12-2015

Spatial Variation in Fine Sediment and Microbial Transport along Stream Cross Sections: Implications to Modeling and Monitoring

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I am submitting herewith a thesis written by Thomas Michael Walton entitled "Spatial Variation in Fine Sediment and Microbial Transport along Stream Cross Sections: Implications to Modeling and Monitoring." I have examined the final electronic copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Environmental Engineering.

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Spatial Variation in Fine Sediment and Microbial Transport along Stream Cross Sections: Implications to Modeling and Monitoring

A Thesis Presented for the

Master of Science

Degree

The University of Tennessee, Knoxville

Thomas Michael Walton

December 2015

Abstract

Spatial variations in suspended concentrations of fine sediment and indicator bacteria in streams is noted as a source of uncertainty in various applications of water quality datasets. Without proper analysis of the persistence and reasoning for this variance in natural systems, a finer resolution of model calibration is necessary to account for spatial variation in stream cross sections. This level of calibration has become of utmost importance due to technological advancements presenting the ability for water quality modeling frameworks to generate a much finer resolution of outputs. While the importance of model calibration has been noted for remediating levels of uncertainty in output datasets, single point sampling along a stream cross section is still predominately utilized for input data acquisition within the field. In order to test the level of variation which might be accounted for by implementing higher resolution sampling strategies, five evenly distributed positions were sampled simultaneously along stream cross sections. Along with lateral variation, vertical variation was addressed by sampling at 20% and 80% of the respective stage. These sample sets were analyzed for: 1) spatial variation in suspended sediment concentrations, 2) spatial variation in microbial concentrations, and 3) association between the variations of these constituents. Results showed spatial and temporal variations clearly existed within both datasets. Due to the sporadic nature of these variations both within and between events on any given site, it is recommended spatial variation be accounted for by higher resolution input calibration steps rather than purely empirical framework improvements. Spearman correlations showed little evidence of particulate to microbial associations within this study, but it is recommended particle size distributions be evaluated in consideration to attempting correlations between total suspended solids and fecal indicator bacteria in future studies.

Table of Contents

List of Tables

List of Figures

1.0 <u>Review of Literature: In-Stream Pollutant Transport and Associated Uncertainties</u>

1.1 Introduction

Accuracy and precision of measured water quality parameters introduces uncertainty for use in assessments and modeling, presenting the necessity to better understand its role in developing watershed management strategies. Two of the most common uncertainties noted in water resources is the sampling methodology and modeling frameworks used to predict potential sediment and microbial concentrations. Excessive fine sediment and pathogen bacteria are both of great concern in regards to ecological and public health (www3.epa.gov 2015). Considering the association between microbial and sediment concentrations through potential attachment effecting transport and fate of microbial populations, attempts were made to understand uncertainty inherent within both parameters simultaneously. Uncertainty and associations within these datasets were analyzed by evaluating variations in concentrations along a stream cross section. This is important due to the fact that technological advancements have led to improvements in computer processing, allowing for water resource modeling applications to predict large skews of variables on very finite scales in all dimensions of a given reach. In accordance with updated modeling resolution, sampling methodologies must be updated to properly assess constituents' lateral variance in the field for proper calibration of fine resolution modeling applications.

1.1.1 Restoration and Management

The EPA lists pathogens and sediment within the top five constituents of concern among streams on state 303(d) lists, in which these compromised water bodies require watershed management and Total Maximum Daily Load (TMDL) targets (www3.epa.gov 2015). Total Maximum Daily Loads (TMDL) were implemented as part of the Clean Water Act in 1977 to help minimize pollution to acceptable levels in surface water. These regulations are based in methodologies which were not designed to encompass the needs of increased technological processing power that generate finer scale outputs, in turn requiring finer scale inputs for calibration (Karr & Yoder 2004, Yager 2007). The older methodology has been adopted by modeling efforts utilized for conceptualizing stream processes on a much finer scale than it was ever intended to analyze. The culmination of this error leads to ineffective management practices due to inappropriate regulatory standards with improper implementation of inadequately understood management designs.

Pathogens have been listed as the number one constituent causing impairment within United States streams and rivers, being a cause of impairment for 10,681 of the 43,180 impaired waters listed (www3.epa.gov 2015). Pathogens are of obvious public concern, especially in consideration to human exposure. While this constituent is of highest concern, the methods utilized for tracking potential contamination have been heavily scrutinized (Characklis et al 2005, Karr & Yoder 2004, Yen 2002). Error associated with analytical methods, along with modeling frameworks which lack to fully account for constituents interactions with their surroundings, contribute to this scrutiny. Indicator bacteria (IB), which are utilized in pathogen tracking efforts due to their more cost effective analytical methodologies, are typically modeled as free floating particles (Jamieson et al 2005, Wilkinson et al 1995). Associations between microbes and particulate matter have been noted through partitioning experiments (Auer & Niehaus 1993, Characklis et al 2005, Krometis et al 2007). This association leads to transport mechanisms which rely on much larger masses than that of the microbes, leading to a source of uncertainty in the form of miscalibrated modeling frameworks.

Fine sediment was fifth in constituents of concern, being listed as the cause of impairment for 6,466 of the 43,180 impaired waters nationally (www3.epa.gov 2015). Sedimentation has been noted as one of the largest pollutant sources to address through watershed management efforts, due to both its ecological and socioeconomical effects (Apitz 2012, USEPA 2006). Excess suspended sediments have been shown to jeopardize ecological integrity, greatly impairing water quality (Schwartz et al 2011). Suspended sediment as particulate matter has also been associated with fate and transport of both microbial communities and other pollutants of concern, making it a common surrogate for overall water quality in a stream. A large amount of the uncertainty associated with random sampling is due to misjudgment of transport potential. This misconception has been primarily contributed to improper calibration of models in consideration to discontinuous bed form roughness.

1.1.2 Uncertainty in Surface Water Quality

Uncertainty is the level of variance noted between calculated and measured datasets which cannot be directly contributed to a known source of error with predefined levels of confidence. The field of water resources engineering is filled with varying levels of uncertainty (Harmel et al 2006, McCarthy et al 2008). In most uncertainty studies within the genre of water quality, error is categorized as sampling, analytical, and empirical (Harmel et al 2006, Harmel et al 2010, Harmel et al 2007, Harmel et al 2009, McCarthy 2008). Essentially, error from the methodology used to collect a sample, error based on the method utilized in laboratory analysis, and error associated with mathematical modeling efforts, respectively. While research has been performed on various constituents of these categories, proper culmination of error has only recently been assessed but not yet applied within the field (Harmel et al 2006, Harmel et al. in press). This is of particular interest when considering the relative error associated with modeling efforts. One must not only consider the error inherent within the model structure itself, but also that represented within the data (Silberstein 2006, Harmel et al. in press). Although more research has been performed to quantify and understand uncertainty in sediment concentrations in surface waters, uncertainties regarding microbial data are only now being explored (Harmel et al 2006). Further, beyond simply quantifying these uncertainties, it is critical to understand why these uncertainties exist in an effort to match fundamental processes to observed data trends.

1.2 Fundamental Pollutant Transport Processes

1.2.1 Suspended Sediment

Stream geomorphology influences flow regimes, sediment transport, channel boundary characteristics, and water quality to produce, maintain, and renew lotic habitats on a spatial and temporal scale (Cluer & Thorne 2014). For error associated with sediment load estimations, the sampling (data acquisition) category has been widely noted as the prevalent form of error (Harmel et al 2010, Harmel et al 2009). One explanation for this, which has been of great interest for sampling uncertainty associated with suspended sediments, is variation in concentrations due to the local effect of roughness elements on the velocity profile (May 2007, Yager et al 2007). Roughness elements are present in two forms during sediment entrainment: immobile and mobile grains (Ghilardi et al 2014). Perhaps the more studied of the two, immobile grains are defined as macroroughness elements which dissipate the relative available energy for entrainment (May 2007, Yen 2002, Yager et al 2007). These increased areas of roughness further dissipate available shear stress, which increases settling while simultaneously decreasing energy available for entrainment. This produces dynamic sediment profiles varying both laterally across and longitudinally along the channel. It has even been noted that grain sorting, interchanging of grain size distribution within the water column due to dissipation of available energy upon entrainment of heavier grains, greatly alters flow velocity and bed morphology (Frey et al 2003, Ghilardi et al 2014, Recking et al 2009). Grain sorting results in pulses of fine sediment followed by bursts of larger coarse grains (Frey et al 2003, Recking et al 2009). Studies have even shown natural rivers experience large variation in sediment load for similar discharges (Recking et al 2009, Turowski et al 2009). Therefore, given ample sediment supply and capacity, roughness factors will control fate and form of geomorphic structures (Buffington & Montgomery 1997).

Being that both headwaters (May 2007, Hassan 2005) and rivers (Recking et al 2009, Turowski et al 2009) have shown evidence of flow variations due to roughness factors along the cross section, it would seem logical that the streams which lie between these two scales would experience similar effects. Bedform resistance has been shown to contribute up to 75% of total channel roughness, in turn defining the effective shear stress (Buffington & Montgomery 1997). Effective shear stress is the empirical variable utilized to represent energy loss due to a stream bed and its banks (Buffington 1997). This variable is typically evaluated in relation to velocity, depth, and a relative roughness value averaged for the reach (Yen 2002). While this method presents a good foundation for evaluation of the energy dissipated by the wetted perimeter, it does not adequately evaluate the system's complex dynamics in consideration to implementation and evaluation of management efforts (Byrd 2000). Velocity profiles vary incrementally with relative roughness, which is not encapsulated by an averaged value (Byrd et al 2000). The relative effectiveness of macroroughness elements in resisting flow is dependent upon its size and density (Papanicolaou et al 2001). Defining the applicable ratio for potential flow variation by roughness element size would allow for better field evaluation of possible sampling sites, which in turn would be utilized in calibrating the model to reach specific characteristics.

1.2.2 Pathogen and Fecal Indicator Bacteria

It has been noted in previous studies that a large amount of error exists within microbial transport modeling (McCarthy et al 2008, Quilliam et al 2011). This is due to both misguided perceptions inherent within respective modeling frameworks (Jamieson et al 2004, Quilliam et al 2011, Wilkinson et al 1995) and dismissal of data uncertainty (Harmel et al in press). Microbial transport is typically modeled as free floating particles (Jamieson et al 2004, Wilkinson et al 1995), despite evidence that certain microbial species associated with particulate matter in regards to entrainment. Research dating back to 1985 shows this association and relates it to two transport methods (Marshall 1985). The first method of connection to particles by microbes is generally through a weak Van der Waahls bond, due to the varying charges between microbes and the ions associated with certain sediment types (Howell et al 1996). Once these microbes have been drawn to the particulate matter, certain species proceed to attach themselves to particles via extracellular polymers (Marshall 1985). Various research studies have shown this bond to be "indefinite" in behavior, thus altering the respective entrainment regime of various microbial species to that of the larger particles they're attached to (Jamieson 2005).

