment and phosphorus export from a lowland catchment: Quantification of sources. *Water, Air and Soil Pollution* 99: 465-476.

Kronvang, B., J. Audet, A. Baattrup-Pedersen, H. S. Jensen, and S. E. Larsen. 2012. Phosphorus loads to surface water from bank erosion in a Danish lowland river basin. *Journal of Environmental Quality* 41: 304-313.

Laubel, A., B. Kronvang, A. B. Hald, and C. Jensen. 2003. Hydromorphological and biological factors influencing sediment and phosphorus loss via bank erosion in small lowland rural streams in Denmark. *Hydrological Processes* 17 (17): 3443-3463.

Lavendel, B., 2002. The business of ecological restoration. *Ecological Research* 20:173-178.

Lawler, D. M., 1993. The measurement of river bank erosion and lateral channel change: A review. *Earth Surface Processes and Landforms* 18: 777-821.

Lawton, L. A., and G. A. Codd. 1991. Cyanobacterial (blue-green algae) toxins and their significance in UK and European waters. *Journal of the Institute of Water Environmental Management* 5: 460-465.

Lee, G. F., R. A. Jones, and W. Rast. 1980. Availability of phosphorus to phytoplankton and its implications for phosphorus management strategies. In *Phosphorus management* *strategies for lakes*, 259-308. R.C. Loehr, C.S. Martin, and W. Rast, eds. Ann Arbor, Mich.: Ann Arbor Science.

Leinweber, P., R. Meissner, K. U. Eckhardt, and J. Seeger. 1999. Management effects on forms of phosphorus in soil and leaching losses. *European Journal of Soil Science* 50: 413-424.

Leopold, L. B., and W. B. Langbein. 1960. River Meanders. *Bulletin of the Geological Society of America* 71(6): 769-793.

Lijklema, L., 1977. Role of iron in the exchange of phosphate between water and sediments. In *Interactions between Sediments and Fresh Water, Proceedings of an International Symposium*. The Hague: Dr. W. Junk.

Logan, T. J., 1982. Mechanisms for release of sediment-bound phosphate to water and the effects of agricultural land management on fluvial transport of particulate and dissolved phosphate. *Hydrobiologia* 92: 519-530.

Lookman, R., D. Freese, R. Merckx, K Vlassak, and W. H. van Riemsdijk. 1995. Long-term kinetics of phosphate release from soil. *Environmental Science and Technology* 29: 1569-1575.

Maguire, R. O., and J. T. Sims. 2002. Measuring agronomic and environmental soil properties and phosphate saturation and predicting phosphorus leaching with Mehlich 3. *Soil Science Society of America Journal* 66: 2033-2039.

Marengo, G., and G. Premazzi. 1985. Biological availability of P-loads to Lake Lugano. *Verhandlungen des Internationalen Verein Limnologie* 22: 3351-3355.

Martin, A., and G. D. Cooke. 1994. Health risks in eutrophic water supplies. *Lake Line* 14: 24-26.

McDowell, R. W., and A. N. Sharpley. 2002. The availability of residual phosphorus in high phosphorus soils. *Communications in Soil Science and Plant Analysis* 33: 1235-1246. doi: 10.1081/CSS-120003884.

Micheli, E. R., and J. W. Kirchner. 2002. Effects of wet meadow riparian vegetation on streambank erosion. 1. Remote sensing measurements of streambank migration and erodibility. *Earth Surface Processes and Landforms* 27: 627-639. doi: 10.1002/esp.338.

Midgley, T. L., G. A. Fox, and D. M. Heeren. 2012. Evaluation of the Bank Stability and Toe Erosion Model (BSTEM) for predicting lateral streambank retreat on composite streambanks. *Geomorphology* 145-146: 107-114.

Miller, R. B., G. A. Fox, C. Penn, S. Wilson, A. Parnell, R. A. Purvis, and K. Criswell. 2014. Estimating sediment and phosphorus loads from streambanks with and without riparian protection. *Agriculture, Ecosystems, and Environment* 189: 70-81, doi: 10.1016/j.agee.2014.03.016.

Minitab. 2009. *Minitab Statistical Software*. Ver. 16. State College, Pennsylvania. Minitab, Inc.

Mittelstet, A., D. M. Heeren, G. A. Fox, D. E. Storm, M. J. White, and R. B. Miller. 2011. Comparison of subsurface and surface runoff phosphorus transport rates in alluvial floodplains. *Agriculture, Ecosystems, and Environment* 141: 417-425.

Morisawa, K. 1985. *Rivers: Forms and Processes*. New York, N.Y.: Longman.

Narshimhan, B., P. M. Allen, R. Srinivasan, S. T. Bednarz, J. Arnold, and J. A Dunbar. 2007. Streambank erosion and best management practice simulation using SWAT. In *Watershed Management to Meet Water Quality Standards and TMDLS (Total Maximum Daily Load)*, eds. A. McFarland, A. Saleh, St. Joseph, Michigan: ASABE.

Nash, D., and M. Hannah. 2011. Using Monte-Carlo simulations and Bayesian networks to quantify and demonstrate the impact of fertiliser best management practices. *Environmental Modelling and Software* 26: 1079-1088. doi: 10.1016/j.envsoft.2011.03.009.

Nürnburg, G. R., and H. Peters. 1984. Biological availability of soluble reactive phosphorus in anoxic and oxic freshwaters. *Canadian Journal of Fisheries and Aquatic Sciences* 41: 757-765.

Odgaard, A. J., 1987. Streambank erosion along two rivers in Iowa. *Water Resources Research* 23: 1225-1236.

Oklahoma Conservation Commission (OCC). 2007. Watershed Based Plan for the Lake Eucha-Lake Spavinaw Watershed. Oklahoma City, Ok.: Oklahoma Conservation Commission Water Quality Division. Available at: www.ok.gov. Accessed 23 September 2014.

Overbeck, J., 1962. Untersuchungen zum phosphathaushalt von Grünalgen. II Die Verwertung von Pyrophosphat und organisch gebundenen Phosphaten un ihre Beziehung zu den Phosphatasen von Scenedesmus quadricauda (Trup.) Bréb. *Archiv für Hydrobiologie* 58: 281-308.

Owens, P. N., and D. E. Walling. 2002. The phosphorus content of fluvial sediment in rural and industrialized river basins. *Water Research* 36: 685-701 doi: 10.1016/S0043-1354(01)00247-0.

Palmstrom, N. S., R. E. Carlson, and G. D. Cooke. 1988. Potential links between eutrophication and formation of carcinogens in drinking water. *Lake and Reservoir Management* 4: 1-15.

Pautler, M. C., and J. T. Sims. 2000. Relationships between soil test phosphorus, soluble phosphorus, and phosphorus saturation in Delaware soils. *Soil Science Society of America Journal* 64: 765-773.

Pierzynski, G. M., J. T. Sims, and G. F. Vance. 2000. Soil phosphorus and environmental quality. In *Soils and Environmental Quality*, 155-207. Boca Raton. CRC Press.

Pote, D. H., T. C. Daniel, A. N. Sharpley, P. A. Moore Jr., D. R. Edwards, and D. J. Nichols. 1996. Relating extractable soil phosphorus to phosphorus losses in runoff. *Soil Science Society of America Journal* 60:855-859.

Premazzi, G., and G. Zanon. 1984. Availability of sediment P in Lake Lugano. *Verhandlungen des Internationalen Verein Limnologie* 22: 1113-1118.

Rawlins, B. G., 2011. Controls on the phosphorus content of fine stream bed sediments in agricultural headwater catchments at the landscape-scale. *Agriculture, Ecosystems, and Environment* 144: 352-363. doi: 10.1016/j.agee.2011.10.002.

Rondeau, B., D. Cossa, P. Gagnon, and L. Bilodeau. 2000. Budget and source of suspended sediment transported in the St. Lawrence River, Canada. *Hydrological Processes* 14: 21-36.

Roseboom, D. P,. 1987. Case Studies of Stream and River Restoration. In: *Management of the Illinois River System: The 1990's and Beyond*, 184-194.

Rosgen, D., 2001. A practical method of computing streambank erosion rate. p. 9–15. In Proc. 7th Federal Interagency Sedimentation Conf., Reno, NV. 25–29 Mar. 2001. Vol. 2. USGS, Reston, VA.

Runkel, R. L., C. G. Crawford, and T. A. Cohn. 2004. Load Estimator (LOADEST): A FORTRAN program for estimating constituent loads in streams and rivers. U.S. Department of the Interior, U.S. Geological Survey.

Sabbagh, G. J., and G. A. Fox. 1999. Statistical method for evaluation of a water table management model. *Transactions of the ASAE* 42(3): 713-719.

Sattari, S. Z., A. F. Bouwman, K. E. Giller, and M. K. van Ittersum. 2012. Residual soil phosphorus as the missing piece in the global phosphorus crisis puzzle. *Proceedings of the National Academy of Sciences* 109:6348-63563. doi: 10.1073/pnas.1113675109.

Schade, J. D., K. MacNeill, S. A. Thomas, F. C. McNeely, J. R. Welter, and J. Hood. 2011. The stoichiometry of nitrogen and phosphorus spiraling in heterotrophic and autotrophic streams. *Freshwater Biology* 56: 424-436. doi: 10.1111/j.1365-2427.2010.02509.x.

Schoumans, O. F., 2000. Determination of the degree of phosphorus saturation in non-calcareous soils. In: *Methods of phosphorus analysis for soils, sediments, residuals, and waters*; 31-34. Ed. G.M. Pierzynski. Kansas State University. Manhattan, Kansas.

Sekely, A. C., D. J. Mulla, and D. W. Bauer. 2002. Streambank slumping and its contribution to the phosphorus and suspended sediment loads of the Blue Earth River, Minnesota. *Journal of Soil Water Conservation* 57 (5): 243-250.

Shapiro, J., 1967. Induced rapid release and uptake of phosphate by microorganisms. *Science* 155: 1269-1271.

Sharpley, A. N., 1995. Dependence of runoff phosphorus on extractable soil phosphorus. *Journal of Environmental Quality* 24: 920-926.

Shapley, A. N., P. J. A. Kleinman, D. N. Flaten, and A. R. Buda. 2011. Critical source area management of agricultural phosphorus: Experiences, challenges and opportunities. *Water Science and Technology* 64: 945-952. doi: 10.2166/wst.2011.712.

Sharpley, A. N., P. J. A. Kleinman, P. Jordan, B. Bergström, and A. L. Allen. 2009. Evaluating the success of phosphorus management from field to watershed. *Journal of Environmental Quality* 38: 1981-1988. doi: 10.2134/jeq2008.0056.

Sharpley, A. N., and S. Rekolainen. 1997. Phosphorus in Agriculture and Its Environmental Implications. In *Phosphorus Loss from Soil to Water*, 1-54. eds. H. Tunney, O.T. Carton, P.C. Brookes, and A.E. Johnston. Oxon, UK: CAB International.

Simon, A., R. L. Bingner, E. J. Langendoen, and C. V. Alonso. 2002. Actual and reference sediment yields for the James Creek Watershed, Mississippi. USDA Agricultural Research Service National Sedimentation Laboratory Research Report No. 31.

Simon, A., A. Curini, S. E. Darby, and E. J. Langendoen. 2000. Bank and near-bank processes in an incised channel. *Geomorphology* 35(3-4): 193-217.

Simon, A., and C. R. Hupp. 1986. Channel evolution in modified Tennessee channels. Proceedings of the Fourth Interagency Sedimentation Conference, 2:5-71 to 5-82.

Simon, A., E. Langendoen, R. Bingner, R. R. Wells, Y.Yuan, and C. Alonso. 2004. Suspended-sediment transport and bed-material characteristics of Shades Creek, Alabama and Ecoregion 67: Developing water-quality criteria for suspended and bed-material sediment. USDA Agricultural Research Service National Sedimentation Laboratory Technical Report No. 43.

Simon, A., N. Pollen-Bankhead, V. Mahacek, and E. Langendoen. 2009. Quantifying reductions of mass-failure frequency and sediment loadings from streambanks using toe protection and other means: Lake Tahoe, United States. *Journal of the American Water Resources Association* 45(1): 170-186. doi: 10.1111/j.1752-1699.2008.00268.x

Simon, A., N. Pollen-Bankhead, and R. E. Thomas. 2011. Development and application of a deterministic bank stability and toe erosion model for stream restoration, In *Stream Restoration in Dynamic Fluvial Systems* (eds A. Simon, S. J. Bennett and J. M. Castro), American Geophysical Union, Washington, D.C., doi: 10.1029/2010GM001006.

Simon, A., and R. E. Thomas. 2002. Processes and forms of an unstable alluvial stream with resistant, cohesive streambeds. *Earth Surface Processes and Landforms* 27: 699-718.

Sims, J. T., 1993. Environmental soil testing for phosphorus. *Journal of Production Agriculture* 6: 501-507.

Sims, J. T., R. O. Maguire, A. B. Leytem, K. L. Gartley, and M. C. Pautler. 2002. Evaluation of Mehlich 3 as an agri-environmental soil phosphorus test for the Mid-Atlantic United States of America. *Soil Science Society of America Journal* 66: 2016-2032.

Smith, J. L., and J. W. Doran. 1996. Measurement and use of pH and electrical conductivity for soil quality analysis. In Methods for Assessing Soil Quality, 169-185, J.W. Doran and A.J. Jones, eds. Madison, WI.: Soil Science Society of America.

Sonzogni, W. C., S. C. Chapra, D. E. Armstrong, and T.J. Logan. 1982. Bioavailability of phosphorus inputs to lakes. *Journal of Environmental Quality* 11: 555-563.

SPSS. 2011. *SigmaPlot* for Windows. Ver. 12.5. Chicago, Ill.: SPSS, Inc.

Stafford-Glase, M., and J. P. Barlow. 1984. The role of lake water in enhancing sorption of phosphorus by stream particulates. In *Sediments and Water Interactions, Proceedings of the Third International Symposium on Interactions between Sediments and Water*, 451-459. Berlin, Germany: Springer.

Stauffer, R. E., 1985. Relationships between phosphorus loading and trophic state in calcareous lakes of southeast Wisconsin. *Limnology and Oceanography* 30: 123-145.

Storm, D.E., M. White, and M.D. Smolen. 2002. Modeling the Lake Eucha Basin using SWAT 489 2000. Final report submitted to the Tulsa Metropolitan Utility Authority, Tulsa, 490 Oklahoma. Department of Biosystems and Agricultural Engineering, Oklahoma State 491 University, Stillwater, Oklahoma.

Storm, D. E., and A. R. Mittelstet. 2015. Watershed Based Plan Support for the Illinois River and Spavinaw Creek Basins. DRAFT FINAL REPORT. Oklahoma Conservation Commission for the US EPA Region VI. Department of Biosystems and Agricultural Engineering, Oklahoma State University, Stillwater, Oklahoma.

Stutter, M. I., B. O. L. Demars, and S. J. Langan. 2010. River phosphorus cycling: Separating biotic and abiotic uptake during short-term changes in sewage effluent loading. *Water Research* 44: 4425-4436. doi: 10.1016/j.watres.2010.06.014.

Svendsen, L. M., and B. Kronvang. 1993. Retention of nitrogen and phosphorus in a Danish lowland river system: Implications for the export from the watershed. *Hydrobiologia* 251: 123-135. doi: 10.1007/BF00007172.

Syers, J. K., A. E. Johnston, and D. C. Curtin. 2008. Efficiency of soil and fertilizer phosphorus use. *Fertilizer and Plant Nutrition Bulletin* 18. FAO, Rome.

