MODELING ONSITE WASTEWATER TREATMENT SYSTEMS IN THE DICKINSON BAYOU WATERSHED

A Thesis

by

AARON ANTHONY FORBIS-STOKES

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

August 2012

Major Subject: Civil Engineering

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Approved by:

Co-Chairs of Committee,	Bryan Boulanger
	Clyde Munster
Committee Members,	Raghupathy Karthikeyan
	Binayak Mohanty
Head of Department,	John Niedzwecki

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ABSTRACT

Modeling Onsite Wastewater Treatment Systems in the Dickinson Bayou Watershed.
(August 2012)

Aaron Anthony Forbis-Stokes, B.S., Texas A&M University

Co-Chairs of Advisory Committee: Dr. Bryan Boulanger
Dr. Clyde Munster

Onsite wastewater treatment systems (OWTSs) are a commonly used means of wastewater treatment in the Dickinson Bayou watershed which is located between Houston and Galveston. The Dickinson Bayou is classified as "impaired" by the Texas Commission on Environmental Quality due to high levels of bacteria, specifically *E. coli*. Failing OWTSs within the bayou's watershed are possible sources for the impairment of the bayou. Conventional OWTSs, comprised of a septic tank and a soil absorption field, rely heavily on soil treatment of effluent. The type of soils is a significant factor in treatment capabilities. In the Dickinson Bayou watershed, soils are primarily composed of clays, which are known to be problematic for conventional systems as they restrict water flow and create perched water tables. These perched water tables may contribute to surface runoff during rainfall events. The HYDRUS modeling software for water and solute flow through variably saturated media was used to simulate OWTSs in the Dickinson Bayou watershed. HYDRUS was used to simulate conventional septic systems with soil absorption fields, aerobic treatment units (ATUs) with spray dispersal systems, and mound systems. Results found that the simulated conventional systems fail due to high water tables and clay soils. However, system failure in the watershed remains uncertain due to lack of field data for validation. The alternative systems mitigate these issues, but ATUs can lead to higher contamination levels without proper maintenance. Therefore, mound systems are the suggested alternative for OWTSs in the watershed.

DEDICATION

In my final step as a student at Texas A&M University, I would like to dedicate this thesis to my family who made everything possible, to the classmates who helped me along the way, and to my friends who made this journey better than I could have imagined.

ACKNOWLEDGMENTS

I would not have come this far without my advisor Dr. Bryan Boulanger. I will be forever grateful to Dr. Boulanger for his helping me decide to pursue a Master's degree and a future doctoral degree, meeting with me weekly to make sure I was on the right track, looking out for me, and always providing inspiration to do something more with engineering. Dr. Boulanger has been a great professor, great advisor, and great role model.

I would like to thank my co-chair Dr. Clyde Munster for taking me on for this project and allowing me this opportunity even as a student outside of his department. I would like to thank him for the time and help throughout this process to make sure this was a well-done project. I would like to thank the rest of my committee, Dr. Raghupathy Karthikeyan and Dr. Binayak Mohanty, for giving me their time and knowledge to bolster this research.

I would also like to thank Raghu Jana, Bhavna Arora, and Dipankar Dwivedi for their help with HYDRUS modeling. The time they volunteered outside of their own work to help me is greatly appreciated.

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CHAPTER I

INTRODUCTION AND LITERATURE REVIEW

Onsite wastewater treatment systems (OWTSs) are commonly used means of wastewater treatment in the Dickinson Bayou watershed. The Dickinson Bayou has been found to be impaired due to higher than acceptable concentrations of *Escherichia coli*. A hypothesized contributor to the impairment of the bayou is runoff from failing OWTSs. In order to find if OWTSs are contributing to the impairment of the Dickinson Bayou, typical designs of OWTSs in the watershed were evaluated by modeling with HYDRUS-2D. The systems were evaluated to determine if they failed. Hydraulic failure in the systems was marked by generating surface runoff and system saturation. Treatment failure was marked by allowing *E. coli* to reach the surface or the water table. Failures at the surface contribute to the impairment of the Dickinson Bayou.