1.2.3 Importance of Linking Evaluation of Constituents

Research has pointed to an increased level of effectiveness in the implementation of TMDL's which utilize a culmination of biological, chemical, and physical characteristics (Apitz 2012, Harmel 2010, Karr & Yoder 2004). To properly deal with these constituents, a hydromorphological viewpoint is crucial (Apitz 2012, Vaughan et al 2009). Hydromorphological views are based within three fields; hydrology, geomorphology, and ecology (Vaughan et al 2009). Without considering all three viewpoints, restoration and management efforts will suffer. Due to the intricacies inherent within a quasi-equilibrium based system, a large range of variables must be considered simultaneously. Without doing so, the alteration generated by opposing constituents will negatively affect model calibrations. With the prevalence of uncertainty within the field of water quality data, it is advantageous to eliminate as much error in as many measurements as possible. The interactions between constituents are also the least parameterized aspect associated with current modeling efforts.

Links between ecology and physical habitat are of great interest in water quality research and management (Vaughan et al 2009). Functional traits and suspended sediment transport metrics have been shown to correlate both negatively and positively at varying frequencies (Schwartz et al 2011). It has been shown in areas with unpredictable flow magnitudes and frequency, abiotic factors control in lieu of biotic trophic factors such as predation (Montgomery 1999, Schwartz et al 2011, Poff & Ward 1989). Species abundance and community structure have been shown to rely on geomorphic processes in four dimensions: longitudinal, lateral, vertical, and temporal (Montgomery 1999, Poff & Ward 1989). While current modeling efforts account for all these dimensions, the lateral alteration in roughness along the cross section is measured on far too broad of a scale for proper calibration of reach specific characteristics within modeling applications (Jamieson et al 2005, May 2007, Nitsche et al 2011, Yager et al 2007). Various species depending on sediment loads in altering ways, makes proper implementation of BMP's for sediment control very difficult. With the current lack of resolution within the modeling frameworks utilized for designing these practices, predictably positive outcomes are arguably impossible.

1.3 Implications

1.3.1 Monitoring

Current automated sampling efforts implement a vertically integrated intake at a singular point within the stream (Harmel et al 2010), and grab samples are typically taken at a single point perceived as center of flow (Harmel & King 2005, Martin et al 1992). While many constituents were mostly assumed thoroughly mixed, making these sampling methodologies valid, more recent studies have stated otherwise (Harmel et al 2010, Harmel & King 2005, Papanicolaou et al 2001, Quilliam et al 2011). It has been established within the field that sediment and microbial transport vary by both vertical and lateral spatial domains (Martin et al 1992, Rode & Suhr 2007, Cluer & Thorne 2014). The USGS even implemented Equal-Width-Increment (EWI) and Equal-Discharge-Increment (EDI) methods for sampling purposes, which have been widely accepted for stormwater sampling (Harmel et al 2006). Unfortunately, cost and time restrictions have limited the utility of these constructs (Harmel et al 2010). Some research even showed variation within dissolved constituents, which are still widely accepted as well mixed within all streams (Harmel et al 2010). In association with variations in shear stress, available degradable material for entrainment varies along the cross section as well (Frey 2003). The culmination of discontinuity between parameters generates numerable facets to uncertainty presenting themselves in undefined variance within the sampled dataset which effects viability of modeled output. Error associated with cross sectional profile variation has been noted from 20% up to an order of magnitude within the literature (Yager et al 2007). Multiple empirical forms have been shown successful in limiting error by assessing variance in roughness along the cross sectional profile, but none of these forms have been widely accepted or utilized within the field (Nitsche et al 2011).

The River Continuum Concept proved that geomorphic and ecologic processes are linked linearly along the flow path for a given region with uniform geology, topography, and climate; however, process domains proved that, given a finer scale resolution, a more diverse set of hydrogeomorphic conditions could be analyzed with the principles of this method (Rosgen 1994, Montgomery 1999). While this allows for assessment of what would be ecologically pristine water standards in regards to geomorphic practices in areas where no pristine examples have persisted, it does not decrease calibration error due to sampling error and outdated empirical frameworks which will eventually mislead BMP implementation. This shows lack of differentiating between assessment and design based modeling applications could potentially mislead restoration efforts by performing system analysis on an inappropriate scale for modeler's intended use. To note, Rosgen associates cross sectional area with varying flow regimes due to slope and material type (Rosgen 1994). While research has spoken in favor of these views since, sampling efforts still neglect collecting at enough points to generate the resolution necessary for accurately modeling geomorphic processes.

1.3.2 Modeling

The US National Research Council has identified model accuracy improvement, especially in the area of bacterial estimates, essential in development of water quality treatment innovations (Krometis et al 2007). Jamieson (2004) claims a complete watershed scale microbial water quality model must include the following: i) characterize production and distribution of waste and associated microorganisms, ii) simulate transport of microorganisms from land to streams, iii) route microorganisms through stream networks. While the microbial to sediment association has been validated in numerous works, most modeling applications still treat microbial transport in streams as that of a free floating particle (Jamieson et al 2004, Wilkinson et al 1995). If microbes are truly indefinitely associated with particulate, then it is the particulates much larger mass which must be considered for entrainment (Davies 2000, Jamieson et al 2004). It has also been shown that microbial communities are not only stored within benthic sediments, but also grow within these systems (Characklis et al 2005, Howell et al 1996, Sherer et al 1992). This implies a background concentration which must be taken in to account, which has been shown in some areas to exceed non-point source input (Sherer et al 1992). Microbial association with sediment particles attributed as one of the last remaining complications in designing a reliable microbial population prediction model (Jamieson et al 2005, Tian et al 2002).

With the human and ecological implications of geomorphic variation on ecosystems, proper modeling is required to appropriately implement management practices (Harmel et al 2007, Karr & Yoder 2004). Entrainment modeling efforts have been shown to over predict sediment loads due to lack of accounting for variation in bed roughness elements (Byrd et al 2000, Tian et al 2002, Yager et al 2007). Cross sectional discontinuity has even been evaluated as the predominant source of uncertainty in regards to sediment load water quality data (Harmel et al 2009). For this to truly be implemented, better sampling must be performed within the field to properly assess the true condition which is being modeled (Silberstein 2006). Along with a better evaluation of the system being modeled, a more diverse modeling framework would allow for implementation throughout varying hydromorphic settings (Montgomery 1999). Current modeling efforts have all been implemented to express very specific hydromorphic regimes, which is the reason no single method has been widely adopted within the field (Nitsche et al 2011).

Uncertainties present themselves both in the evaluation of what management to implement, and in understanding management efficiency (Harmel et al 2009, Silberstein 2006). Even when the correct methods are implemented, inadequate resolution within the methodologies used to assess these systems can lend to inadequacy in efficiency of evaluations for the systems used (Karr & Yoder 2004). Once research congruently agrees upon updated viewpoints, it becomes irresponsible to misuse modeling applications which were designed on a much broader spectrum for assessment purposes. Design based modeling systems require a much more refined view to adequately address the issues inherent within infinitely dynamic natural systems. To calibrate a model in consideration to the delicate quasi equilibrium which maintains sustainable lotic communities, proper implementation and evaluation is necessary. In consideration to TMDL, more streams are listed than are cleared from 303d listing (Karr & Yoder 2004). Karr and Yoder contribute this to improper evaluation of management practices leading to management which miss the mark in consideration to mitigating TMDL constituents.

1.3.3 Ecological

The Clean Water Act section 502(19) defines pollution as human-induced alteration of waters caused by pollutants as well as non-pollutant agents, such as flow alteration, loss of riparian zone, physical habitat alteration, and introduction of alien taxa (Karr & Yoder 2004). Relative levels of sedimentation are essential for physical and trophic structures in lotic ecosystems (Apitz 2012, Schwartz et al 2011). However, many urban and agricultural activities result in heightened levels of sediment. This increased sediment load has been linked to impaired waterways, and is listed as one of the most detrimental pollutants to lotic ecosystems (Apitz 2012, USEPA 2006). Even in urbanized areas where sediment loadings from impervious surfaces are not high, increased runoff volumes illicit high flows which can deplete available sediment stores. Not only is either the abundance or lack of sediment adverse to the system, but other pollutant sources are often transported and stored within sediments (Apitz 2012). Sediment is therefore, arguably, the most detrimental non-point source pollutant source within surface waters.

1.4 Summary

Due to the importance of both sediment and indicator bacteria to environmental policy and public health, the level of relative uncertainty involved in evaluation of these parameters is unacceptable (Apitz 2012). For sediment, a more accurate sampling methodology should be implemented to properly calibrate modeling on a fine enough scale for designing sustainable restoration and management efforts (Harmel et al 2010, May 2007, Nitsche et al 2011, Papanicolaou et al 2001, Silberstein 2006, Yager et al 2007). For microbial evaluation, further understanding in regards to the level of association between particles and microbes must be assessed in natural systems. While partitioning has been performed to evaluate the particle size and type most associated with a given microbe (Auer 1993, Davies & Bavor 2000, Jamieson et al 2005, Wilkinson et al 1995), the effects of high energy flows and roughness elements on partitioning needs further evaluation. Current partitioning evaluations have been largely performed with dosed samples in a laboratory. While these experiments provide datasets which are necessary in establishing the fundamentals of this relationship, they are not absolutely representative of natural systems. Once the level to which microbial communities associate with particulate matter in transport is properly assessed, it will be possible to empirically represent the ratio of relative masses in correspondence to relative entrainment.

Considering the potential uncertainty of these parameters has been predominately evaluated in laboratory conditions, efforts need to be made towards evaluating their relative existence within natural systems. While the dynamic processes within natural systems has widely been noted, little effort has been put forth toward evaluating methods of analyzing this variation within sampling methodologies. For proper model calibration to occur, sampling frequency must be evaluated in relation to both time and space. To evaluate potential benefits to scale based on spatial frequency, five points were simultaneously sampled laterally along a cross section. Samples were duplicated for each storm to evaluate the alterations due to sediment pulse sheets. Once concentrations were derived, the structures of graphs of concentration vs lateral position were evaluated for correlations between TSS and IB concentrations. In conclusion, the issues addressed are as follows: 1) lateral variations in concentrations of both TSS and IB, 2) associations between particulate matter and IB, and 3) consistency of this variation and association between duplicate sampling sets.