Thorne, C. R., 1981. Field measurements of rates of bank erosion and bank material strength. In Proc. *Florence Symposium on Erosion and Sediment Transport Measurement*, June 1981. International Association of Hydrological Sciences.

Thorne, C. R., 1982. Processes and mechanisms of river bank erosion. In *Gravel-Bed Rivers*, eds. R.D. Hey, J.C. Bathurst, and C.R. Thorne. New York, N.Y.: John Wiley & Sons.

Tortorelli, R. L., 2006. Nutrient concentrations, loads, and yields in the Eucha-Spavinaw Basin, Arkansas and Oklahoma, 2002-2004. U.S. Department of the Interior and U.S. Geological Survaye Scientific Investigations Report 2006-5250.

Triplett, L. D., D. R. Engstrom, and M. B. Edlund. 2009. A whole-basin stratigraphic record of sediment and phosphorus loading to the St. Croix River, USA. *Journal of Paleolimnology* 41(4): 659-677.

Tulsa Metropolitan Utility Authority. 2001. Eucha/Spavinaw Watershed Project. *http://www.tulsawater.com/watrhist.html*

U.S. Army Corps of Engineers (USACE). 1983. Sacramento River and tributaries bank protection and erosion control investigation. California sediment transport studies. Sacramento District. Sacramento, CA.: USACE.

U.S. Department of Agriculture – Natural Resources Conservation Service (USDA-NRCS). 2013. Web Soil Survey. *http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm.*

U.S. Environmental Protection Agency (US EPA). 1996. Acid digestion of sediments, sludges, and soils, Revision 2. *http://www.epa.gov/osw/hazard/testmethods/sw846/pdfs/3050b.pdf*. Accessed 9/1/2014.

U.S. Geological Survey (USGS). 2015. Mineral Commodity Summaries-Phosphate Rock. *http://www.minerals.usgs.gov*.

Vadas, P. A., M. S. Srinivasa, P. J. A. Kleinman, J. P. Schmidt, and A. L. Allen. 2007. Hydrology and groundwater nutrient concentrations in a ditch-drained agroecosystem. *Journal of Soil and Water Conservation* 62: 178-188.

van der Zee, S. E. A. T. M, and W. H. van Riemsdijk. 1988. Model for long-term phosphate reaction kinetics in soil. *Journal of Environmental Quality* 17: 35-41.

Vitousek, P. M., R. Naylor, T. Crews, M. B. David, L. E. Drinkwater, E. Holland, et al.. 2009. Nutrient imbalances in agricultural development. *Science* 324: 1519-1520. doi: 10.1126/science.1170261.

Walling, D. E., P. Owen, and G. J. L. Leeks. 1999. Fingerprinting suspended sediment sources in the catchment of the River Ouse, Yorkshire. *Hydrological Processes* 13: 977-992.

Walton, C. P., and C. F. Lee. 1972. A biological evaluation of the molybdenum blue method for orthophosphate analysis. *Verhandlungen des Internationalen Verein Limnologie* 18: 676-684.

Wilkin, D. C., and S. J. Hebel. 1982. Erosion, redeposition, and delivery of sediment to Midwestern streams. *Water Resources Research* 18: 1278-1282.

Williams, J. D. H., J. -M. Jaquet, and R. L. Thomas. 1980. Availability of *Scenedesmus quadricauda* of different forms of phosphorus in sedimentary materials from the Great Lakes. *Limnology and Oceanography* 25: 1-11.

Wilson, C. G., R. A. Kuhnle, D. D. Bosch, J. L. Steiner, P. J. Starks, M. D. Tomer, and G. V. Wilson. 2008. Quantifying relative contributions from sediment sources in Conservation Effects Assessment Project watersheds. *Journal of Soil and Water Conservation* 63(6): 523-531.

Young, T. C., and J. V. DePinto. 1982. Algal-availability of particulate phosphorus from diffuse and point sources in the lower Great Lakes basin. *Hydrobiologia* 91: 111-119.

Zaimes, G. N., R. C. Schultz, and T. M. Isenhart. 2008. Streambank soil and phosphorus losses under different riparian land-uses in Iowa. *Journal of the American Water Resources Association* 44(4):935-947. DOI: 10.1111/j.1752-1688.2008.00210.x.

Zhang, H. 2001. Fertility of Oklahoma agricultural soils. *Better Crops* 85: 18-20.

APPENDIX A

2-way ANOVA for retreat with and without a buffer (Figure 2.6)

General Linear Model: Retreat (m/y versus Buffer Width, Period_1_1

Factor	Type	Levels	Values
Buffer Width_1_1	fixed	2	0, 1
Period_1_1	fixed	3	1, 2, 3

Analysis of Variance for Retreat (m/yr)_1_1, using Adjusted SS for Tests

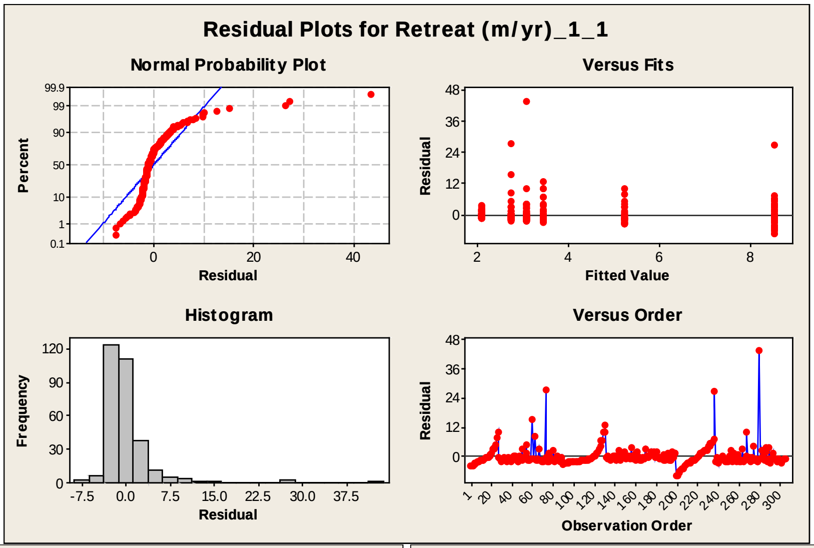
Source	DF	Seq SS	Adj SS	Adj MS	F	P
Buffer Width_1_1	1	640.38	638.07	638.07	33.37	0.000
Period_1_1	2	333.46	462.07	231.04	12.08	0.000
Buffer Width_1_1*Period_1_1	2	218.39	218.39	109.19	5.71	0.004
Error	299	5717.22	5717.22	19.12		
Total	304	6909.45				

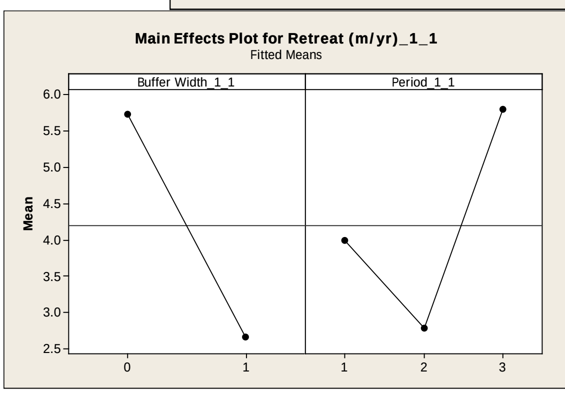
S = 4.37277   R-Sq = 17.26%   R-Sq(adj) = 15.87%

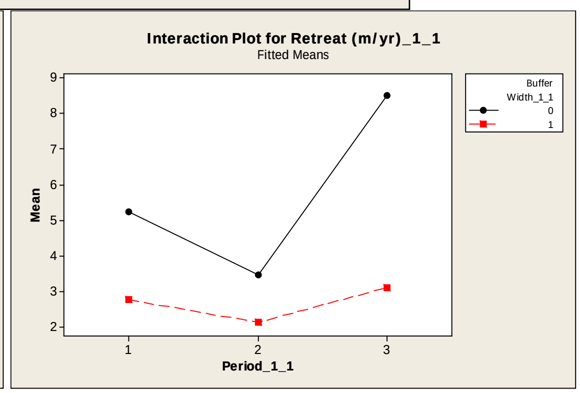
Unusual Observations for Retreat (m/yr)_1_1

Obs	Retreat (m/yr)_1_1	Fit	SE Fit	Residual	St Resid	
27	15.3800	5.2312	0.8415	10.1488	2.37	R
60	17.9320	2.7612	0.5553	15.1708	3.50	R
73	30.0000	2.7612	0.5553	27.2388	6.28	R
129	13.2800	3.4500	0.6747	9.8300	2.28	R
130	13.3050	3.4500	0.6747	9.8550	2.28	R
131	16.0400	3.4500	0.6747	12.5900	2.91	R
237	34.9800	8.5079	0.7094	26.4721	6.14	R
268	12.9333	3.1038	0.5303	9.8295	2.26	R
280	46.5867	3.1038	0.5303	43.4828	10.02	R

R denotes an observation with a large standardized residual.







Grouping Information Using Tukey Method and 95.0% Confidence

Buffer Width_1_1	N	Mean	Grouping
0	107	5.730	A
1	198	2.662	B

Means that do not share a letter are significantly different.

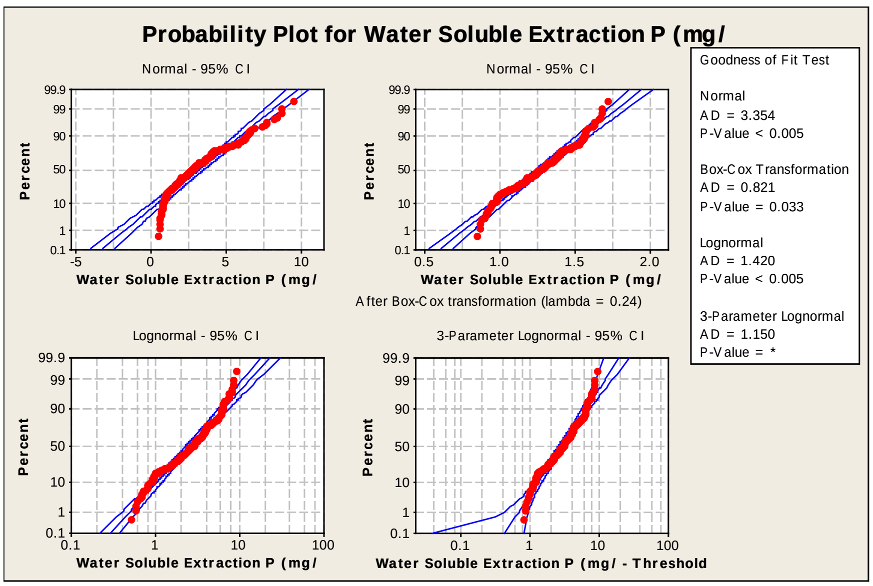
Grouping Information Using Tukey Method and 95.0% Confidence

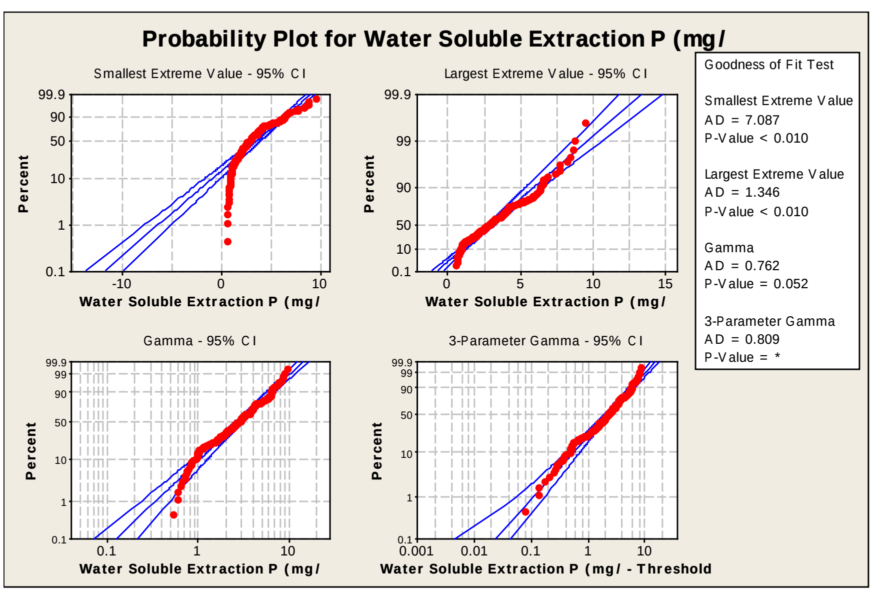
Period_1_1	N	Mean	Grouping
3	106	5.806	A
1	89	3.996	B
2	110	2.786	B

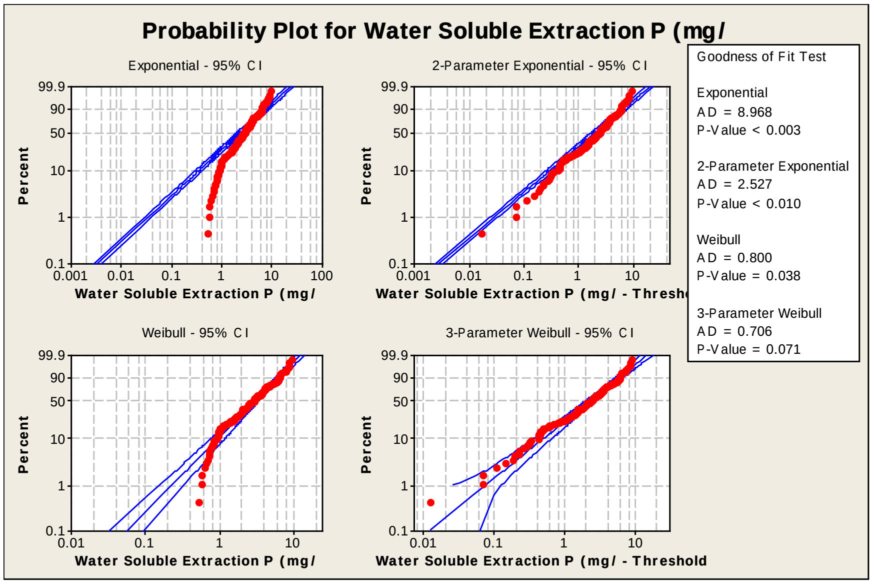
Means that do not share a letter are significantly different.