Onsite Wastewater Treatment Systems

Onsite wastewater treatment systems (OWTSs) are commonly used for treating household wastewater across the United States. Systems have developed over time from latrines, to cesspools, to septic systems, to current advanced systems (USEPA, 1997). Approximately 25 percent of the population is serviced by onsite systems, most commonly by conventional septic tank systems with soil absorption fields (USEPA, 2002). According to a 2001 study, 1.5 million households in Texas use OWTSs (Reed et al., 2001). OWTSs are often used out of necessity because of a lack of access to centralized treatment systems, but they are also often used because they can be the most economical and practical option. An EPA study found that the initial and operating costs of an OWTS can be 22-80% less than that of a centralized system (USEPA, 1997). The majority of these systems are found in rural areas because of practicality and cost. Of rural households 65% use OWTSs (Motz et al., 2011). Traditionally, OWTSs were often seen as temporary solutions in rural areas before further development, but they are now being seen as more permanent solutions. OWTSs are effective wastewater treatment

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solutions and adequately protect public health in the environment when placed in areas with appropriate capabilities for type of system, designed and installed properly, and regularly maintained. OWTSs can also be more effective than centralized systems in ecologically sensitive areas in that some systems provide disinfectant and nutrient removal that centralized systems do not (USEPA, 2002).

Conventional Septic Tank and Drain Field Systems

A conventional OWTS is made up of a septic tank with effluent flowing into a soil absorption field. A typical layout of a conventional system is shown below in Figure 1. This system is the simplest and most cost-efficient option for OWTSs. The septic tank provides primary treatment for the system. The tank equalizes wastewater flow; stores solids, oils, and grease; and promotes anaerobic digestion for a portion of the waste (USEPA, 2002). Of the settled solids, up to 50 percent will decompose while the remainder will form sludge at the bottom of the tank. The soil below the field acts as the final treatment stage. Microorganisms in the soil break down remaining organics and nutrients while soil particles filter solids and pathogens. Primary removal mechanisms for bacteria removal in soil are straining, filtration, and inactivation (Motz et al., 2011).

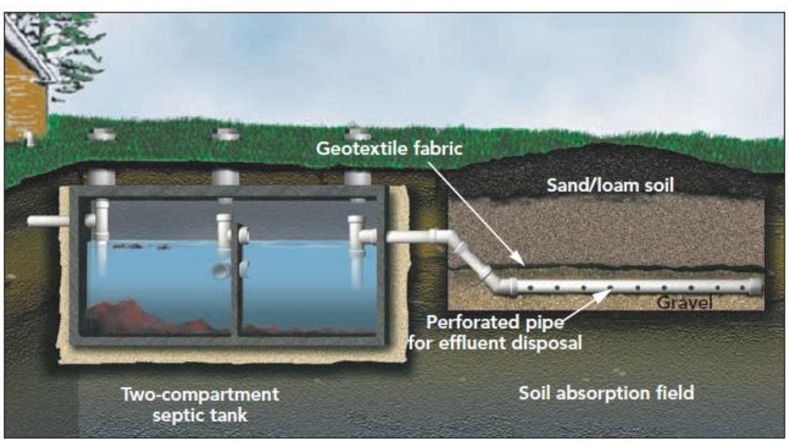

Figure 1. Cross-sectional side view of the typcial layout and design for a conventional septic tank and drain field (Lesikar, 2008b)**.**

The soil absorption field is commonly made of trenches containing a perforated pipe surrounded by a course media such as gravel. A layer of geotextile fabric is placed on top of the gravel portion of the trench to protect the gravel layer from sediments in above layers and to keep plant roots from intruding. The trench is then backfilled to become level with the surrounding ground. Sand or loam soils are preferred for the backfill, and soil removed to form the pit can often be used to backfill.

Soil type is a major factor in OWTS treatment efficiency. A conventional system does not function properly in clay soils, rocky soils, soils saturated for long periods, or soils with a high water table. The Texas Commission on Environmental Quality (TCEQ) requires that soil absorption fields only be used in suitable soils, Class Ib, II, and III (Texas Commission on Environmental Quality, 2001). Class Ia and IV soils are defined as unsuitable. A site evaluator or professional engineer must determine a site's soil characteristics before installation to determine if a soil absorption field may be applied in the desired area. Table 1 below lists soil types with corresponding class and saturated hydraulic conductivity (K_s) (Clapp and Hornberger, 1978) (Soil Conservation Service, 1975).

Table 1. Soil types with USDA classification and saturated hydraulic conductivity (K_s).