2.0 Suspended Sediment and Fecal Indicator Bacteria Spatial Covariance

2.1 Abstract

Lack of accounting for spatial covariance in microbial and suspended sediment concentration have been contributed to generation of uncertainty with use of measured data for load estimations and model calibration. While current sampling methodologies focus on single point sampling strategies, updated modeling frameworks which utilize improved technological processing capabilities generate output on a much finer resolution than what these sampling strategies were intended. To analyze the potential for suspended sediment concentration and microbial density variation to be an issue in water quality data the following questions were evaluated: 1) presence of lateral variance in microbial densities and suspended sediment concentrations within natural systems, 2) potential persistence of microbial to particulate associations during transport within the water column, and 3) whether this variance is distributed in a systematic manner which could potentially be contributed to cross sectional characteristics. By implementing a finer resolution sampling methodology on streams with varying levels of roughness, it is hypothesized that further insights toward the contributions of these relationships can be derived by graphical and statistical analysis. Some of the uncertainty generated within modeling outputs has been contributed to lack of accounting for lateral variation in roughness elements on a fine enough scale, and while positive results have been noted in utilization of models which account for this error none have completely eliminated the uncertainty within the dataset (Nitsche et al 2011). There was clear evidence of lateral variation within the samples gathered, with relative standard deviations over 40% for the majority of sample sets. The variance noted was not systematic, showing different characteristics between duplicate samples, various events, and altering sites. Another noted contributor to uncertainty is the effects on microbial concentration's fate and transport due to particulate

association (Characklis et al 2005, Jamieson et al 2005, Yager et al 2007). While these associations were clearly noted in the literature (Characklis et al 2005, Jamieson et al 2005, Yager et al 2007), the variability in Spearman correlation statistics within this study would lead to a different view. It is speculated lack of consistent correlation in these sample sets is due to the fact that TSS can easily be more heavily weighted toward larger grains which microbial populations do not associate with. Future correlation assessments in the field are recommended to incorporate particle size distribution analysis.

2.2 Introduction

Fecal coliform and *E.coli*, two of many indicator bacteria utilized in tracking potential pathogen presence in surface waters, are of great concern to public health (Auer & Niehaus 1993, USEPA 2002). Pathogens have been listed as the number one constituent causing impairment within United States streams and rivers, being a cause of impairment for 10,681 of the 43,180 impaired waters listed (www3.epa.gov 2015). The presence of waterborne pathogens has been associated with disease outbreaks following storm events due to both re-entrainment of bed material and non-point source pollution (Curriero et al 2001, Jamieson et al 2004, Wilkinson et al 1995). Studies have shown a single day of storm sediment and microbial loads can be equivalent to several years of dry weather loads (Krometis et al 2007). In addition to the temporal variation between baseflow and flood stages, spatial variation in fecal indicator numbers per volume have been noted in streams (Jamieson et al 2005, Krometis et al 2007) and lakes (Auer & Niehaus 1993, Gannon et al 1983). These spatial variations have been accredited to associations between sediment and microbes, altering flow regimes, and natural microbial stores (Characklis et al 2005).

Along with effecting fate and transport of microbial communities, sedimentation plays a large role in the disturbance of lotic ecosystems itself. Sedimentation is attributed to the disturbance of 6,466 of the 43,180 impaired waterways nationally (www3.epa.gov 2015). Suspended sediment models that predict suspended sediment loads, e.g., Einstein method and van Rijn, have been based on the concept of vertical concentration profiles containing smallest concentrations near the water surface and the greatest concentrations near the channel bed as defined by the Rouse number (Rouse 1965, Sturm 2009). While Rouse (1965) laid the framework for evaluating geomorphic conditions of a reach, it was never meant to be an all-encompassing method by which all systems were evaluated (Yen 2002). Rouse set forth the foundations for evaluating sediment entrainment in open channels in the most general case, considering channels which were assumed wide and shallow, to allow broadest applicability of the initial analysis of these parameters (Yen 2002). While this assumption is valid for rivers in which ratio of bed far exceeds that of the banks along the wetted perimeter, turbulence from both bank and bed influence velocity patterns along the cross-section of a stream and may not create a logarithmic vertical sediment profile. Recent studies have also highlighted the importance of evaluating variation in roughness regimes in consideration to alteration in sediment entrainment laterally along a given cross section due to large immobile grains impeding flow (Nitsche et al 2011, Papanicolaou et al 2001, Yager et al 2007). Macroroughness elements, immobile grains which perturb flow, have various effects dependent upon density, size, and roughness (Papanicolaou et al 2001, Yager et al 2007). These effects have been shown in both headwaters and larger open channels such as rivers (Hassan et al 2005, May 2007, Yager et al 2007, Recking et al 2009, Turowski et al 2009).

Research dating back to Marshall (1985) begins to inform our understanding of microbial associations with particles. Since, sediment association has been widely noted as a potential mechanism by which microbes fate is altered (Auer & Niehaus 1993, Characklis et al 2005, Gannon et al 1983, Jeng et al 2005, Krometis et al 2007). While modern research points to confounding factors in microbial transport, such as persistence of microbial communities within benthic soil groups and association of microbes to particulate matter, these factors are typically ignored within modeling applications (Nitsche et al 2011). While this natural persistence has been noted for various species of microbes, which microbes will survive naturally in the environment depends greatly upon soil type and particle size distribution (Jeng et al 2005). Most modern modeling applications assume all constituents to be evenly mixed, which has actually only been shown applicable to dissolved constituents (Harmel and King 2005, Martin et al 1992). Many microbial modeling applications also assume microbes to be free floating particles, which has been shown to not always be the case (Jamieson et al 2005). In consideration to the groundwater study, the only study which showed 100% association between microbes and particulate matter (Mahler 2000), it would seem that the only defining factor in whether microbes should be modeled in consideration to particulate mass rather than their own mass is potential exposure to particulates. With the level to which microbe to particulate associations has been shown to occur, neglecting this interaction within model calibration will continuously implement empirical based error on to the output dataset. Stream properties effects on microbial fate and transport should be more heavily analyzed as well (Jamieson et al 2005). The effects of cross sectional roughness discontinuity on flow regimes have been established, but must also be considered for in-stream sediment and microbial fate and transport (Quilliam et al 2011).

While the literature noted strong evidence of spatial variance for many constituents of interest, little effort has been made to evaluate the presence, persistence, and distribution of these spatial patterns within the field. Sediment profiles have been noted to vary laterally due to alteration of roughness elements. Spatial variation has also been noted for microbial communities due to alteration in roughness elements and availability of natural stores based upon bed sediment regimes. Variation in fate and transport have also been noted for microbial communities due to sediment to microbial association. Due to these varying concentrations, the error associated with current single point sampling methodologies must be understood. While these methods still hold valid utility in consideration to analyzing broad scale water quality characteristics, evaluation of the potential error they could generate in consideration to sampling efforts for TMDL development and Stormwater Control Measure (SCM) effectiveness studies presents itself as inherent uncertainty within implementation of management design efforts as well as improper regulatory decisions. For initial evaluation, an arbitrary number of 5 sections were utilized in evaluating the presence of these elements. The variation of these elements were analyzed to: 1) evaluate presence of lateral variance in microbial densities and suspended sediment concentrations within natural systems, 2) examine potential persistence of microbial to particulate associations during transport within the water column, and 3) determine whether this variance is distributed in a systematic manner which could potentially be contributed to cross sectional characteristics which could be accounted for within empirical frameworks to negate the necessity of utilizing more rigorous sampling strategies as a means of calibrating models to the level of variance present within a given reach.

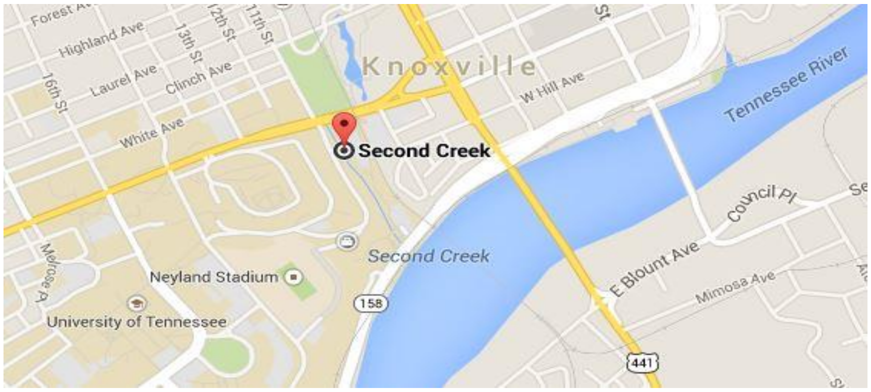
2.3 Methods

2.3.1 Site Descriptions

Sampling was performed on 4 separate runoff events, at the three sites described in Table 1 below. Locations of these streams were in East Tennessee and can maps can be seen in Figure 1. All sampling was performed while streams were elevated, with clear signs of entrainment presenting as turbidity. Duplicate samples were taken at all events except Beaver Creek due to a quicker response than could be captured for the targeted event. Duplicate events were sampled at Second Creek, due to the low microbial densities on other sites presenting questionable results. Due to the lower stage present at the banks of Second Creek, a single depth of 60% stage was sampled at these respective points along the cross section. While this negates the ability to assess the stage relative variance at these points, it still allows for the variance between the two banks themselves to be evaluated.

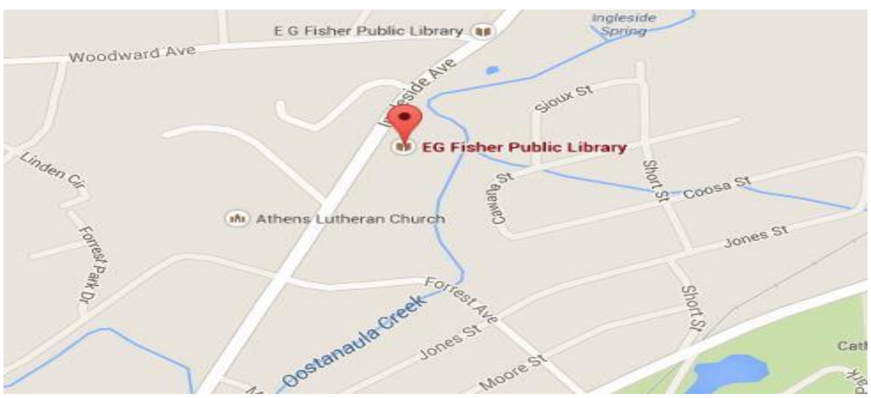
Table 1: Site Description

Site	Soil Type	Bed Description	Bank Description
Second Creek	Silty Clay	Mix of Clay and Rip rap stone	Slightly sloping mix of clay and stone
Beaver Creek	Silty Clay	Clay only	Highly incised with predominant root wad structure
Oostanaula Creek	Silty Clay	Clay with some woody debris	Highly incised with smooth soil and root wad mixture



(a)

(b)



(c)

Figure 1: Site locations: (a) Second Creek, (b) Beaver Creek, (c) Oostanaula Creek

2.3.2 Cross Sectional Monitoring Apparatus

A metal strut was run across the top of the channel for connecting ¾" electrical metallic tubing (EMT) with ¾" strut straps. A 3/8" section of rebar was lowered down to the bottom of the channel supported by the ¾" EMT. At these five points, depths equal to 80% and 20% of the measured stage were taken (Byrd et al 2000). Zip ties attached the 3/8" sample tubing for setting sampling points at respective depths. Tubing from the 10 points then connected to individual 500 mL vacuum safe Nalgene bottles. These bottles then connected to the side of a singular 1000 ml vacuum safe Nalgene bottle. A single vacuum pump was collected to the larger bottle to facilitate sample collection. Essentially, the pump generates pressure within the larger bottle, spreading the pressure to the smaller bottles and finally tubing, collecting a sample from the stream. To account for the variation in pressures generated by altering tubing lengths, control valves were placed on the lines leading from the cross section to sampling bottles. An example of this setup in the field can be found in Figure 2.