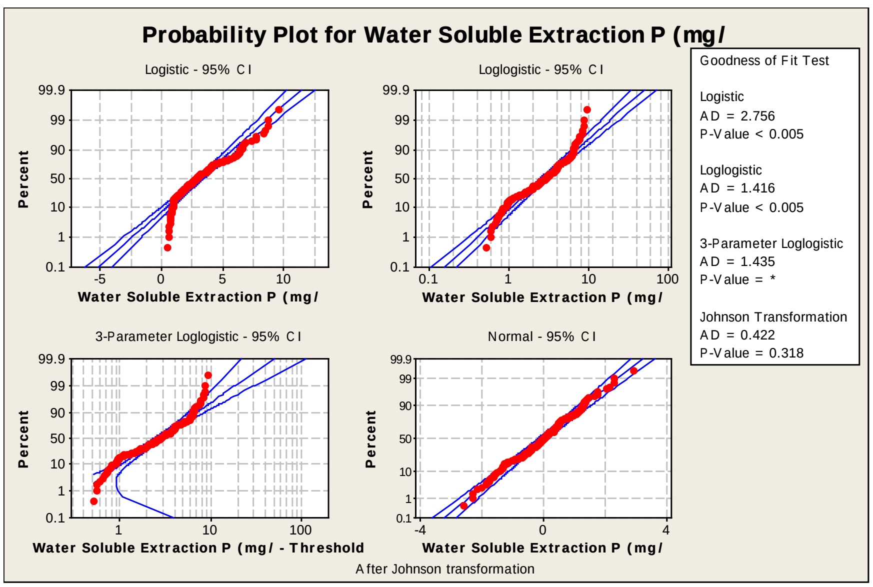
APPENDIX B

Minitab probability plots for Observed WSP

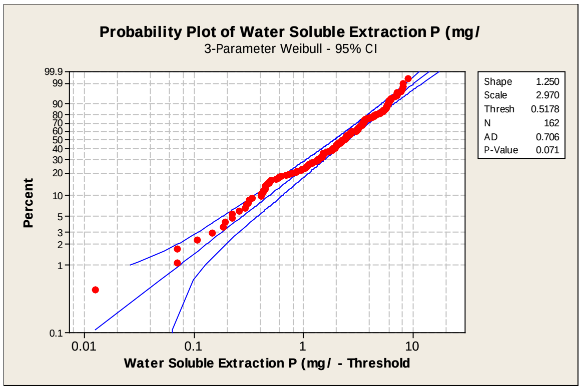




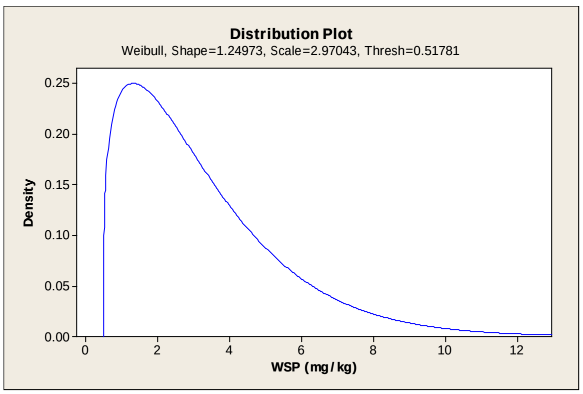

Probability Plot for Water Soluble Extraction P (mg/
Exponential - 95% CI
2-Parameter Exponential - 95% CI
Weibull - 95% CI
3-Parameter Weibull - 95% CI
Percent
Water Soluble Extraction P (mg/
Water Soluble Extraction P (mg/ - Threshold
Goodness of Fit Test
Exponential
AD = 8.968
P-Value < 0.003
2-Parameter Exponential
AD = 2.527
P-Value < 0.010
Weibull
AD = 0.800
P-Value = 0.038
3-Parameter Weibull
AD = 0.706
P-Value = 0.071


Probability Plot for Water Soluble Extraction P (mg/
Logistic - 95% CI
Loglogistic - 95% CI
3-Parameter Loglogistic - 95% CI
Normal - 95% CI
Percent
Water Soluble Extraction P (mg/
Water Soluble Extraction P (mg/ - Threshold
After Johnson transformation
Goodness of Fit Test
Logistic
AD = 2.756
P-Value < 0.005
Loglogistic
AD = 1.416
P-Value < 0.005
3-Parameter Loglogistic
AD = 1.435
P-Value = *
Johnson Transformation
AD = 0.422
P-Value = 0.318

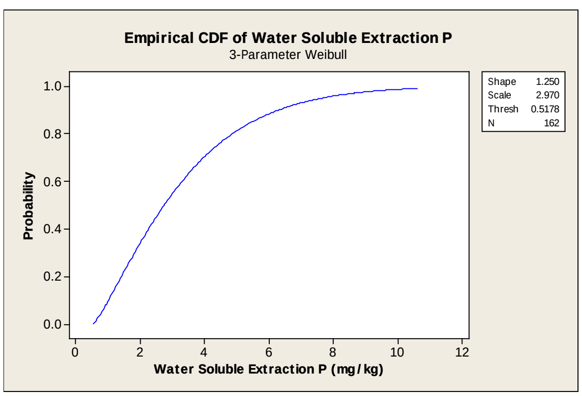
Observed WSP 3 Parameter Weibull Probability Plot



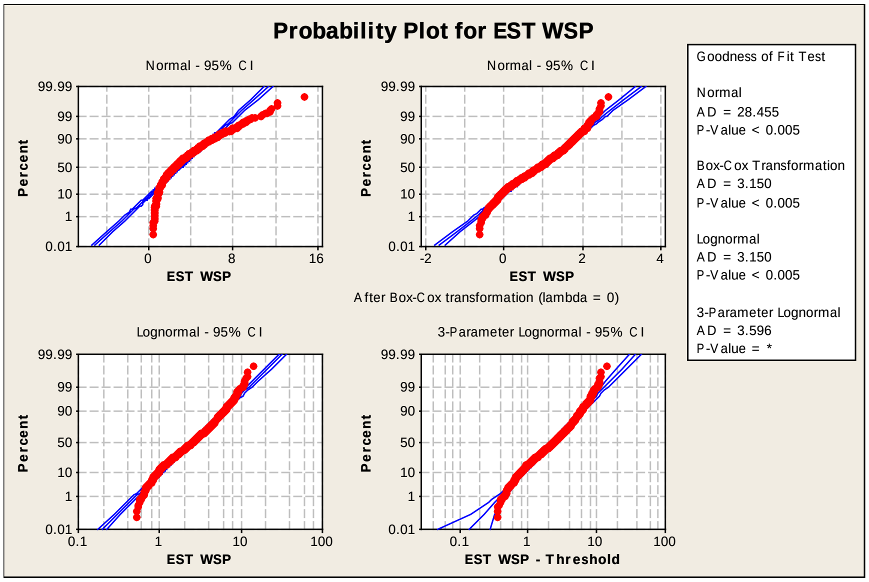
Observed WSP PDF

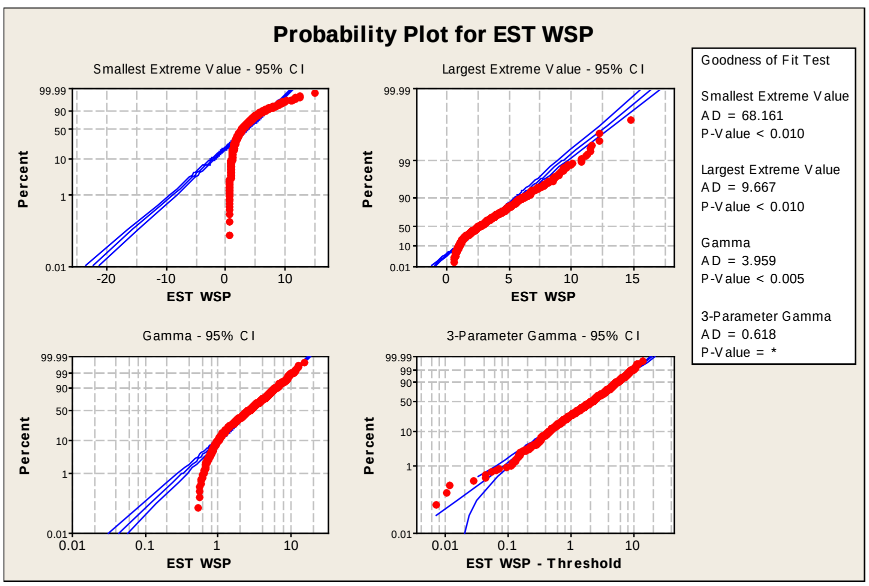


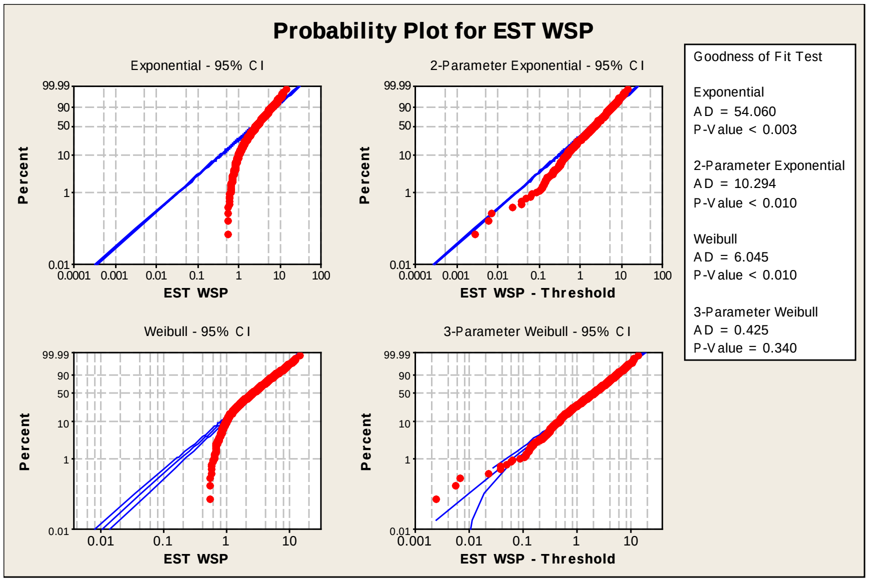
Observed WSP CDF

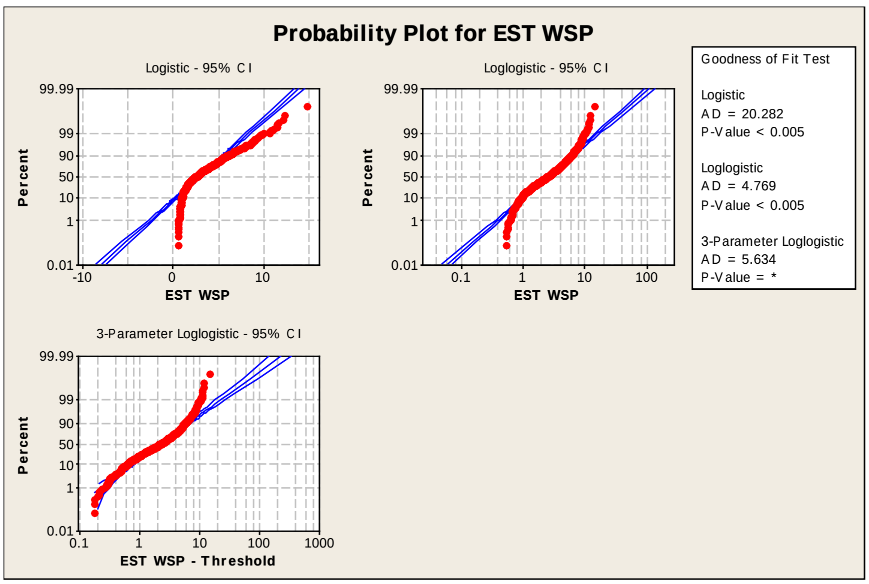


Minitab probability plots for Monte Carlo WSP

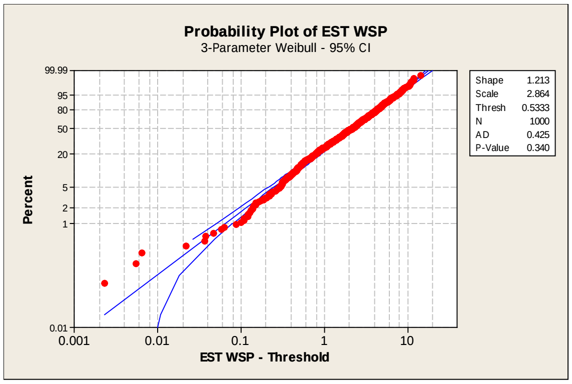




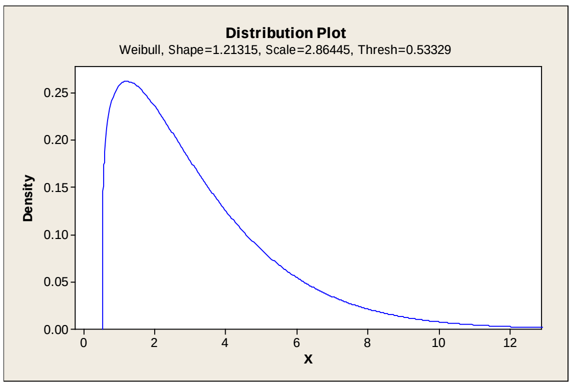

Probability Plot for EST WSP
Exponential - 95% CI
2-Parameter Exponential - 95% CI
Weibull - 95% CI
3-Parameter Weibull - 95% CI
Percent
EST WSP
EST WSP - Threshold
Goodness of Fit Test
Exponential
AD = 54.060
P-Value < 0.003
2-Parameter Exponential
AD = 10.294
P-Value < 0.010
Weibull
AD = 6.045
P-Value < 0.010
3-Parameter Weibull
AD = 0.425
P-Value = 0.340


Probability Plot for EST WSP
Logistic - 95% CI
Loglogistic - 95% CI
3-Parameter Loglogistic - 95% CI
Percent
EST WSP
EST WSP - Threshold
Goodness of Fit Test
Logistic
AD = 20.282
P-Value < 0.005
Loglogistic
AD = 4.769
P-Value < 0.005
3-Parameter Loglogistic
AD = 5.634
P-Value = *

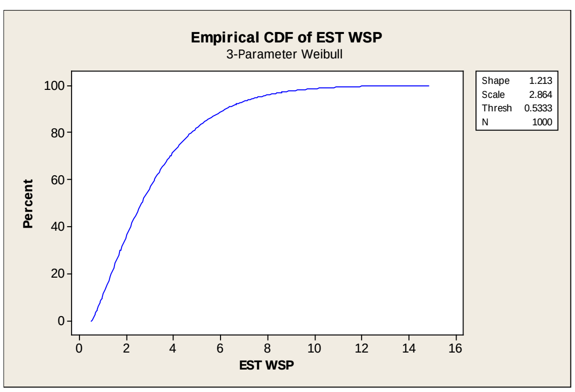
Monte Carlo WSP 3 Parameter Weibull probability plot