Soil	Class	K_s [cm/d]
Sandy soil with more than 30% gravel	Ia	>1520.6
Sand	Ib	1520.6
Loamy sand	Ib	1350.7
Sandy loam	II	299.5
Loam	II	60.0
Silt loam	III	62.2
Sandy clay loam	III	54.4
Silty clay loam	III	14.7
Clay loam	III	21.2
Sandy clay	III	18.7
Silty clay	IV	8.9
Clay	IV	11.1

Figure 2 displays the USDA Soil Textual Classification chart, and Table 2 displays soil particle sizes to determine soil type and classification.

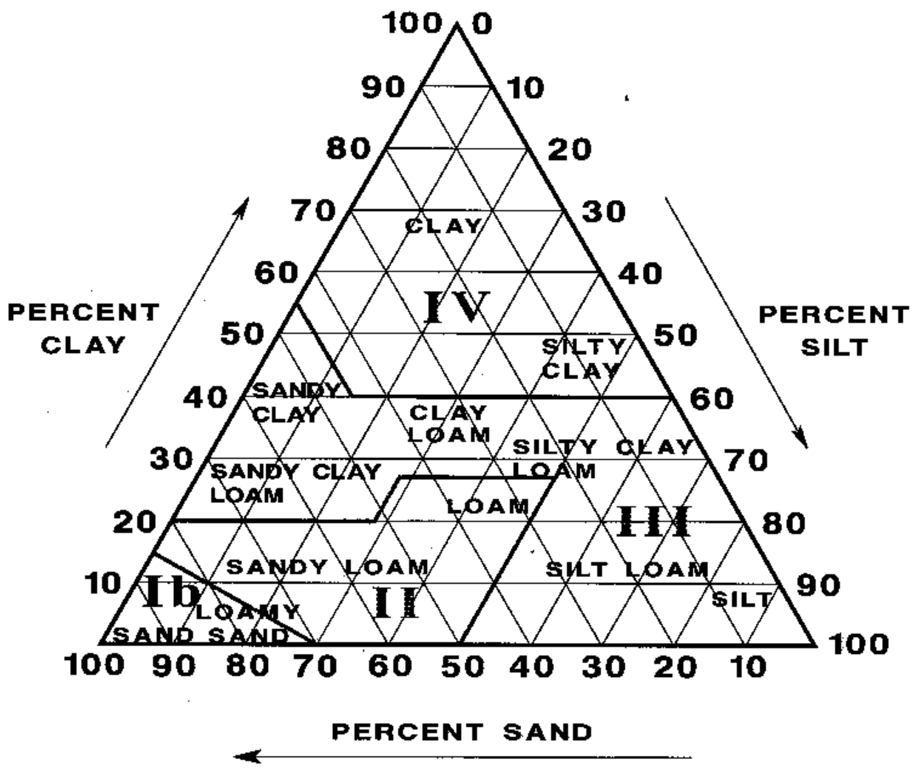

Figure 2. USDA Soil Textural Classification chart (Soil Conservation Service, 1975)**.**

Table 2. Particle sizes in diameter for soil types (Soil Conservation Service, 1975)**.**

Soil	Diameter [mm]
Clay	<0.002
Silt	0.002-0.05
Sand	0.05-2.0
Gravel	>2.0

Soils with clay are advantageous for contaminant removal as organic matter and clay promotes the removal of pathogens due to its negatively charged surfaces. Finer textured soils like clay also provide more filtration and sedimentation than coarse soils with larger pore spaces, thereby decreasing microbial movement (Bitton et al., 1974; Huysman and Verstraete, 1993; Tan et al., 1991). The issue with clay soils is associated with water transport, however. Clay soils have high retention times and low hydraulic conductivity values that prevent water from draining quickly. Clays can create perched water tables and increase the likelihood of a saturated drain field. Rocky and sandy soils have the opposite problem in that they allow effluent to pass through too quickly, not allowing enough time for proper soil treatment. Rocky and sandy soils also have larger pore spaces that allow for increased microbial movement. Saturated soils in the drain field are detrimental for contaminant removal. Studies have shown that the transport of bacteria and viruses is increased in saturated soils as saturated water flow will prevent much of the filtering processes of soils (Mawdsley et al., 1995). This finding is why soils saturated for long periods and soils with a high water table are not suitable for conventional systems. In order to prevent issues related to high water tables, a standard of having at least two feet of unsaturated soils between the outflow and water table has been created for adequate treatment (Lesikar, 1999a).