Figure 2: Sampling Apparatus in Field

To sample the variation in sediment concentrations due to discontinuous roughness factors along a cross section, five, evenly distributed points were sampled simultaneously across three streams in eastern Tennessee (Figure 1). The five points were place at 20% intervals of width. Sampling ports were set at 20% and 80% of stage. The first site at Second Creek was performed with a width of 26’ and had the following stages from left to right: 1) 1.00’ 2) 1.85’ 3) 1.85’ 4) 1.65’ 5) 1.25’. The second site sampled at Second Creek was performed on a cross section of width 24.5’ and had the following stages from left to right: 1) 0.70’ 2) 1.50’ 3) 1.50’ 4) 1.35’ 5) 0.93’. The site sampled at Beaver Creek had a width of 15’ with stages from left to right of: 1) 2.75’ 2) 2.70’ 3) 2.13’ 4) 1.90’ 5) 1.90’. The site sampled along Oostanaula Creek had a width of 30’ with stages from left to right of: 1) 1.75’ 2) 2.92’ 3) 3.58’ 4) 2.25’ 5) 2.25’.

2.3.3 Flow Rate Measurement

For Second Creek, flows were measured using a gauging station from another project. The first event was collected with an ISCO 350 ADV. Unfortunately equipment malfunction generated issues with gathering flow data on the second event sampled at Second Creek.

For Beaver Creek and Oostanaula Creek the Marsh Mcbirney FloMate 2000™ was first used to calculate velocities. The device was calibrated using the standard 5 gallon bucket of still water to evaluate a zero velocity point. Increments of one foot were used on during velocity measurements along the cross section, and duplicates were taken to mitigate the error associated with the device itself. After gathering the velocity profile along the cross section, the velocity area method was utilized to calculate flow rates.

2.3.4 Total Suspended Solids

Total Suspended Solids (TSS) was measured in accordance with ASTM D5907. This methodology involves filtration of the samples to evaluate relative non-filterable material defined as TSS. Standard 1.5 μm filters were weighed within predesignated drying trays. Filters were then placed on filtration bells over Erlenmeyer flasks connected to a vacuum system. Sampling bells which can measure 250 ml volumetrically were then clamped on top of the filtration apparatus. The sample volumes were then measured with graduated cylinders before pouring them in to the sampling bell. While 100 ml are typically used in these analyses, the entirety of the sample in the 250 ml bottle was used due to low concentrations. Following filtration, the samples were dried at 105 ^{O}F for one hour. The samples were then reweighed, with the sediment load being derived by subtracting the initial weight and dividing by the volume used.

2.3.5 Idexx Colilert

Microbial concentrations were quantified through the Idexx Colilert analytical methodology (Yakub 2002). Two dilutions, 0.01 and 0.001, were used to account for potential concentration fluctuations affecting the viability of the methodology. The samples were incubated in a 37 ^{O}C oven for 4 hours, followed by a 44 ^{O}C oven for 20 hours (Yakub, 2002). At the end of the 24 hour period, counts were taken for both fecal coliform and *E.coli* and statistically converted to MPN values.

2.3.6 Statistical Analysis

In order to assess potential spatial variation along the cross section, several statistical evaluations were used to quantify the differences in microbial and suspended sediment concentrations. First means and standard deviations were calculated for each sampling sets concentrations using Microsoft Excel. Following this Relative Standard Deviation (RSD) was derived by dividing standard deviations by their respective means to normalize the average variance associated with each sampling set. Then residuals were calculated to ascertain whether any given sampling position along the cross section experienced less variance and tighter clustering in regards to the mean than for all other sampling sets. Finally multivariate correlation analysis was performed using JMP Pro, with Spearman correlations applied to analyze the correlation between suspended sediment and indicator bacteria within the given non-normally distributed dataset.

2.4 Results

2.4.1 Fine Suspended Sediment

A summary of the means and standard deviations for TSS can be found in Table 2. Residuals between -24 ppm and 27 ppm (Figure 3) were summarized for the ranges of TSS between 1 ppm and 48 ppm (Table 6). The triangles seen at point three along the x axis is in reference to the preferred grab sample position for single point sampling, and represents some of the lowest precision noted within these sampling sets for those collected at 80% stage. For these sites specifically, it appears that the left side of the channel represents the sites of greatest precision and accuracy in consideration to 80% stage sampling. Percentage of vertical position which had the greatest concentration were summarized to validate concerns with utilizing Rouse sediment profiles within streams (Figure 4). While the bottom should have had the greatest concentration, this was only true on 31% of the sampling points. The top which should have the smallest concentration, showed the greatest concentration in the majority, 46%, of the cases. The remaining cases, 23% of the sampling positions, showed equivalent concentrations.

Table 2: Total Suspended Solids Summary Table

Site	Event	Sample	Average TSS (ppm)	Standard Deviation TSS (ppm)	RSD (σ/μ)	Flow Rate (cfs)
Second Creek	1	a	34	8	24	61
	1	b	27	13	48	
	2	a	40	4	11	NA
	2	b	33	7	20	
Beaver Creek	1	a	15	6	41	18
Oostanaula Creek	1	a	13	5	40	56
	1	b	17	14	87	

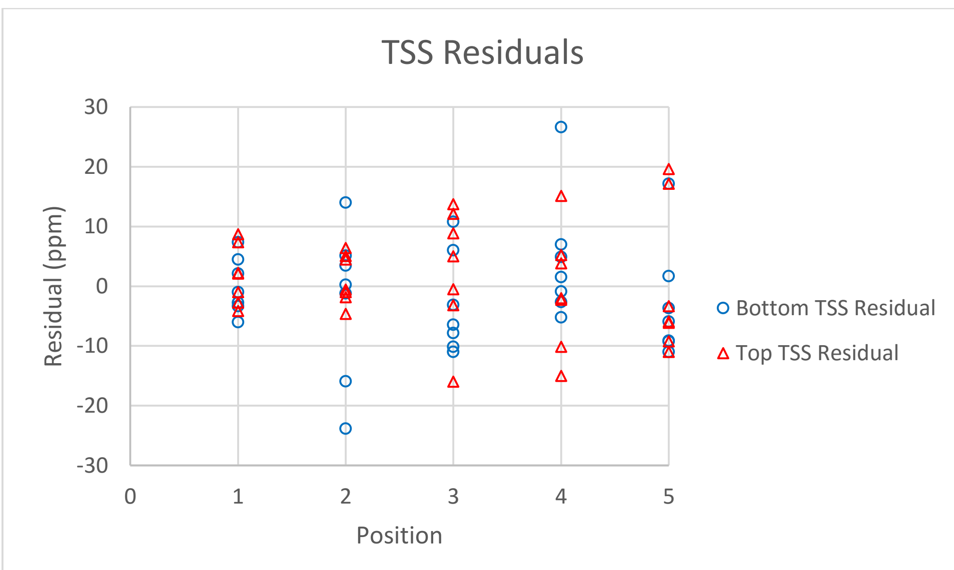


Figure 3: Suspended Sediment Residuals from TSS samples

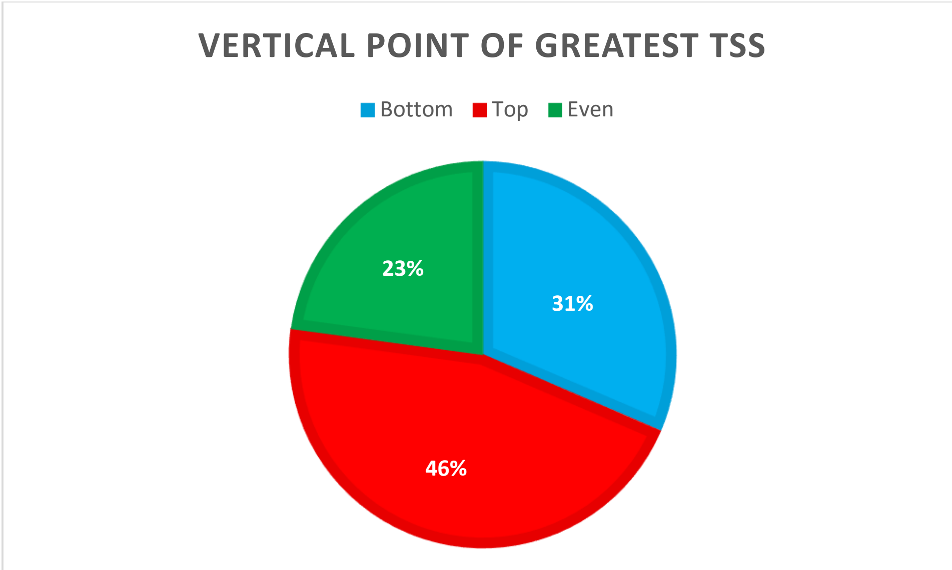


Figure 4: Vertical Sampling Position with Greatest Concentration

The first event at Second Creek had TSS means of 34 ppm and 27 ppm with standard deviations of 8 ppm and 13 ppm respectively (Figure 5). The RSD, relative standard deviation, in this set of sampling was 24% and 48% respectively. The second event at Second Creek had means of 40 ppm and 33 ppm with standard deviations of 4 ppm and 7 ppm respectively (Figure 6). Relative standard deviations were found to be 11% and 20% respectively.

The second site sampled was Beaver Creek, which had a mean of 15 ppm and a standard deviation of 6 ppm (Figure 7). The relative standard deviation was found to be 41%.While this precludes this set of sampling from evaluation of rate pulsing, evaluations can still be made toward the variation of entrained material along the given cross section.

Oostanaula Creek, had means of 13 ppm and 17 ppm with standard deviations of 5 ppm and 14 respectively (Figure 8).