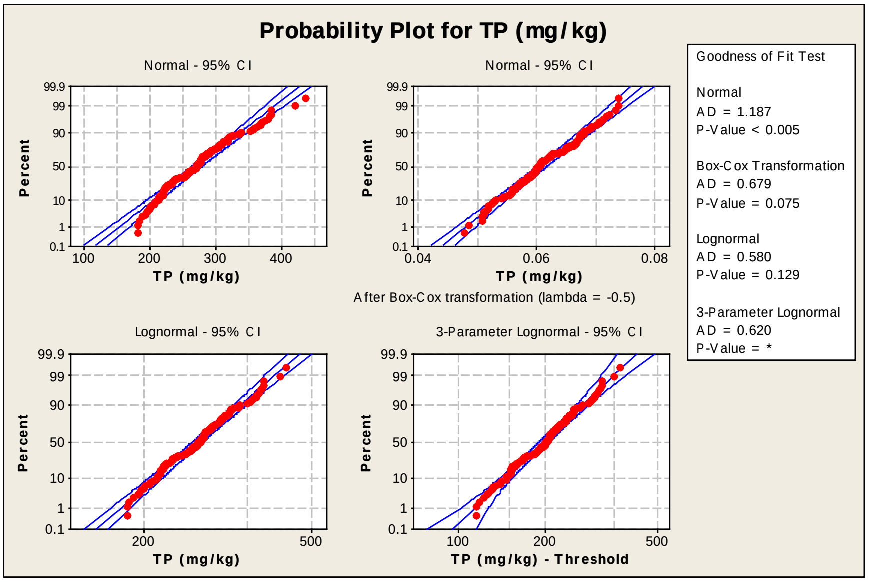
Monte Carlo WSP PDF

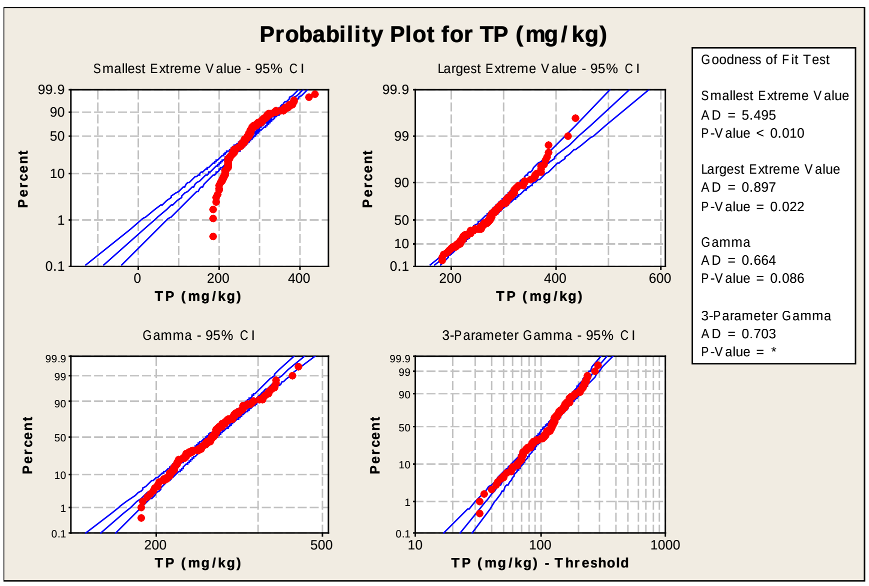


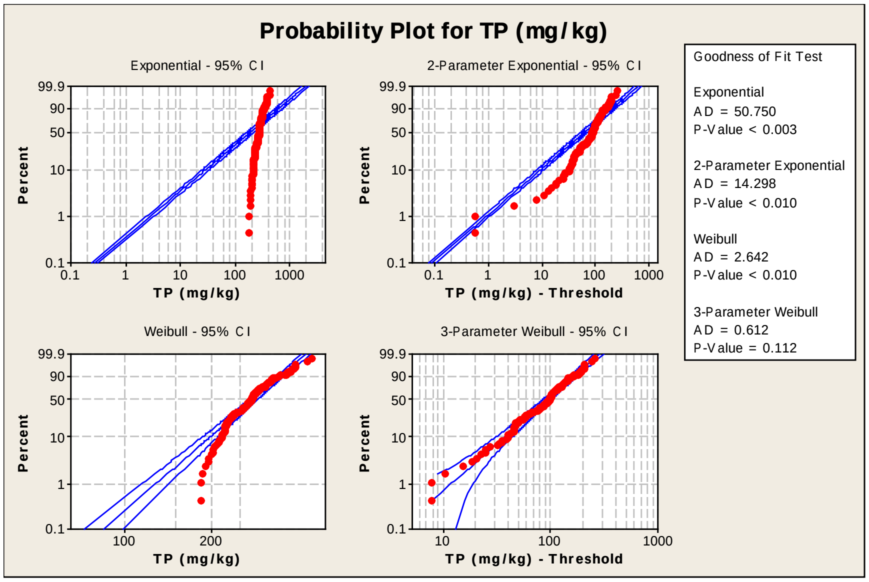
Monte Carlo WSP CDF



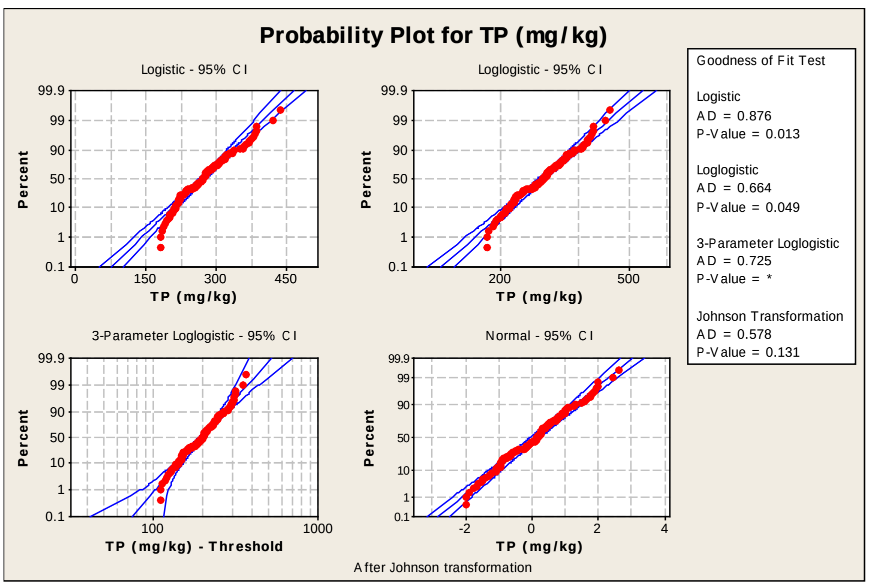
Minitab probability plots for Observed TP





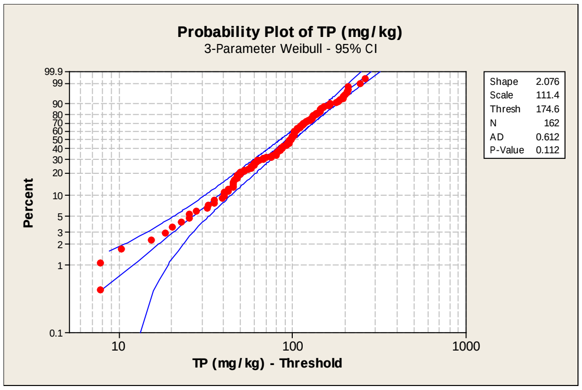


Probability Plot for TP (mg/kg)
Exponential - 95% CI
2-Parameter Exponential - 95% CI
Weibull - 95% CI
3-Parameter Weibull - 95% CI
Percent
TP (mg/kg)
TP (mg/kg) - Threshold
Goodness of Fit Test
Exponential
AD = 50.750
P-Value < 0.003
2-Parameter Exponential
AD = 14.298
P-Value < 0.010
Weibull
AD = 2.642
P-Value < 0.010
3-Parameter Weibull
AD = 0.612
P-Value = 0.112

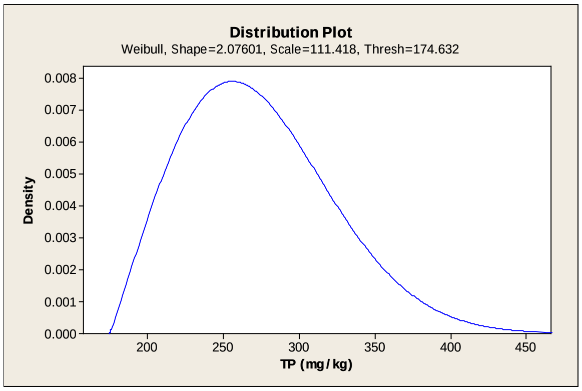


Probability Plot for TP (mg/kg)
Logistic - 95% CI
Loglogistic - 95% CI
3-Parameter Loglogistic - 95% CI
Normal - 95% CI
Percent
TP (mg/kg)
TP (mg/kg) - Threshold
After Johnson transformation
Goodness of Fit Test
Logistic
AD = 0.876
P-Value = 0.013
Loglogistic
AD = 0.664
P-Value = 0.049
3-Parameter Loglogistic
AD = 0.725
P-Value = *
Johnson Transformation
AD = 0.578
P-Value = 0.131

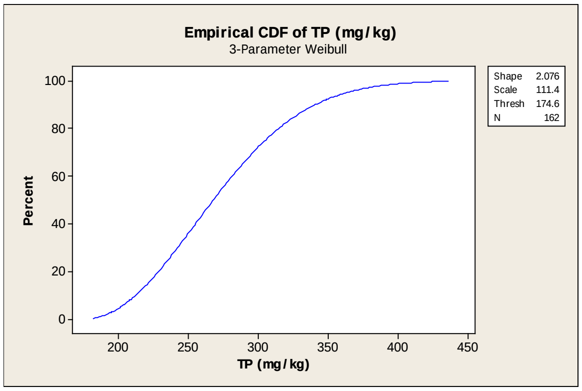
Observed TP 3 Parameter Weibull probability plot



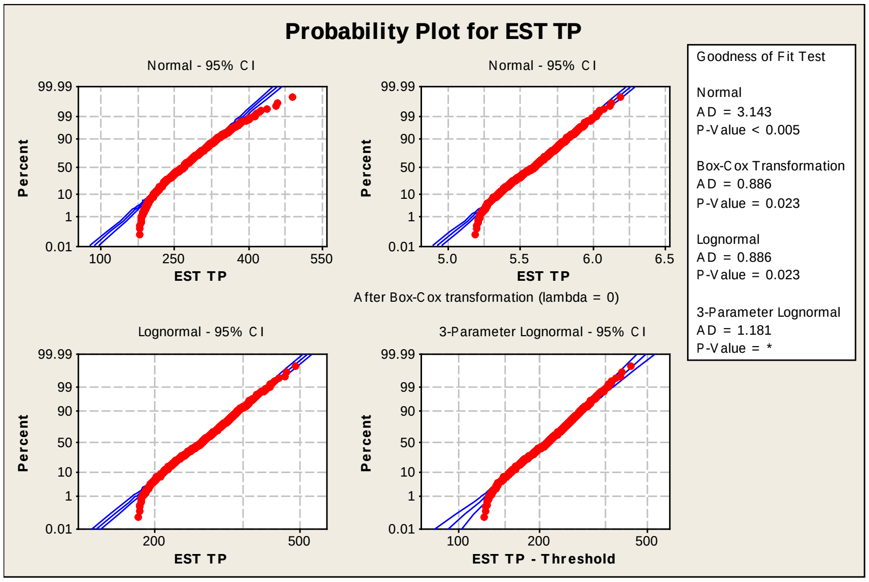
Observed TP PDF

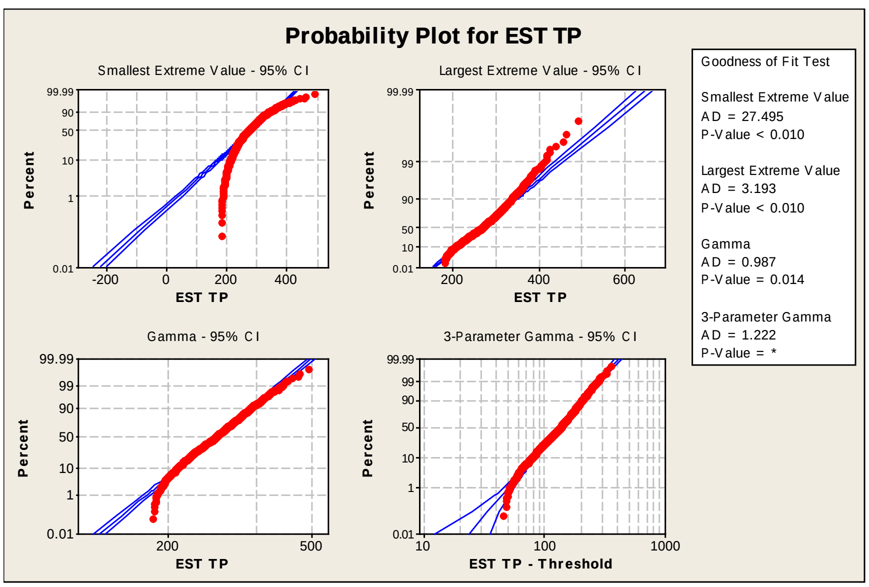


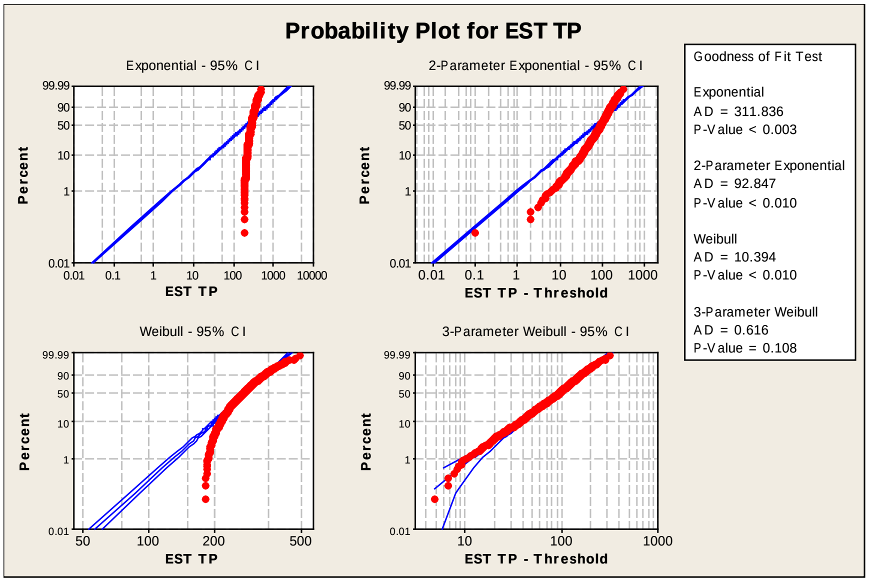
Observed TP CDF

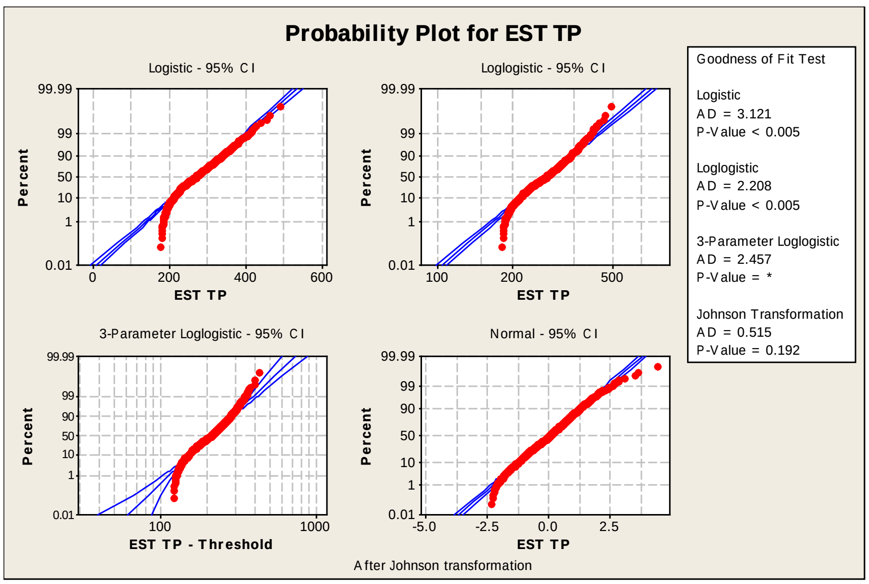


Minitab probability plots for Monte Carlo TP

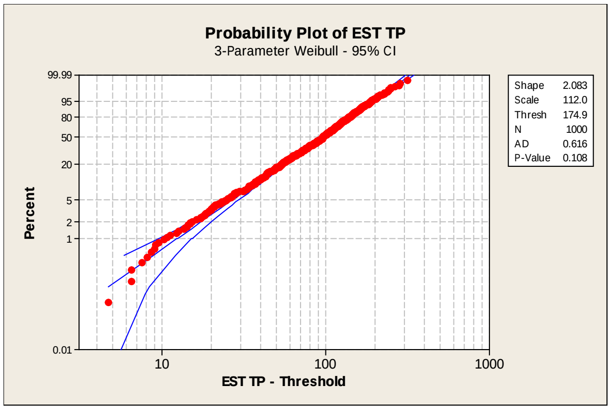




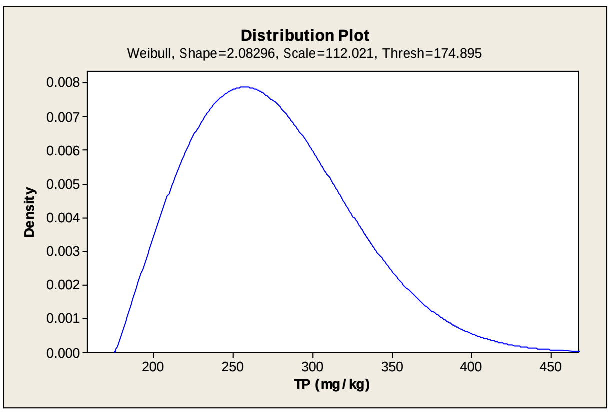

Probability Plot for EST TP
Exponential - 95% CI
2-Parameter Exponential - 95% CI
Weibull - 95% CI
3-Parameter Weibull - 95% CI
Percent
EST TP
EST TP - Threshold
Goodness of Fit Test
Exponential
AD = 311.836
P-Value < 0.003
2-Parameter Exponential
AD = 92.847
P-Value < 0.010
Weibull
AD = 10.394
P-Value < 0.010
3-Parameter Weibull
AD = 0.616
P-Value = 0.108


Probability Plot for EST TP
Logistic - 95% CI
Loglogistic - 95% CI
3-Parameter Loglogistic - 95% CI
Normal - 95% CI
Percent
EST TP
EST TP - Threshold
After Johnson transformation
Goodness of Fit Test
Logistic
AD = 3.121
P-Value < 0.005
Loglogistic
AD = 2.208
P-Value < 0.005
3-Parameter Loglogistic
AD = 2.457
P-Value = *
Johnson Transformation
AD = 0.515
P-Value = 0.192

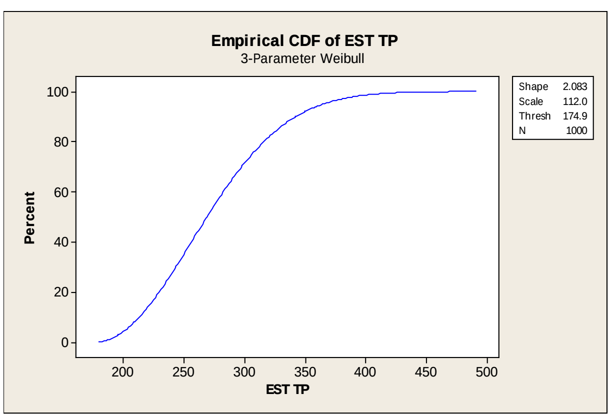
Monte Carlo TP 3 Parameter Weibull probability plot



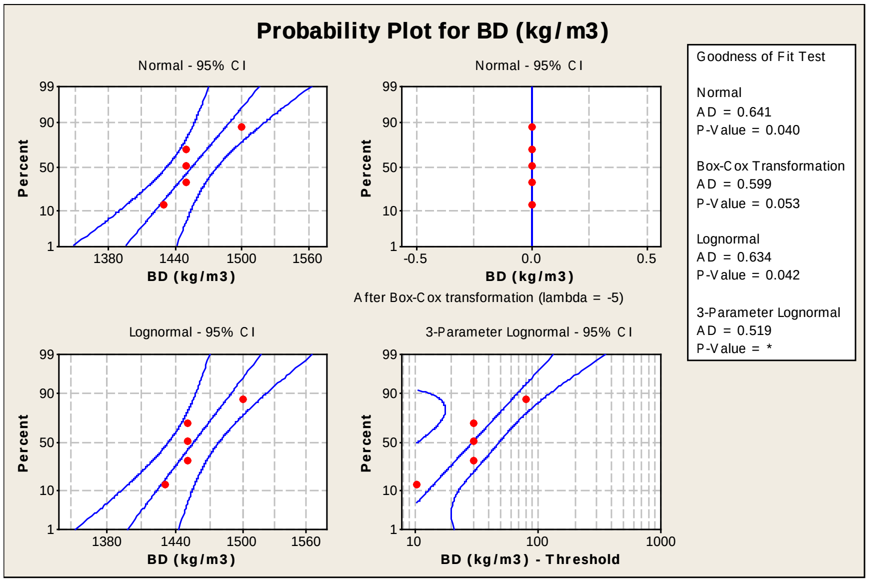
Monte Carlo TP PDF

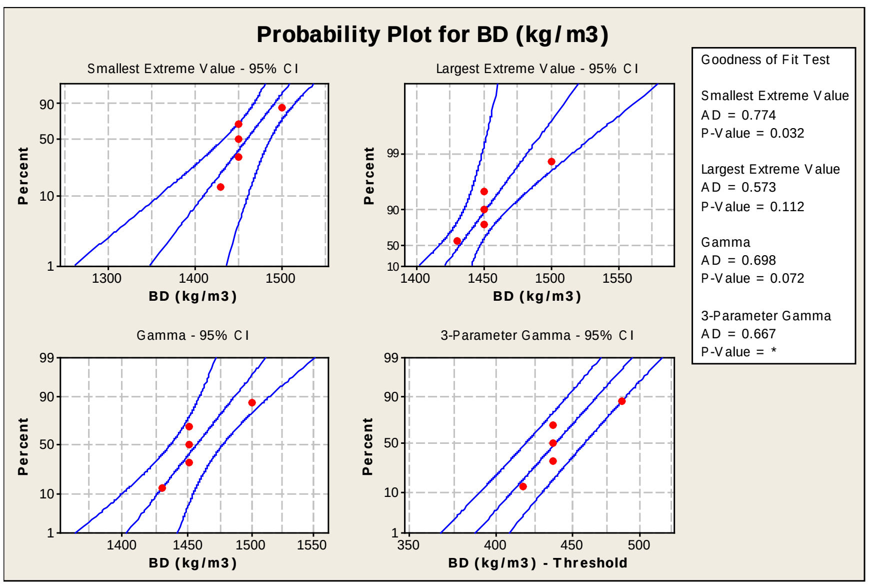


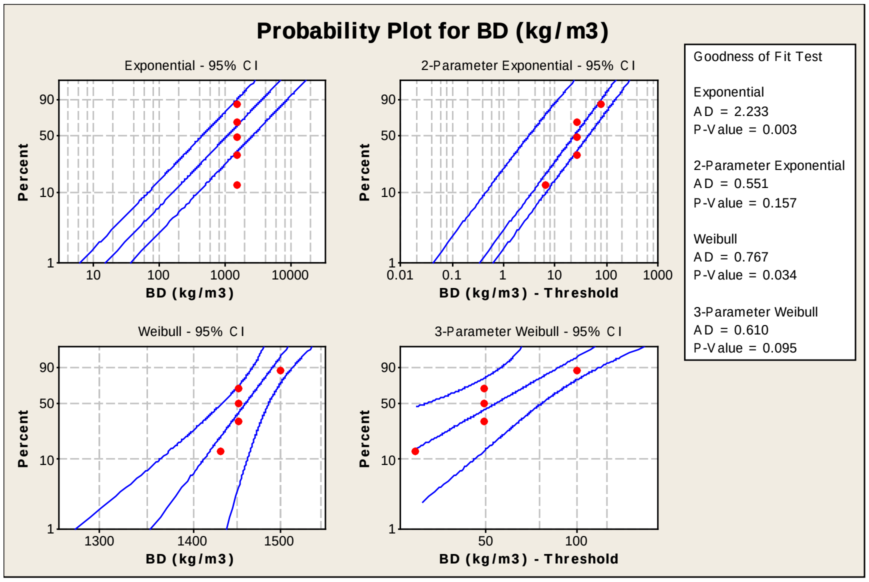
Monte Carlo TP CDF

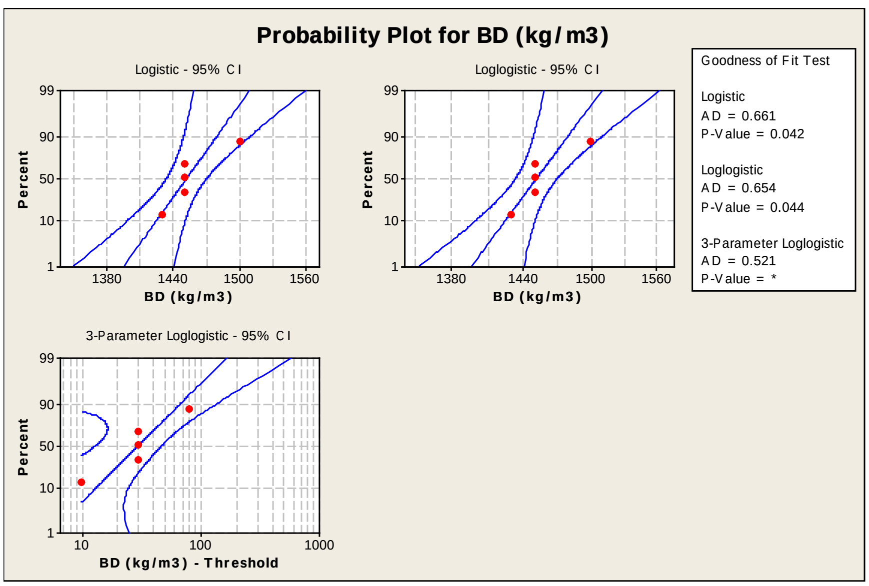


Minitab probability plots for Observed Bulk Density

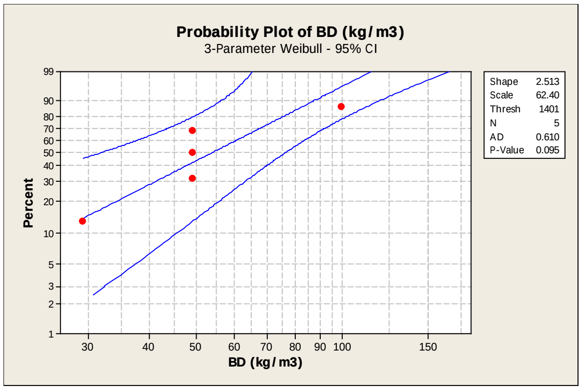




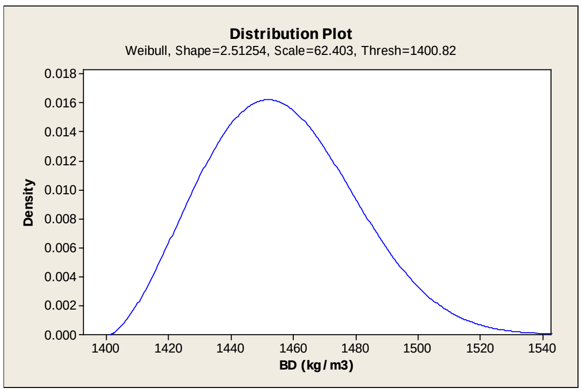

Probability Plot for BD (kg/m3)
Exponential - 95% CI
2-Parameter Exponential - 95% CI
Weibull - 95% CI
3-Parameter Weibull - 95% CI
Percent
BD (kg/m3)
BD (kg/m3) - Threshold
Goodness of Fit Test
Exponential
AD = 2.233
P-Value = 0.003
2-Parameter Exponential
AD = 0.551
P-Value = 0.157
Weibull
AD = 0.767
P-Value = 0.034
3-Parameter Weibull
AD = 0.610
P-Value = 0.095


Probability Plot for BD (kg/m3)
Logistic - 95% CI
Loglogistic - 95% CI
3-Parameter Loglogistic - 95% CI
Percent
BD (kg/m3)
BD (kg/m3) - Threshold
Goodness of Fit Test
Logistic
AD = 0.661
P-Value = 0.042
Loglogistic
AD = 0.654
P-Value = 0.044
3-Parameter Loglogistic
AD = 0.521
P-Value = *

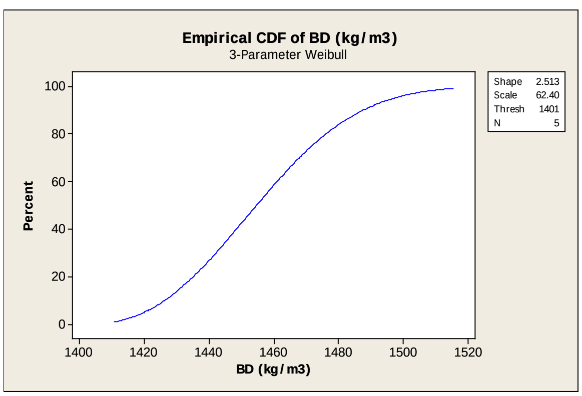
Observed Bulk Density 3 Parameter Weibull probability plot