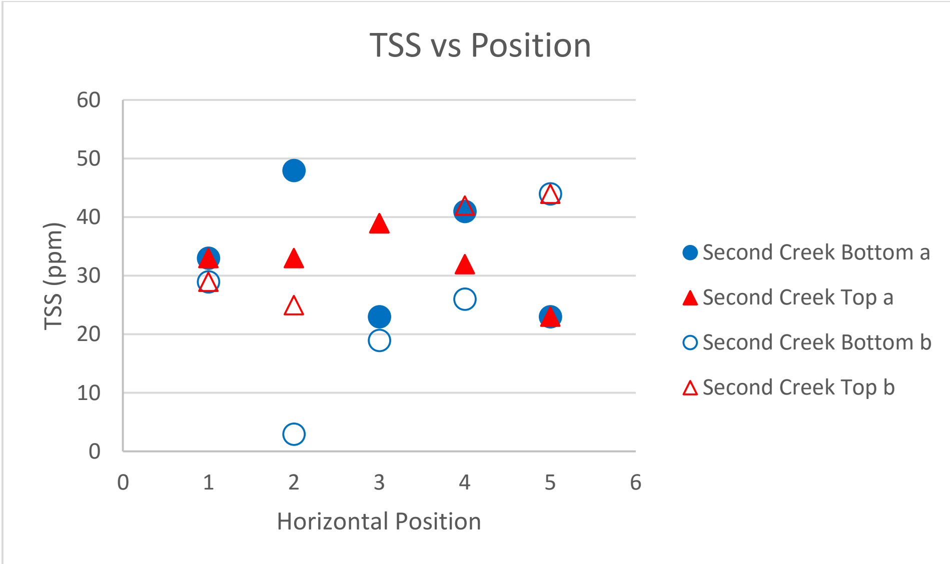


Figure 5: TSS v. Position Event 1 on Second Creek

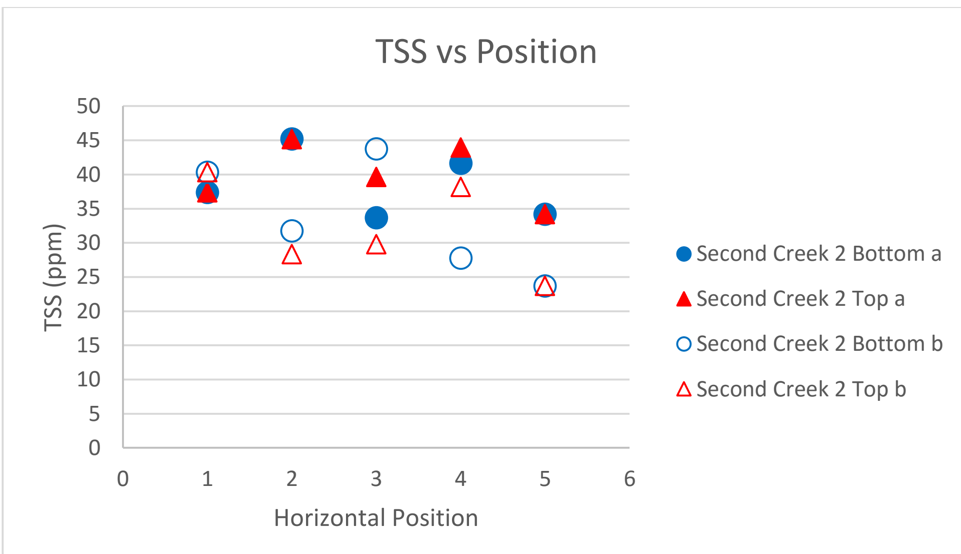


Figure 6: TSS v. Position Event 2 on Second Creek

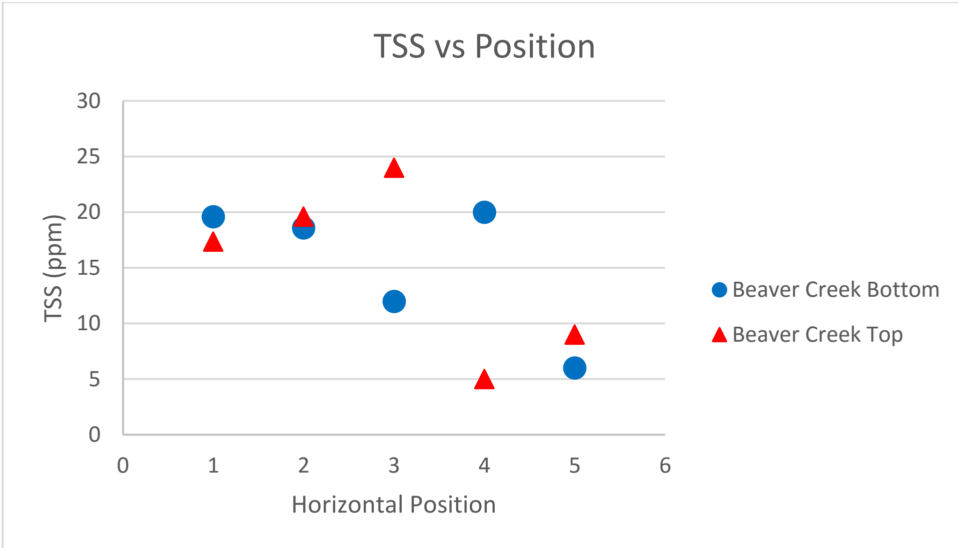


Figure 7: TSS v. Position Beaver Creek

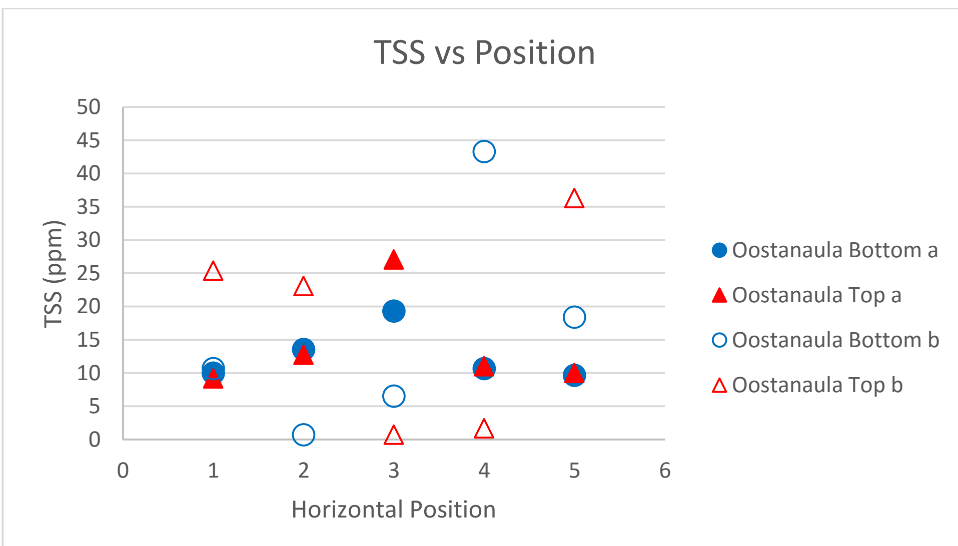


Figure 8: TSS v. Position Oostanaula Creek

This site was the widest with the largest stage, and the RSD greatly reflects this ppm respectively. Relative standard deviations were found to be 40% and 87% through greatest level of standard deviation respective to mean values. It should also be noted that this site had the largest level of variation in roughness due to a mix of woody debris and soft sediments along the cross section.

2.4.2 Indicator Bacteria

Summary results for means and standard deviations of microbial densities can be found in Table 3 for fecal coliform and in Table 4 for *E.coli*. Statistical results for nonparametric Spearman's correlations can be found in Table 5. While the graphs of residuals for both fecal coliform (Figure 9) and *E.coli* (Figure 10) show tighter clustering toward the banks, no position maintains great precision in consideration to the mean.

The first event at Second Creek had means of 73 MPN/ml and 61 MPN/ml with standard deviations of 13 MPN/ml and 16 MPN/ml respectively for fecal coliform (Figure 11). Means for *E.coli* were both 31 MPN/ml with standard deviations of 4 MPN/ml and 9 MPN/ml respectively (Figure 12). The standard deviation in this set of sampling accounted for 18% and 25% of the means for fecal coliform respectively. Alternatively standard deviations accounted for 4% and 9% respectively of the means for *E.coli*. Correlation in this sampling set was extremely low with the following Spearman p values: 0.2410 for fecal coliform to TSS along with 0.2048 for *E.coli* to TSS for the first set, and 0.5946 for fecal coliform to TSS along with 0.1071 for *E.coli* to TSS for the second set.

The second event at Second Creek had fecal coliform means of 96 MPN/ml and 84 MPN/ml with standard deviations of 34 MPN/ml and 12 MPN/ml respectively (Figure 13). Standard deviations of fecal coliform account for 35% and 15% of the total means respectively. In consideration to

Table 3: Fecal Coliform densities (MPN/ml) Summary

Site	Event	Sample	Average Fecal (MPN/ml)	Standard Deviation Fecal (MPN/ml)	RSD (σ/μ)	Flow Rate (cfs)
Second Creek	1	a	72.92	13.26	18.18	61
	1	b	61.41	15.56	25.35	
	2	a	4.49	2.17	48.27	NA
	2	b	2.38	1.87	78.70	
Beaver Creek	1	a	3.46	2.60	75.19	18
Oostanaula Creek	1	a	96.10	33.81	35.18	56
	1	b	83.57	12.38	14.81	

Table 4: *E.coli* densities (MPN/ml) Summary

Site	**Event**	**Sample**	**Average *E.coli* (MPN/ml)**	**Standard Deviation *E.coli* (MPN/ml)**	**RSD (σ/μ)**	**Flow Rate (cfs)**
Second Creek	**1**	**a**	**31.29**	**4.41**	**14.09**	61
	1	**b**	**30.70**	**9.32**	**30.35**	
	2	**a**	**3.73**	**1.58**	**42.43**	NA
	2	**b**	**1.64**	**1.40**	**85.45**	
Beaver Creek	**1**	**a**	**3.05**	**2.78**	**91.41**	18
Oostanaula Creek	**1**	**a**	**51.85**	**12.31**	**23.73**	56
	1	**b**	**37.68**	**11.21**	**29.77**	

Table 5: Spearman's p Nonparametric Correlation of Log Transformed Dataset

Site	**Event**	**Sample Set**	**Fecal Coliform vs. TSS**	***E.coli* vs TSS**
Second Creek	**1**	**a**	**0.2410**	**0.2048**
	1	**b**	**0.5946**	**0.1071**
	2	**a**	**0.5422**	**0.3494**
	2	**b**	**0.2395**	**-0.6386**
Beaver Creek	**1**	**a**	**-0.5908**	**-0.5503**
Oostanaula Creek	**1**	**a**	**-0.5140**	**-0.3634**
	1	**b**	**-0.1050**	**-0.0185**