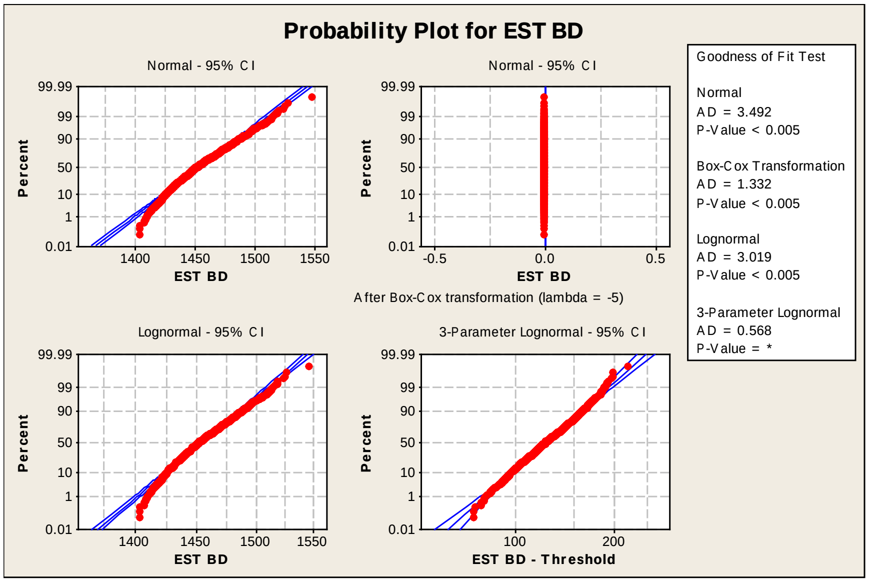
Observed Bulk Density PDF

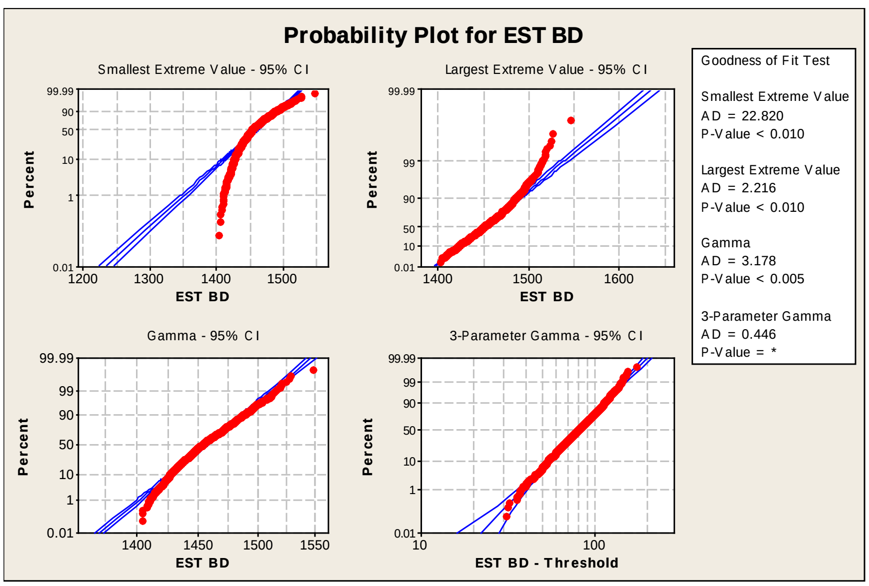


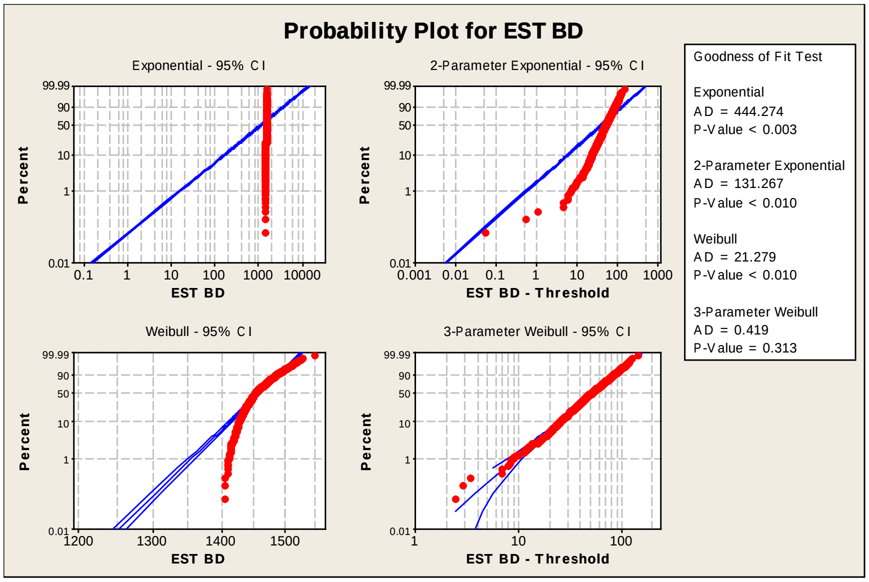
Observed Bulk Density CDF

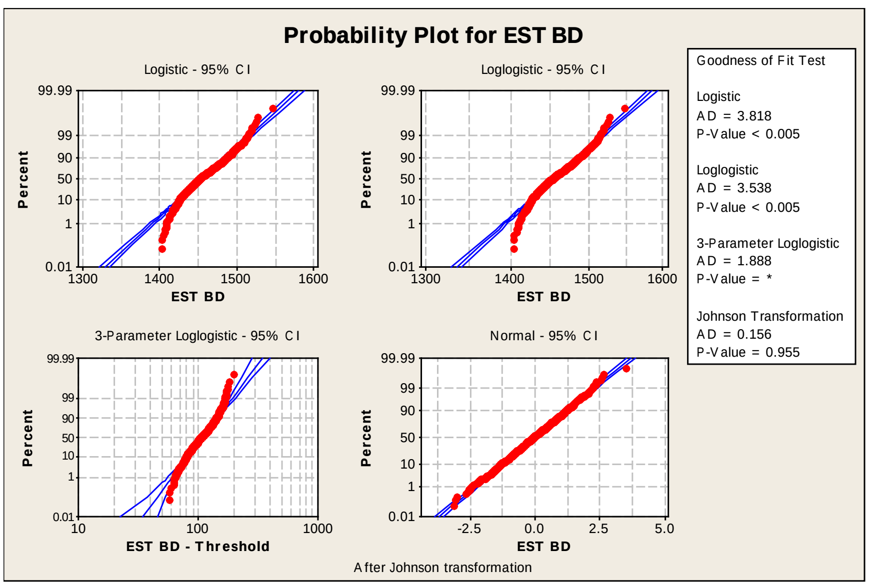


Minitab probability plots for Monte Carlo Bulk Density

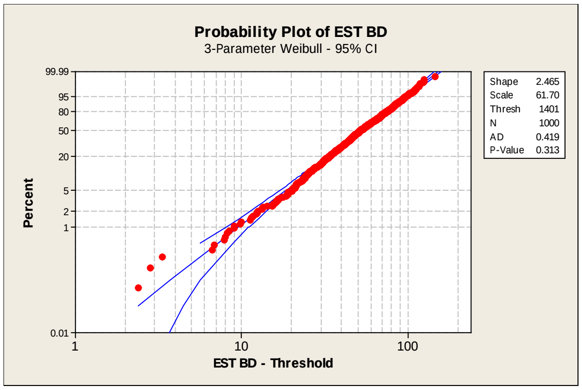




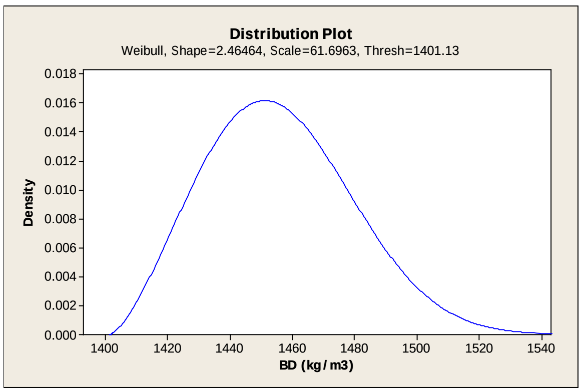

Probability Plot for EST BD
Exponential - 95% CI
2-Parameter Exponential - 95% CI
Weibull - 95% CI
3-Parameter Weibull - 95% CI
Percent
EST BD
EST BD - Threshold
Goodness of Fit Test
Exponential
AD = 444.274
P-Value < 0.003
2-Parameter Exponential
AD = 131.267
P-Value < 0.010
Weibull
AD = 21.279
P-Value < 0.010
3-Parameter Weibull
AD = 0.419
P-Value = 0.313


Probability Plot for EST BD
Logistic - 95% CI
Loglogistic - 95% CI
3-Parameter Loglogistic - 95% CI
Normal - 95% CI
Percent
EST BD
EST BD - Threshold
After Johnson transformation
Goodness of Fit Test
Logistic
AD = 3.818
P-Value < 0.005
Loglogistic
AD = 3.538
P-Value < 0.005
3-Parameter Loglogistic
AD = 1.888
P-Value = *
Johnson Transformation
AD = 0.156
P-Value = 0.955

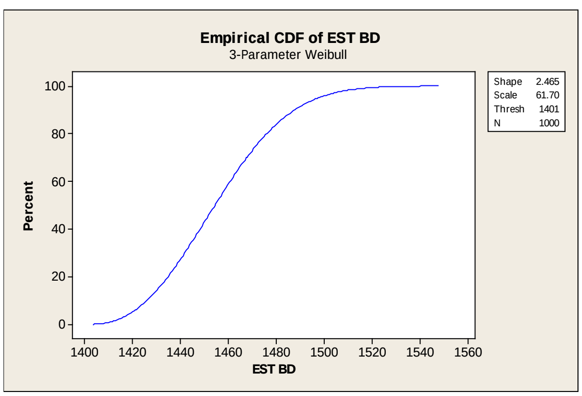
Monte Carlo Bulk Density 3 Parameter Weibull probability plot



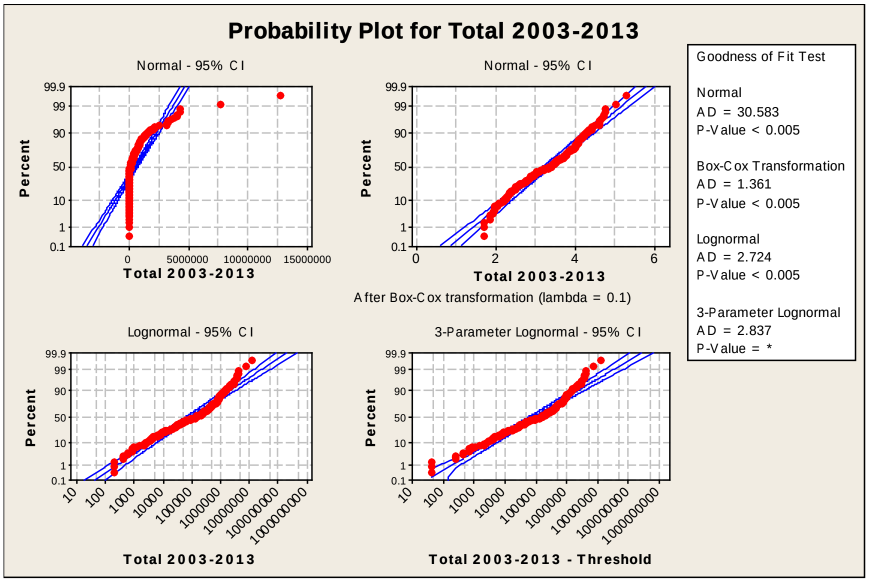
Monte Carlo Bulk Density PDF

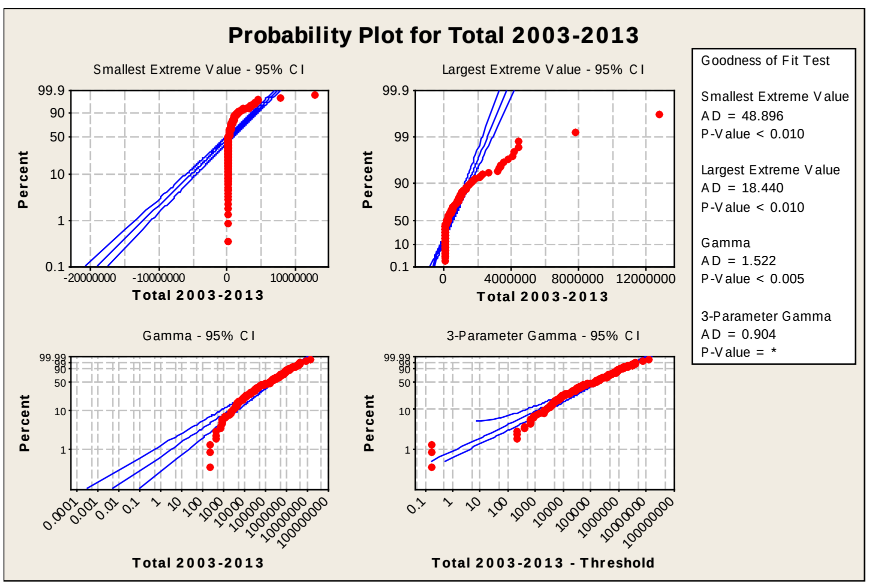


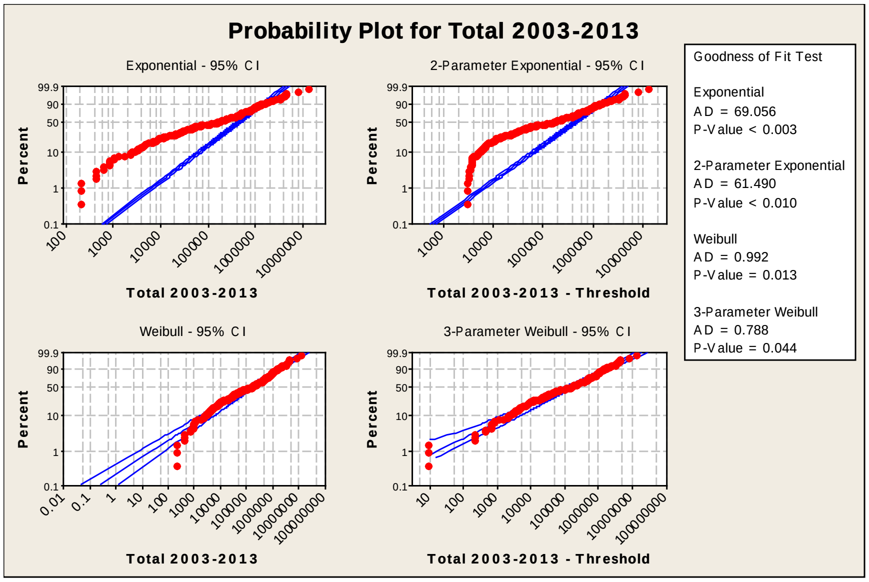
Monte Carlo Bulk Density CDF

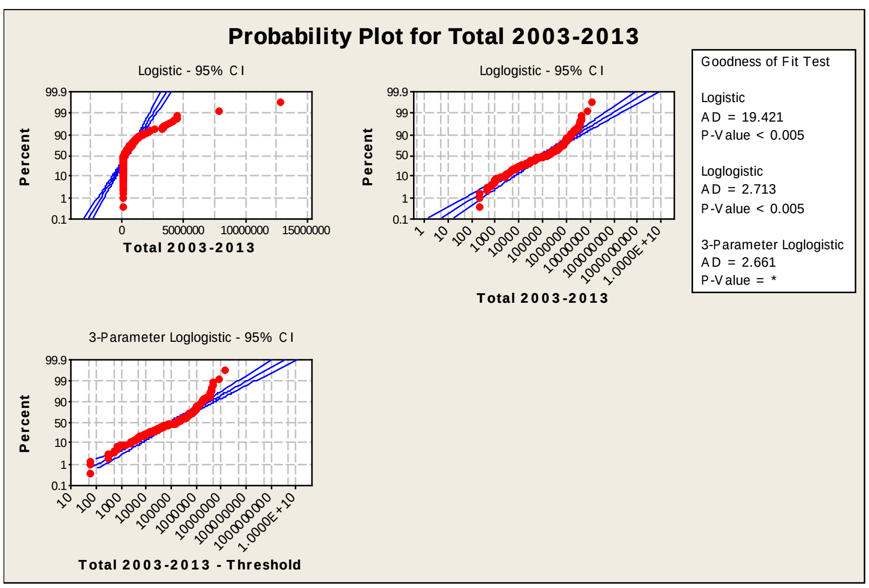


Minitab probability plots for Observed Eroded Area

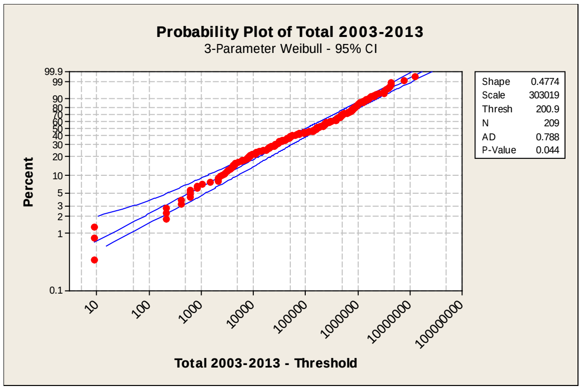




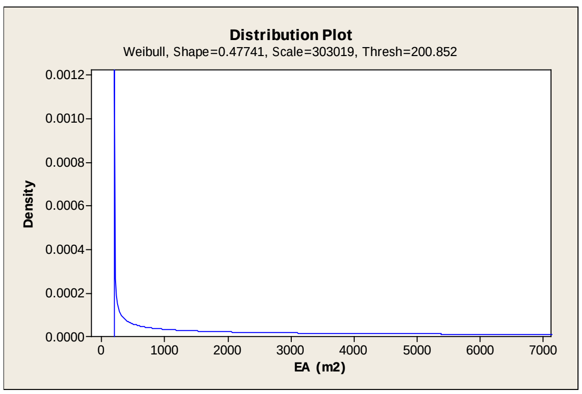

Probability Plot for Total 2003-2013
Exponential - 95% CI
2-Parameter Exponential - 95% CI
Weibull - 95% CI
3-Parameter Weibull - 95% CI
Percent
Total 2003-2013
Total 2003-2013 - Threshold
Goodness of Fit Test
Exponential
AD = 69.056
P-Value < 0.003
2-Parameter Exponential
AD = 61.490
P-Value < 0.010
Weibull
AD = 0.992
P-Value = 0.013
3-Parameter Weibull
AD = 0.788
P-Value = 0.044