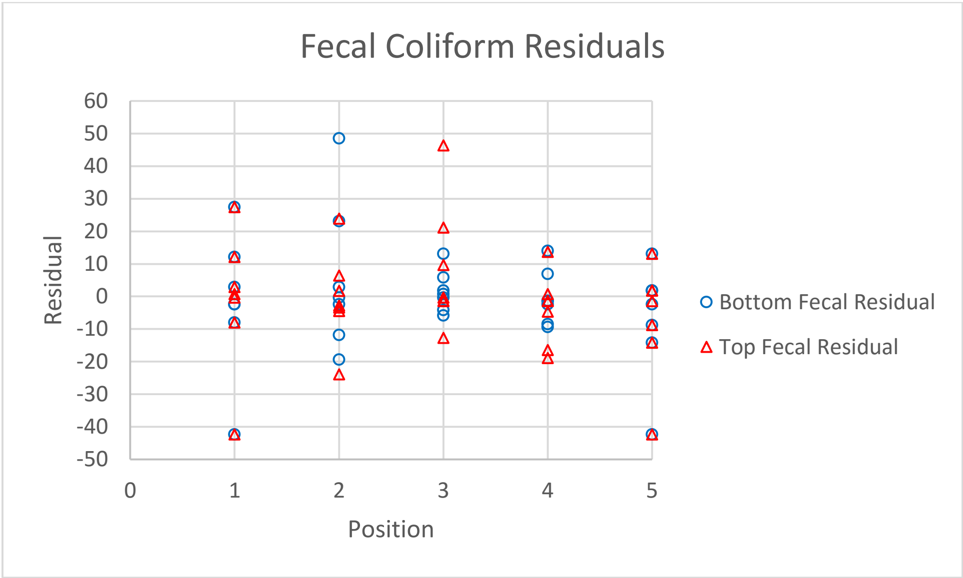


Figure 9: Fecal Coliform Residuals

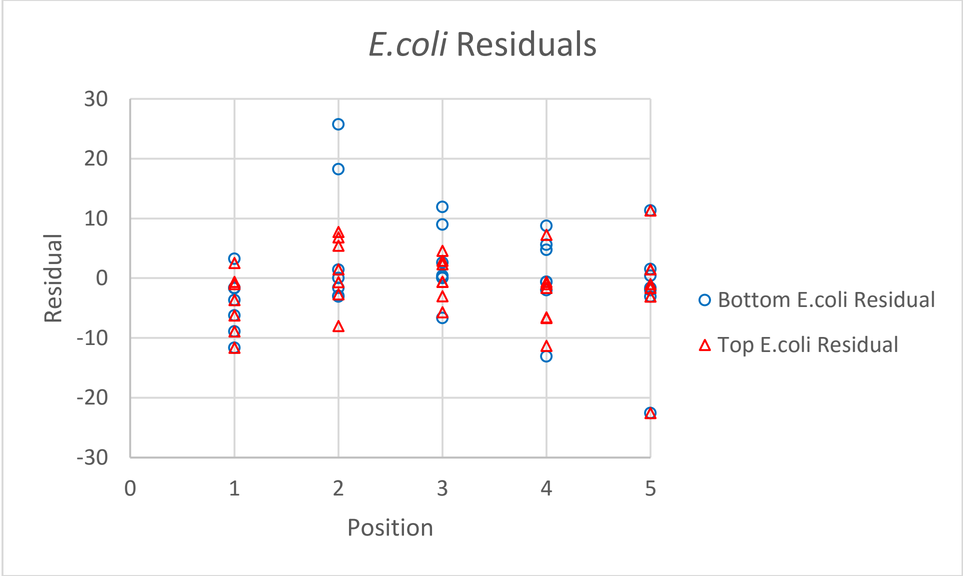


Figure 10: *E.coli* Residuals

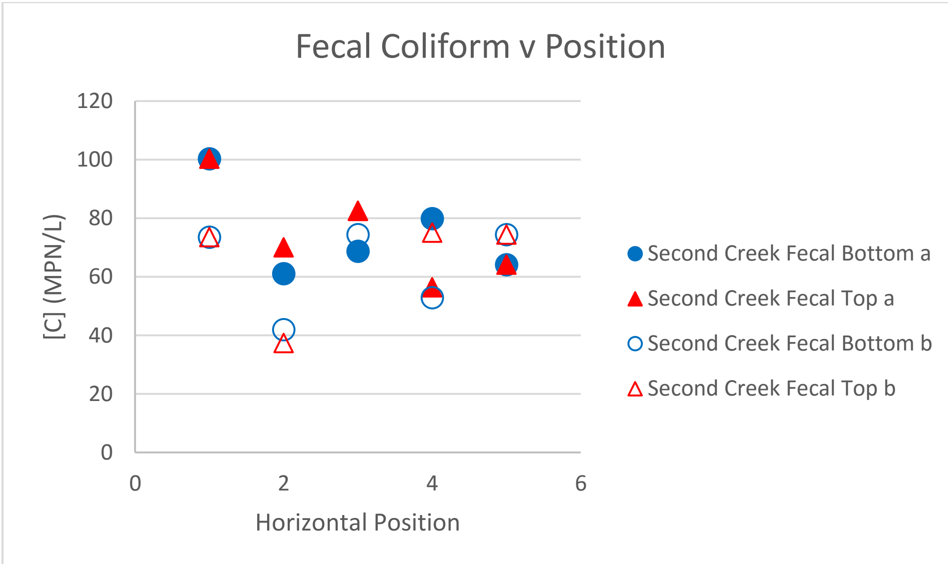


Figure 11: Fecal Coliform v. Position Event 1 on Second Creek

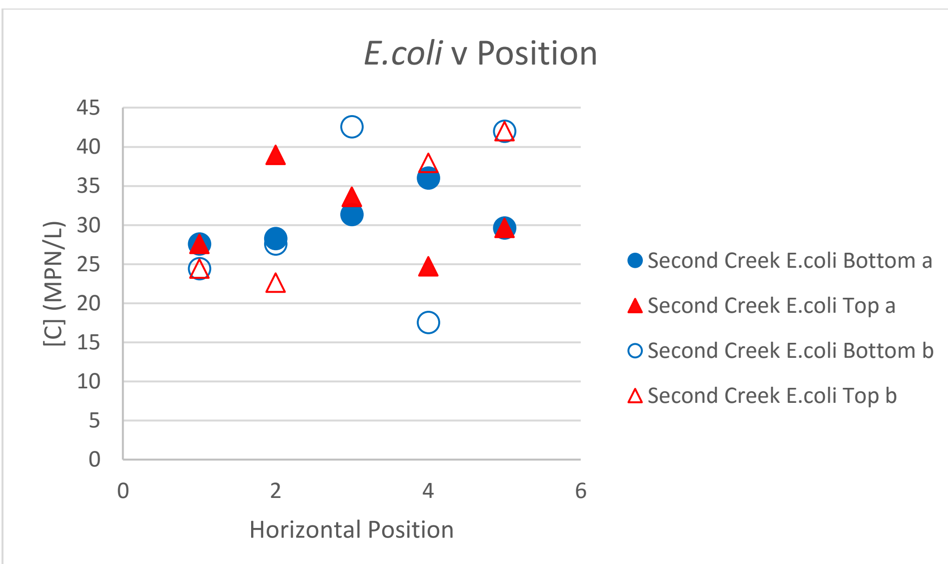


Figure 12: *E.coli* v. Position Event 1 on Second Creek

E.coli, means were found to be 52 MPN/ml and 38 MPN/ml with standard deviations of 12MPN/ml and11 MPN/ml (Figure 14). The standard deviations of *E.coli* account for 24% and 30% of their respective means. Correlation in this sampling set was extremely low with the following Spearman p values: 0.5422 for fecal coliform to TSS along with 0.3494 for *E.coli* to TSS for the first set, and 0.2395 for fecal coliform to TSS along with -0.6386 for *E.coli* to TSS for the second set.

The second site sampled was Beaver Creek, which had a fecal coliform mean of 4 MPN/ml and a standard deviation of 2.2 MPN/ml which accounted for 48% of its respective mean (Figure 15). In consideration to *E.coli* the mean was 4 MPN/ml with a standard deviation of 1.6 MPN/ml which

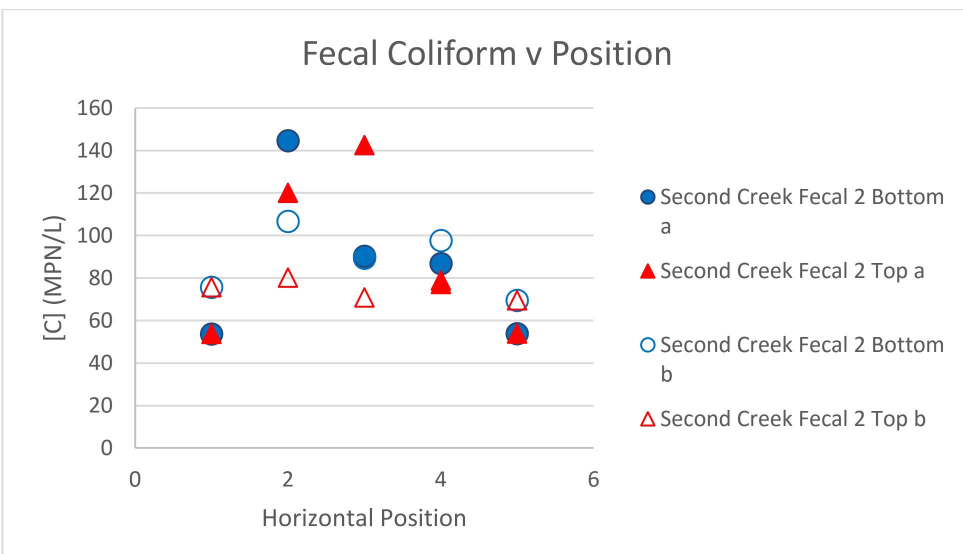


Figure 13: Fecal Coliform v. Position Event 2 on Second Creek

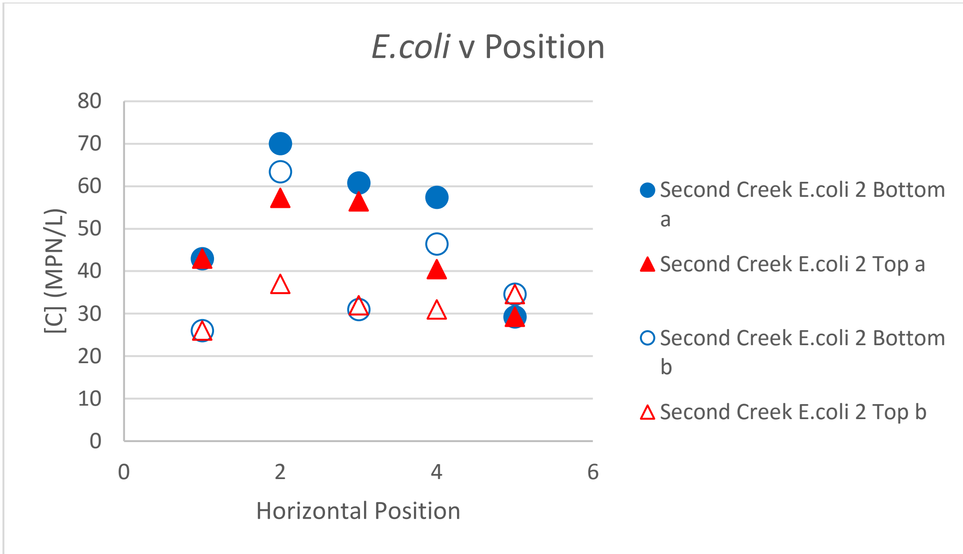


Figure 14: *E.coli* v. Position Event 2 on Second Creek

accounted for 42% of its respective mean (Figure 16). Upon attempting a second run the stage and turbidity had lowered below what was considered acceptable on site for sampling. While this precludes this set of sampling from evaluation of rate pulsing, evaluations can still be made toward the variation of entrained material along the given cross section. Correlation in this sampling set was extremely low with the following Spearman p values: -0.5908 for fecal coliform to TSS, and -0.5503 for *E.coli* to TSS.

Oostanaula Creek, had fecal coliform means of 2 MPN/ml and 3 MPN/ml with standard deviations of 1.9 MPN/ml and 2.6 MPN/ml respectively (Figure 17). Relative standard deviations of 79% and 75% respectively for the sampling sets. With regards to *E.coli* means were2 MPN/ml and 3 MPN/ml with standard deviations of 1.4 MPN/ml and 3 MPN/ml respectively (Figure 18). This site was the widest with the largest stage, and the relative variance in consideration to means greatly reflects this through respective ratios of standard deviations to means. It should also be

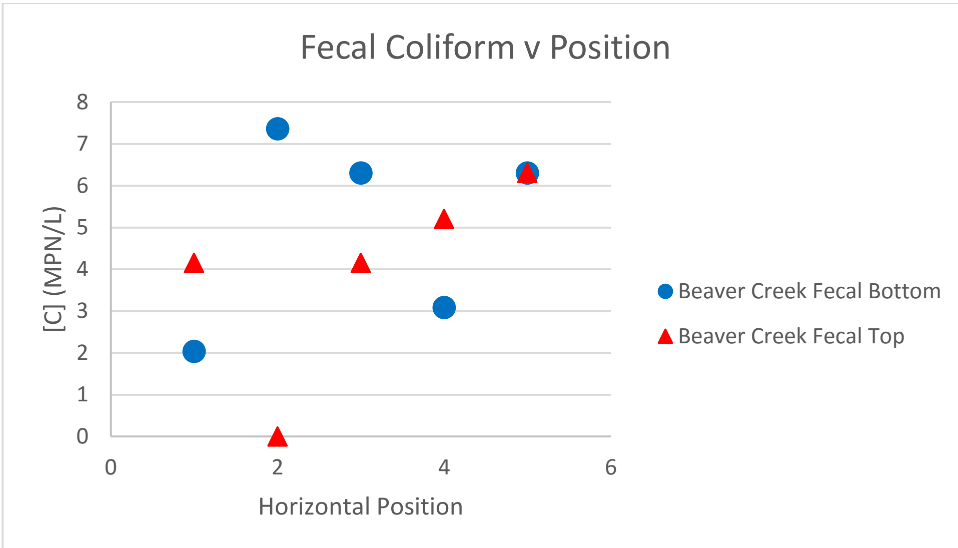


Figure 15: Fecal Coliform v. Position Beaver Creek

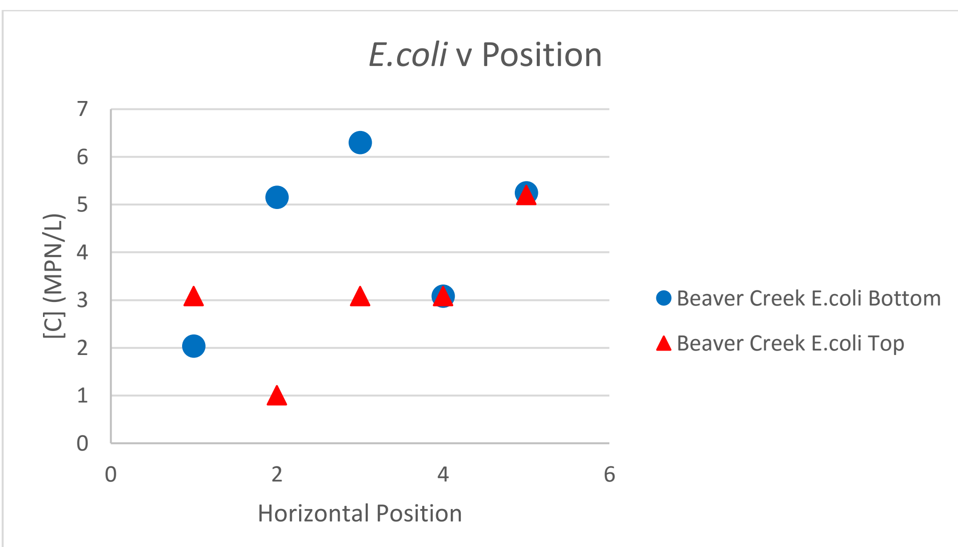


Figure 16: *E.coli* v. Position Beaver Creek

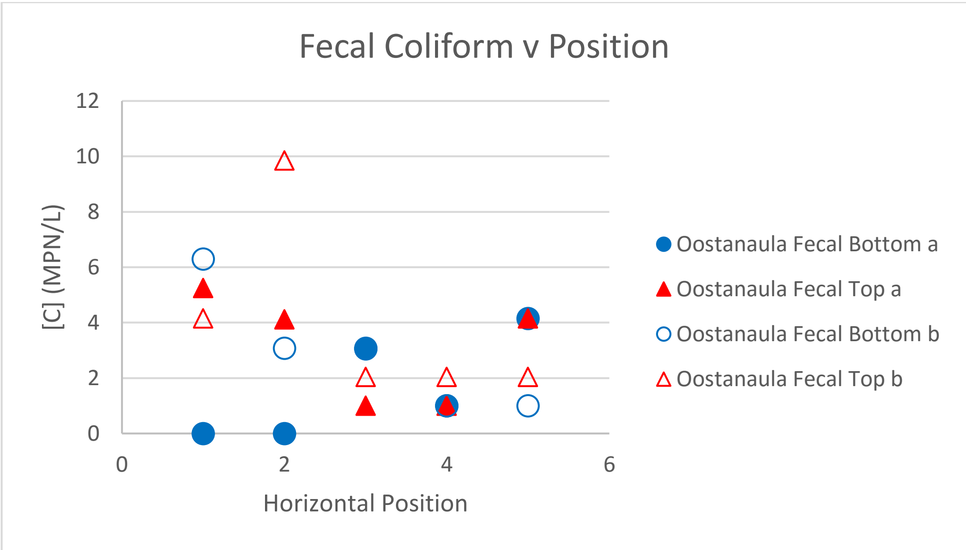


Figure 17: Fecal Coliform v. Position Oostanaula Creek

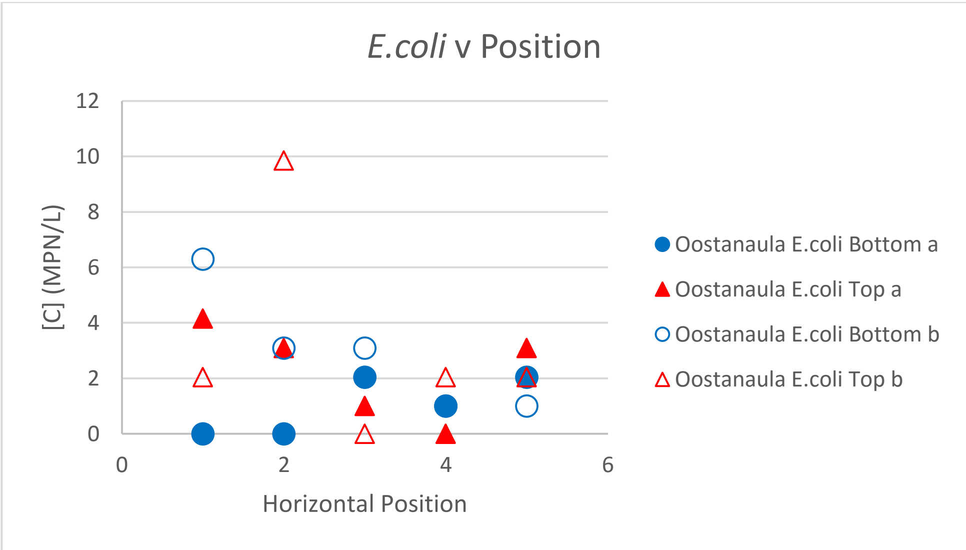


Figure 18: *E.coli* v. Position Oostanaula Creek

noted that this site had the largest level of variation in roughness due to a mix of woody debris and soft sediments along the cross section. Correlation in this sampling set was extremely low with the following Spearman p values: -0.5140 for fecal coliform to TSS along with -0.3634 for *E.coli* to TSS for the first set, and -0.1050 for fecal coliform to TSS along with -0.0185 for *E.coli* to TSS for the second set.

2.5 Discussion

The first event at Second Creek showed decreases in average TSS concentrations with increase in standard deviations for both events between the first and second sampling sets. Considering the second sampling sets were taken farther in to the falling limb of the hydrograph, it is thought this potentially could be indicative of an increase in variation corresponding with a decrease in flow. In conjunction with this speculation, it is theorized that the areas of greater roughness are more greatly subjected to the effects of decreasing flow than those which lack structures implementing as large effects of energy dissipation on the system. In turn, energy dissipation should not be considered a linear function laterally across the stream. This speculation lends to the assertion that roughness effects both spatial and temporal variation along a given cross section. While current sampling methodologies assume microbes to act like dissolved particles (Harmel & King 2005), these results elude to the opposite viewpoint. With the level of variance noted spatially present along the cross section, it is fairly obvious no single point can adequately represent the mean alone.

The second event sampled was much smaller with a much quicker response. The main difference being the scale at which variation occurred. The RSD still doubled as the event proceeded, but the values themselves were half of the initial event. While the percentages were lower in the second event, the overall concentrations were actually higher. It is speculated this is due to the system having depleted less of the stores available for entrainment during the smaller event, also lending to less variation due to the abundance of sediment to be entrained within areas of lower available energy. Higher initial RSD values for fecal coliform were also noted. While in all other cases the standard deviations increased in accordance with decreasing stage, it can be seen that there was a positive relationship between stage and standard deviations in this case. It is speculated this is due to the greater variance in energy regimes, due to the theorized effects of lower flows mentioned above, being expressed as larger variations in the particle size distribution captured within the sampling scenarios. During the extent where energy was still elevated enough to entrain larger particles, a larger variance in presence of bacteria was noted.

Both events at Second Creek showed tighter clustering of residuals between sampling sets on the bank for TSS, similar to what the residuals of all events sampled proposed. While the first event of the cross section through an event, the second event shows variance on the order of 25% as the event progressed. The first event also showed greater accuracy to the mean concentration along the banks, while the second event showed greatest accuracy in consideration to mean at the center of the cross section. This lends to the view that no single point can be defined along a cross section which will consistently lend to optimal precision or accuracy even at a respective site chosen.