Probability Plot for Total 2003-2013
Logistic - 95% CI
Loglogistic - 95% CI
3-Parameter Loglogistic - 95% CI
Percent
Total 2003-2013
Total 2003-2013 - Threshold
Goodness of Fit Test
Logistic
AD = 19.421
P-Value < 0.005
Loglogistic
AD = 2.713
P-Value < 0.005
3-Parameter Loglogistic
AD = 2.661
P-Value = *

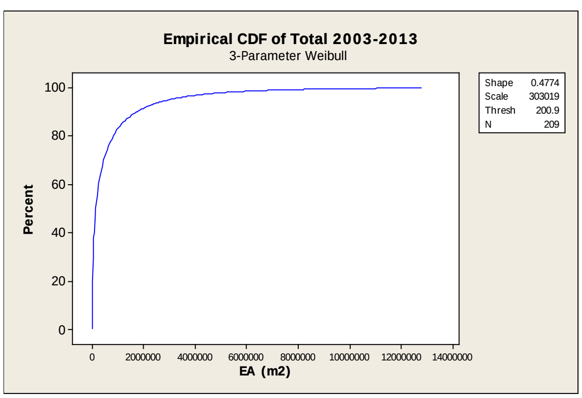
Observed Eroded Area 3 Parameter Weibull probability plot



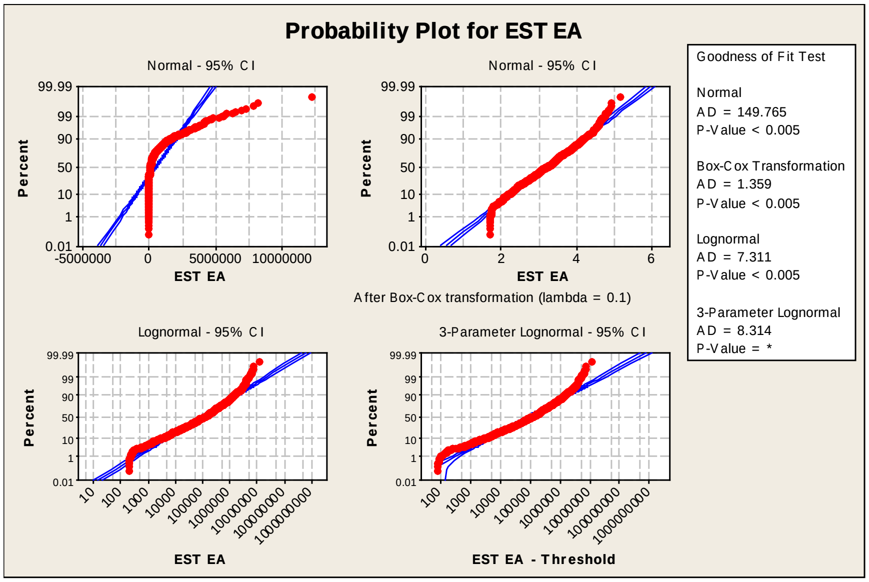
Observed Eroded Area PDF

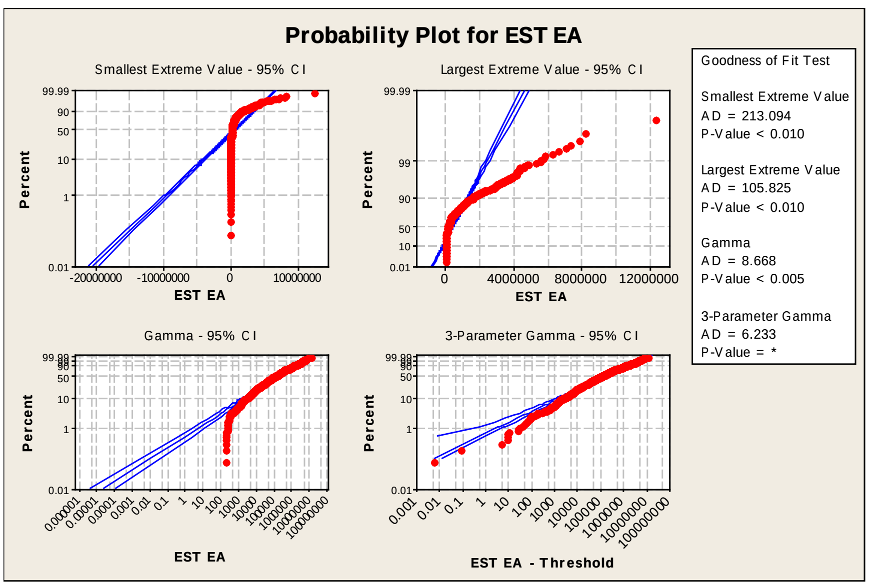


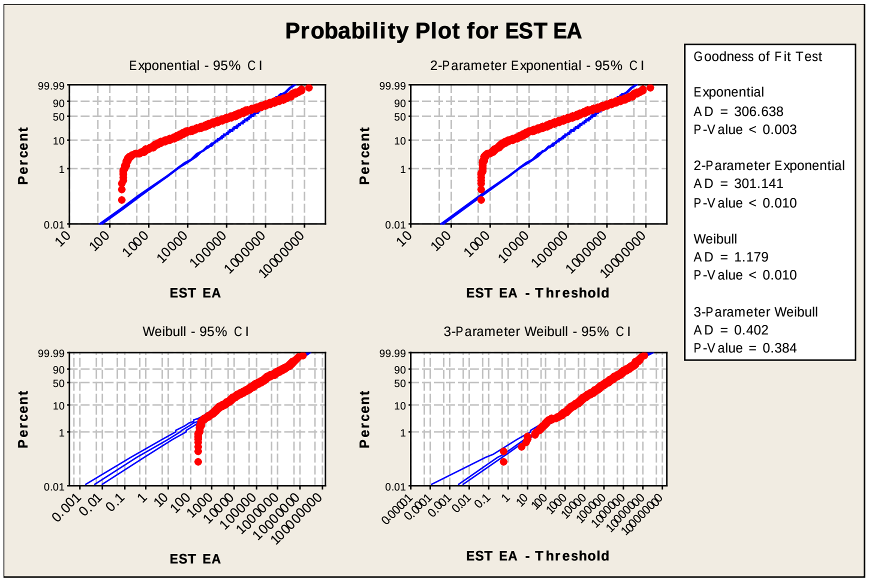
Observed Eroded Area CDF

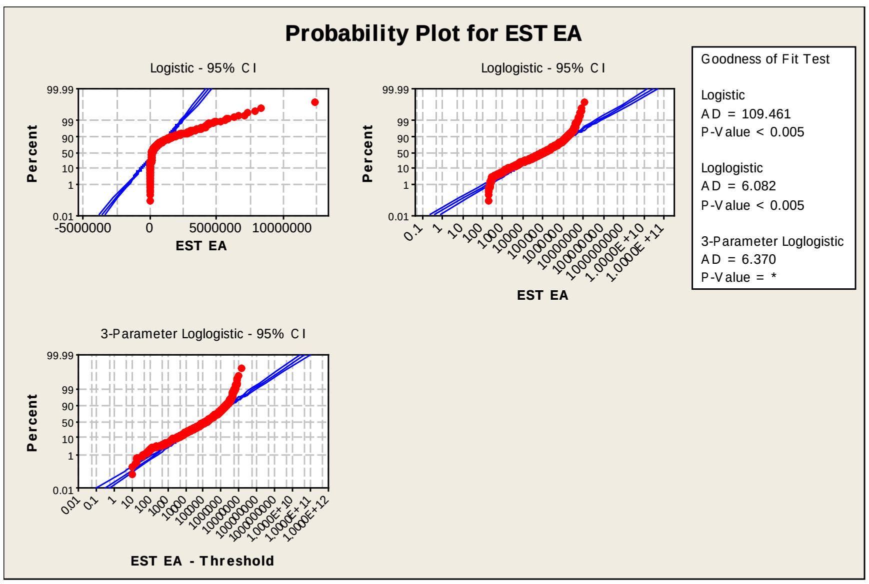


Minitab probability plots for Monte Carlo Eroded Area

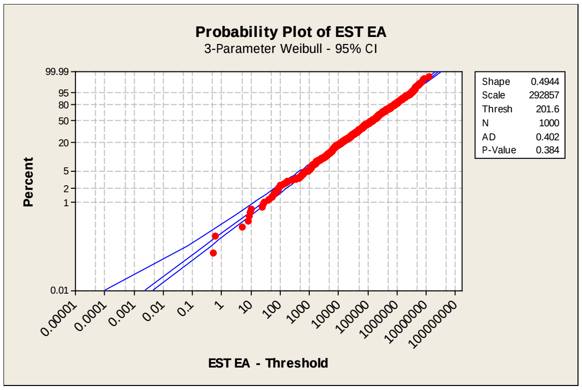




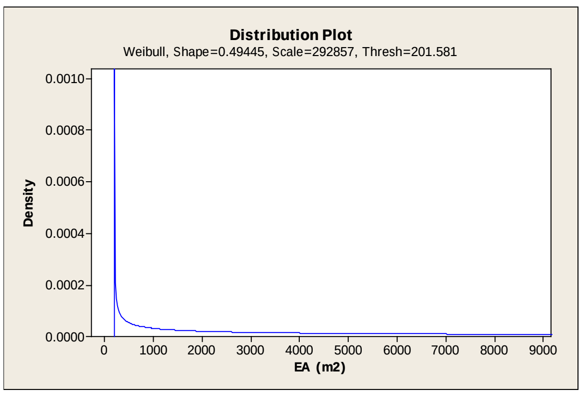

Probability Plot for EST EA
Exponential - 95% CI
2-Parameter Exponential - 95% CI
Weibull - 95% CI
3-Parameter Weibull - 95% CI
Percent
EST EA
EST EA - Threshold
Goodness of Fit Test
Exponential
AD = 306.638
P-Value < 0.003
2-Parameter Exponential
AD = 301.141
P-Value < 0.010
Weibull
AD = 1.179
P-Value < 0.010
3-Parameter Weibull
AD = 0.402
P-Value = 0.384


Probability Plot for EST EA
Logistic - 95% CI
Loglogistic - 95% CI
3-Parameter Loglogistic - 95% CI
Percent
EST EA
EST EA - Threshold
Goodness of Fit Test
Logistic
AD = 109.461
P-Value < 0.005
Loglogistic
AD = 6.082
P-Value < 0.005
3-Parameter Loglogistic
AD = 6.370
P-Value = *

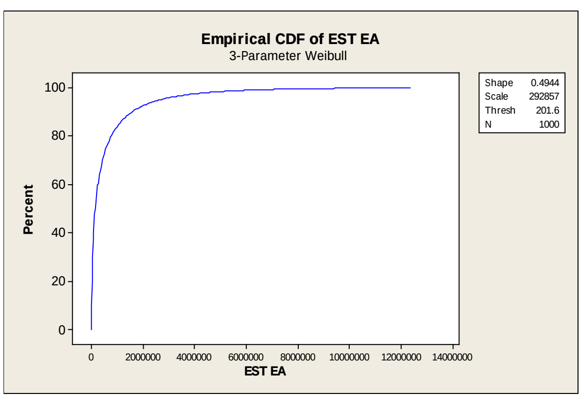
Monte Carlo Eroded Area 3 Parameter Weibull probability plot