Beaver Creek was only sampled once during the targeted high flow event; however, the RSD is clear proof of cross sectional variation. While the reach was straight far before the point sampled, one bank was notably higher in concentration than the other. Both banks presented root packs, yet only one was notably effected by their existence. It is also interesting to note there is less variance between vertical points sampled on the left bank in comparison to the right. With roughness clearly affecting entrainment on a fine enough scale that even associating factors in patches based upon material type alone would appear inadequate, the necessity for implementing more rigorous sampling applications for properly calibrating modeling applications to their respective reach is clearly validated. While counts were extremely low on Beaver Creek, the standard deviations accounted for their means in corresponding manners with those found on Second Creek. These percentages were slightly higher than those found on sites with reasonable microbial counts, but the point could still easily be made that they are representative of values that prove the persistence of cross sectional variation. They also directly correspond to the percentage of variation noted within the sediment data for this site, further supporting the association theorem presented within the hypothesis. This site also showed evidence of potential negative correlations between TSS and indicator bacteria through Spearman correlations. While this may seem to depose the theory of microbial to particulate association, an argument can also be made that it supports it. As larger particles are entrained, TSS potentially increases in overall weight. Considering that the indicator bacteria evaluated have been strictly shown to associate with smaller particles (Auer & Niehaus 1993, Characklis et al 2005, Jamieson et al 2005), as TSS increases it would be logical that in turn these microbial concentrations decrease. This lends to the evidence that microbial to particulate associations should be evaluated through constituents such as turbidity rather than TSS in the field.

Oostanaula Creek was both the widest and deepest stream evaluated, and the largest amount of variance in concentrations were noted on this site accordingly. This site consisted of the most sporadic variations in concentrations along the cross section between the two sampling events. Oostanaula Creek shared similar characteristics to that of the events at Second Creek, except at Oostanaula Creek the means increased along with the standard deviations. It is intriguing to note that the point which experienced the greatest variation between events was in the middle at 80% of stage, which is typically the point recommended for collecting grab samples. While the center of the stream had concentrations higher than average during the initial sampling set, it had concentrations much lower than average during the second sampling. While the largest ratios of standard deviation in respect to their corresponding means was noted in this channel, significant variation was noted throughout. The relative counts may be debatable in consideration to microbial variation along the channel due to how low the level of concentrations were found to be, but similar variation was noted within the suspended solids evaluated at this site. Not only were similar variations noted, but the graphs appear to show systematic variations occurring in accordance with sediment values. This eludes to a prevalence of both variation and microbial partitioning occurring at these sites. Also to note, while the percentage by which the standard deviation accounted for the mean during the duplicate sample did not follow a similar trend of increasing in regards to fecal coliform, the increases in standard deviation within the duplicate sample did follow the trend noted within sediment regimes. The variation noted in percentages was likely due to the small counts which were received during this event, rather than being indications of any significant trends. Spearman correlations were also very misconstrued within this sampling effort. They maintained a negative correlation between sediment loads and indicator bacteria concentrations, and then decreased as the event progressed. The most likely explanation from the literature reviewed is that grain sorting was driving the variation found in this stream, similar to the scenario explained on Second Creek.

2.6 Conclusion

With sampling being the initial form of uncertainty in water quality datasets, it should be of critical importance to understand and minimize uncertainty to generate accurate models which can inform watershed management. With RSDs ranging between 11% to 87% for TSS and 14% to 91% for indicator bacteria the 7 sampling trials, it is safe to assert that lateral variation in entrainment occurs. With the majority of events showing evidence of RSDs over 40%, it becomes obvious that lateral variation cannot simply be ignored in model calibration efforts. Also with the fact that concentration profiles do not follow the typical assumptions of greatest concentration at the bed made for generating them, it is validated that the methods utilized be reassessed. While the processes involved with variation are not fully understood, enough research has been done to initiate these theories in to current sampling and modeling methodologies. Multiple modeling efforts have been made to account for these phenomena (Nitsche et al 2011, Tian et al 2002, Yager et al 2007), however none have been able to completely mitigate the uncertainty inherent within the dataset. It is theorized that this is due to the lack of effort initiated toward better evaluation of the system through more rigorous sampling efforts and lack of calibration due to insufficient data input for modeling programs. Uncertainty in any dataset is a trickledown effect beginning with the initial collection of the data itself, hence the situation will never improve while the uncertainty incorporated with the initial efforts are ignored.

While many other studies have shown associations between microbes and particulate matter (Auer & Niehaus 1993, Characklis et al 2005, Jamieson et al 2005), Spearman correlations were extremely variable within this study lending to the opposite viewpoint. Correlations ranged from 0.5946 to -0.6386, showing both positive and negative associations between microbial counts and suspended sediment concentrations. With little consistency noted even between sample sets, no association can be made between suspended sediment and indicator bacteria in this dataset. Even in the cases which showed stronger correlations. One potential explanation for this begins with the fact microbes have been shown to only associate with smaller particles (Auer & Niehaus 1993, Characklis et al 2005, Jamieson et al 2005). This generates issues in consideration to how particle size distribution effects TSS concentrations. With the largest particles having the greatest effect on the outcome of the weight used to calculate this concentration, the correlation to prove particle to microbial association exists could easily be misconstrued. With the evidence of associations which has presented itself within partitioning studies, it is recommended that further research be implemented within the field which analyzes the particle size distribution of the TSS concentrations gathered in order to attempt correlations purely to the particle sizes which microbial populations have been shown to associate with.

While no clear association between particles and microbes were noted in this study, the lateral variation of both constituents was prevalent in all sampling sets. Not only was variation noted, but alterations between these lateral variations was noted between sampling sets within the same event. With variation being both prevalent and sporadic in nature, modeling frameworks must utilize sampled data to appropriately calibrate models in consideration to the uncertainty generated by this variance. If some systematic variance had been noted on a site specific scale, it might be possible to simply adjust the model for site specific characteristics. Unfortunately with the dynamic nature of entrainment, depending upon stores of material along with characteristics of flow, variance in entrainment has been noted in other studies for similar events on a particular reach (Buffington & Montgomery 1997). Without the implementation of a sampling based calibration step, management programs will continuously miss the mark unless routinely overdesigned. Considering the necessity of sedimentation to maintain healthy stream life, sediment load cannot be overly removed from the system without generating negative effects. With persistence and growth recognized in the system, natural input being virtually unavoidable in any area with living species, and the introduction method in to the system being primarily broad scale runoff, it is infeasible for management practices to ever truly be able to completely remove microbial communities from surface waters. Hence fate and transport need to be properly analyzed for appropriate management techniques to be developed, much less implemented, to contend with potential arising issues in concern to microbial evaluation (Davies 2000). Therefore while it may cost money to initiate these practices, it will indefinitely save money in the grand scheme to more efficiently manage systems rather than overly manage systems in attempts to account for the uncertainty inherent within these datasets.

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Chapter 1

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Appendix

Table 6: Cumulative Results

Sample	Type	TSS(ppm)	[C]Fec$_{0.01}$ (MPN/L)	[C]Ecol$_{0.01}$ (MPN/L)
SC001	L	33	100.358	27.62379
SC002	MLT	33	70.07202	38.99282
SC003	MLB	47.9798	61.12883	28.28843
SC004	MT	39	82.56925	33.61952
SC005	MB	23	68.75286	31.38734
SC006	MRT	32	56.44373	24.72165
SC007	MRB	41	79.85808	36.02776
SC008	R	23	64.17123	29.64223
SC009	L	29	73.52448	24.4434
SC010	MLT	25	37.45742	22.67084
SC011	MLB	3	42.02225	27.62379
SC012	MB	19	74.45998	42.62032
SC013	MRT	42	75.04892	37.95359
SC014	MRB	26	52.88459	17.5847
SC015	R	44	74.45998	42.02225
SC020	L	37	53.72715	42.94553
SC021	MLB	45	144.6096	70.07202
SC022	MLT	45	120.0009	57.26831
SC023	MB	34	90.20702	60.83425
SC024	MT	40	142.4939	56.44373
SC025	MRB	42	86.76378	57.4205
SC026	MRT	44	77.20358	40.50297
SC027	R	34	53.75985	29.28679
SC028	L	40	75.61274	26.01028
SC029	MLB	32	106.7048	63.40904
SC030	MLT	28	80.17804	36.98875
SC031	MB	44	89.39787	31.00241
SC032	MT	30	70.81261	31.97508
SC033	MRB	28	97.57433	46.41915
SC034	MRT	38	78.88636	31.00241
SC035	R	24	69.41447	34.60555
BC001	LT	17.36842	4.156506	3.086889
BC002	LB	19.58763	2.038123	2.038123
BC003	MLT	19.60784	0	1.00941
BC004	MLB	18.59296	7.363701	5.155638
BC005	MT	24	4.156506	3.086889
BC006	MB	12	6.304554	6.304554

Table 6 Continued

BC007	MRT	5	5.201314	3.086889
BC008	MRB	20	3.086889	3.086889
BC009	RT	9	6.304554	5.201314
BC010	RB	6	6.304554	5.247822
0A001	RB	9.655172	4.156506	2.038123
0A002	RT	10	4.156506	3.086889
0A003	MRB	10.66667	1.00941	1.00941
0A004	MRT	11.03448	1.00941	0
0A005	MB	19.33333	3.060268	2.038123
0A006	MT	27.08333	1.00941	1.00941
0A007	MLB	13.54839	0	0
0A008	MLT	12.75168	4.120175	3.086889
0A009	LB	10	0	0
0A010	LT	9.166667	5.247822	4.156506
0A011	RB-2	18.4	1.00941	1.00941
0A012	RT-2	36.2963	2.038123	2.038123
0A013	MRB-2	43.33333	1.00941	1.00941
0A014	MRT-2	1.666667	2.038123	2.038123
0A015	MB-2	6.542056	3.086889	3.086889
0A016	MT-2	0.714286	2.038123	0
0A017	MLB-2	0.746269	3.086889	3.086889
0A018	MLT-2	23.07692	9.848915	9.848915
0A019	LB-2	10.68702	6.304554	6.304554
0A020	LT-2	25.38462	4.156506	2.038123

Vita

Mr. Thomas Michael Walton was born in Knoxville , TN on March 2 1989. After graduating from Central High School in December 2006, he then pursued work as an Electrician through an apprenticeship program with Action Electrical. Along with his pursuits in the field, Michael also began his college career taking night classes at Pellissippi State Technical Community College. Unfortunately economic stress cut this electrical career short prior to completing apprenticeship training, but fortunately this began his path toward engineering in August 2008 at Pellissippi Technical Community College. After completing his Associates of Science with a focus in Mechanical Engineering in May 2011 while working with Woolf McClane Bright Allen and Carpenter PLLC, Michael then transferred to the University of Tennessee Knoxville in August 2011. Within a semester of study at University of Tennessee, it became evident that his passions were more in the field of Civil rather than Mechanical engineering. After switching programs, he discovered his passion in Water Resources Engineering while working as an Undergraduate Research Assistant for Dr. John Steven Schwartz during his senior year in 2013. Upon graduating with a Bachelor of Science in Civil Engineering Summer of 2014, he then continued his educational journey by pursuing a Master of Science degree in Environmental Engineering with a focus in Water Resources while working as a Graduate Research Assistant for Dr. Jon Michael Hathaway and Dr. John Steven Schwartz. Michael lives on campus and enjoys hiking, camping, and strumming a guitar.