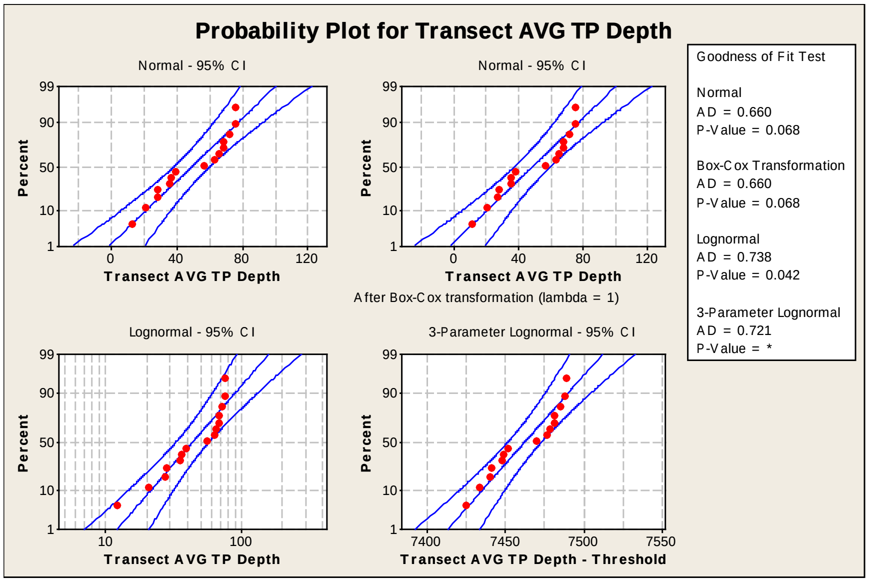
Monte Carlo Eroded Area PDF

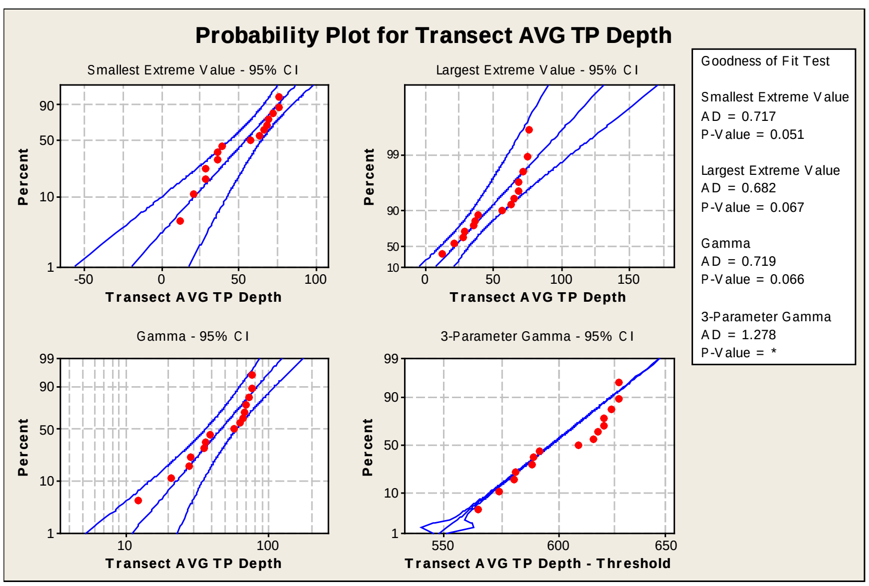


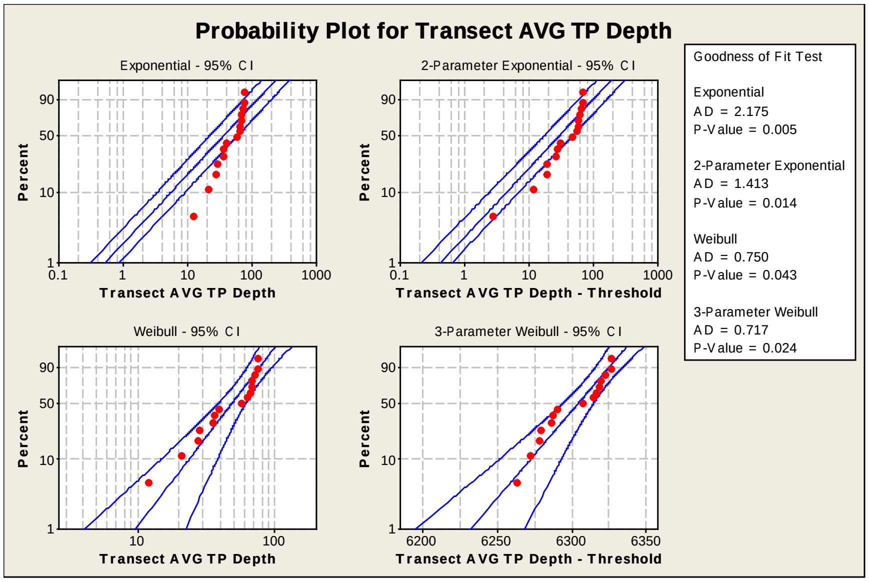
Monte Carlo Eroded Area CDF



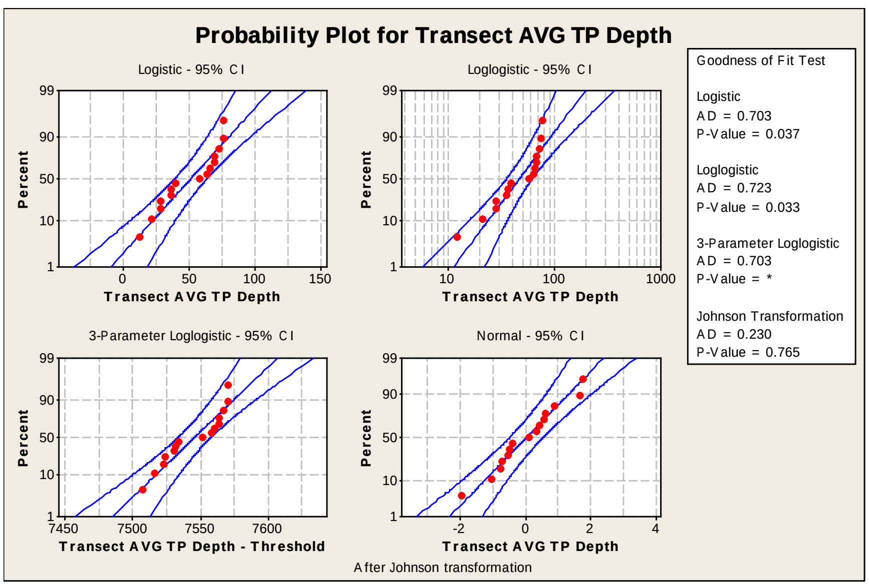
Minitab probability plots for Observed Topsoil Depth





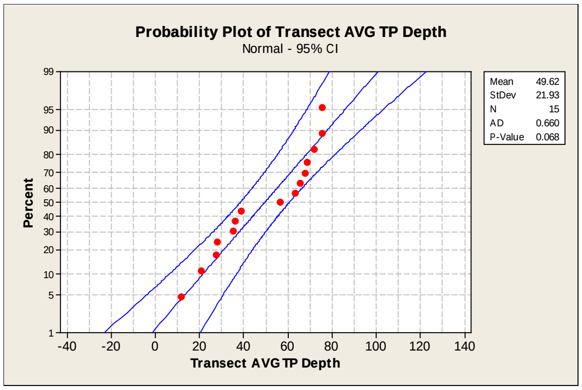


Probability Plot for Transect AVG TP Depth
Exponential - 95% CI
2-Parameter Exponential - 95% CI
Weibull - 95% CI
3-Parameter Weibull - 95% CI
Percent
Transect AVG TP Depth
Transect AVG TP Depth - Threshold
Goodness of Fit Test
Exponential
AD = 2.175
P-Value = 0.005
2-Parameter Exponential
AD = 1.413
P-Value = 0.014
Weibull
AD = 0.750
P-Value = 0.043
3-Parameter Weibull
AD = 0.717
P-Value = 0.024

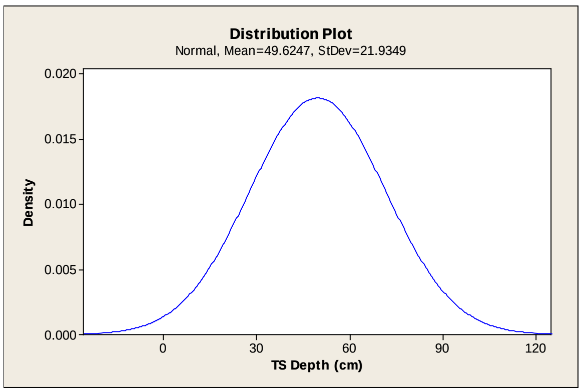


Probability Plot for Transect AVG TP Depth
Logistic - 95% CI
Loglogistic - 95% CI
3-Parameter Loglogistic - 95% CI
Normal - 95% CI
Percent
Transect AVG TP Depth
Transect AVG TP Depth - Threshold
After Johnson transformation
Goodness of Fit Test
Logistic
AD = 0.703
P-Value = 0.037
Loglogistic
AD = 0.723
P-Value = 0.033
3-Parameter Loglogistic
AD = 0.703
P-Value = *
Johnson Transformation
AD = 0.230
P-Value = 0.765

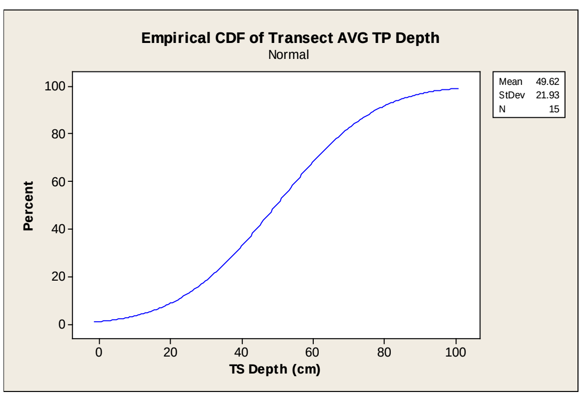
Observed Topsoil Depth Normal probability plot



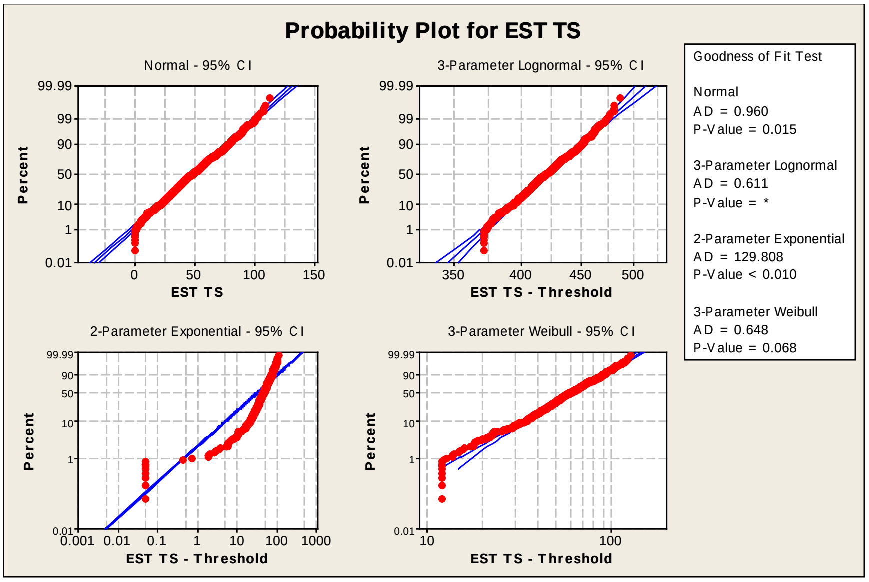
Observed Topsoil Depth PDF

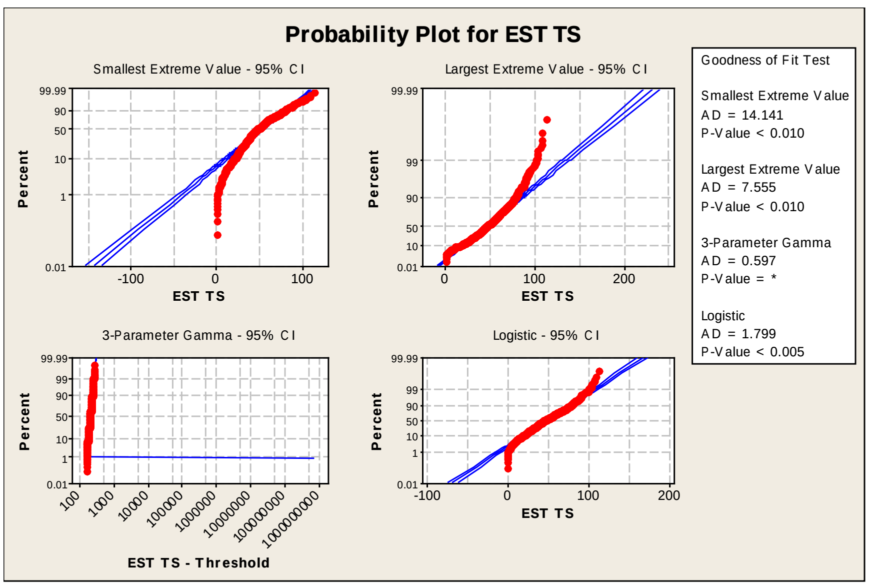


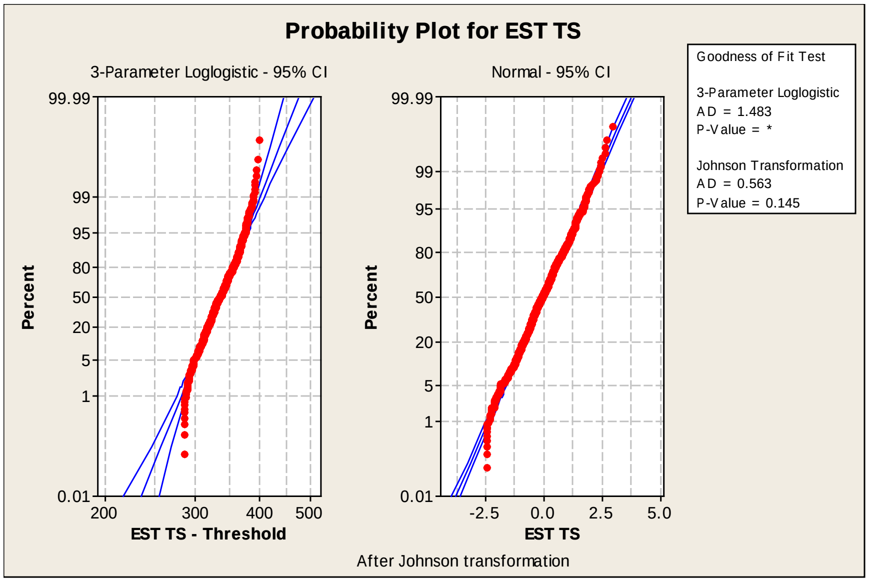
Observed Topsoil Depth CDF



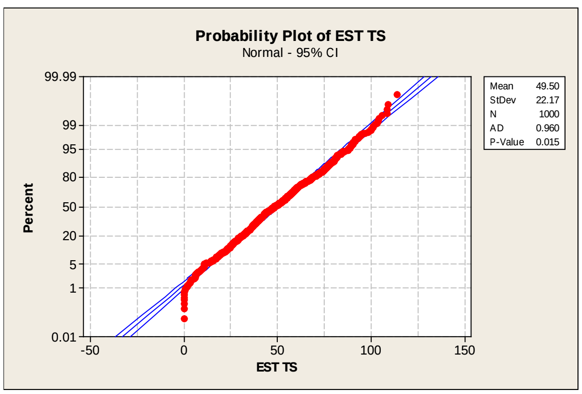
Minitab probability plots for Monte Carlo Topsoil Depth



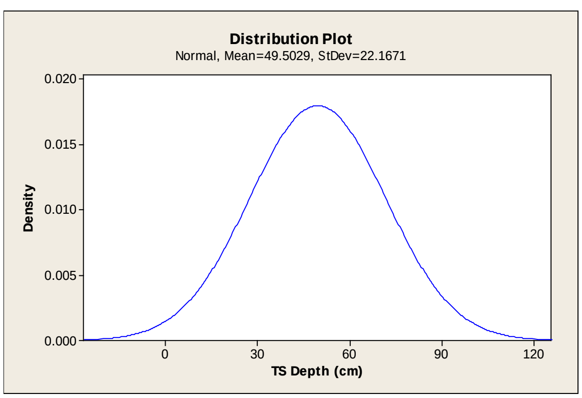




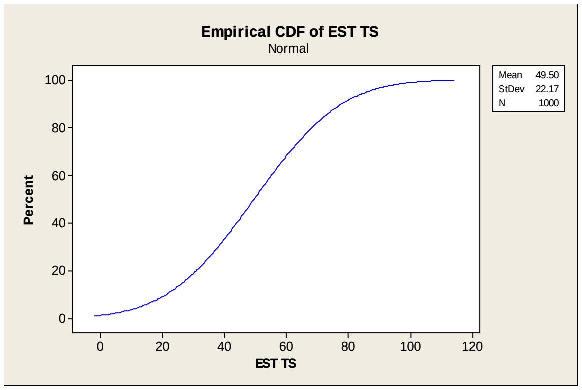
Monte Carlo Topsoil Depth Normal probability plot



Monte Carlo Topsoil PDF



Monte Carlo Topsoil CDF



VITA

Rebecca Anne Purvis

Candidate for the Degree of

Master of Science

Thesis: SEDIMENT AND PHOSPHORUS LOADS FROM STREAMBANK EROSION AND FAILURE: A SOURCE OF LEGACY PHOSPHORUS IN WATERSHEDS

Major Field: Biosystems and Agricultural Engineering

Biographical:

Education:

Completed the requirements for the Master of Science in Biosystems Engineering at Oklahoma State University, Stillwater, Oklahoma in July, 2015.

Completed the requirements for the Bachelor of Science in Biosystems Engineering at Oklahoma State University, Stillwater, Oklahoma in 2014.

Experience:

2014-2015: Graduate Research Associate, Oklahoma State University
2013: Undergraduate Teaching Assistant, Oklahoma State University
2011-2014: Research Assistant, Oklahoma State University
2011-2012: Wentz Undergraduate Research Scholar, Oklahoma State University

Professional Memberships:

American Society of Agricultural and Biological Engineers
American Society of Civil Engineers
Society of Women Engineers