The conventional system is advantageous in that it is cost efficient in installation and maintenance and is often the cheapest treatment system option. Costs associated with conventional septic systems and soil absorption fields are $2,000 to $6,000 for installation and about $75 a year for maintenance (Lesikar, 2008b). The main maintenance component of conventional septic tank systems is having solids pumped out as the tank becomes full, typically every two or three years (Lesikar, 2008b). In spite of cost-efficiency, conventional systems have a major disadvantage in that they cannot be universally installed because of the dependence on suitable soils.

Aerobic Treatment Unit & Spray Distribution Systems

Aerobic treatment units (ATUs) with spray distribution systems are an alternative to conventional septic systems. A typical design for this system is displayed in Figure 3. These systems do not rely on soils for treatment but, instead, treat wastewater through aeration and chlorination. An aerobic treatment unit uses a similar process as a municipal wastewater treatment system and releases an effluent roughly the same quality. ATUs remove 85 to 98 percent of organic matter and solids, an effluent much cleaner than conventional systems that only remove up to 50% in the septic tank (Lesikar, 2008a). ATUs can be applied to areas with soils unsuitable (Class Ia and IV, Table 1) for conventional systems or with high water tables, such as the Dickinson Bayou watershed, because of higher effluent quality and the lack of a dependence on soil treatment.

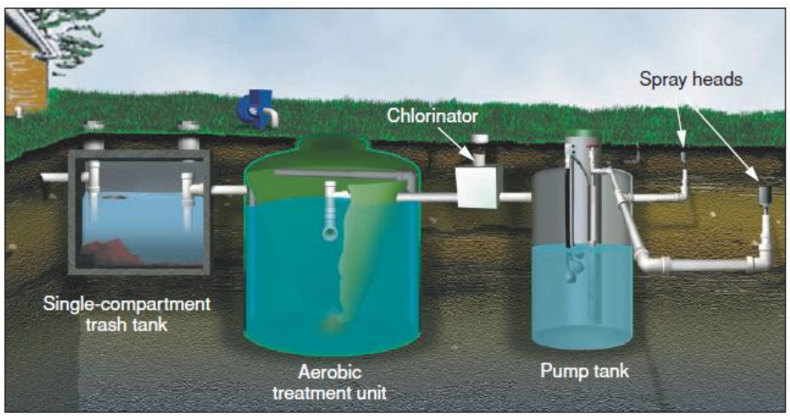

Figure 3. Cross-sectional side view of the typcial layout and design of an aerobic treatment unit system with spray distribution (Lesikar, 2008a)**.**

An aerobic treatment unit is made up of four processes: pretreatment, aeration, settling, and final treatment and dispersal. The pretreatment tank removes trash and other materials that cannot be degraded. The aeration chamber is where aerobic microbes decompose wastes. Oxygen is injected into this chamber by an air pump in order to create an aerobic environment conducive to microbial growth while it also causes mixing of wastewater for more contact with microbes. In the biological treatment process, wastes are consumed by microbes and transformed into harmless substances such as cell mass, non-degradable material, and gases (Lesikar, 2008a). The cell mass and nondegradable material is then removed in the settling or clarification chamber.

Material in this chamber settles from the water before the water leaves the system (Lesikar, 2008a). The water is then dispersed through a spray distribution system. The spray distribution system includes disinfection, storage, and dispersal. The disinfection process removes pathogens remaining in the wastewater after aerobic treatment and clarification by either ultraviolet light or, more commonly, chlorination. A pump tank stores wastewater after disinfection and pumps it to spray irrigation heads for distribution. Spray heads distribute the treated wastewater on the surface to dispose the wastewater and to use soil as a final treatment process. An additional advantage of the spray distribution system is that while the water is being treated by soils, it is also being used as irrigation. However, the rate at which water may be distributed is regulated and will require more surface area than a conventional system. Figure 4 below displays the amount of gallons of wastewater that can be applied per square foot of soil surface area per day (g/ft^2d) in Texas (Lesikar, 2008c). As seen in the map, the amount of wastewater that may be applied increases from east to west throughout the state. The reason for the increase is that precipitation decreases and evapotranspiration increases from east to west in Texas. These changes allow for more surface application because soils are less likely to become saturated with higher application further west. The maximum for the Dickinson Bayou watershed is approximately 0.04 g/ft^2d.

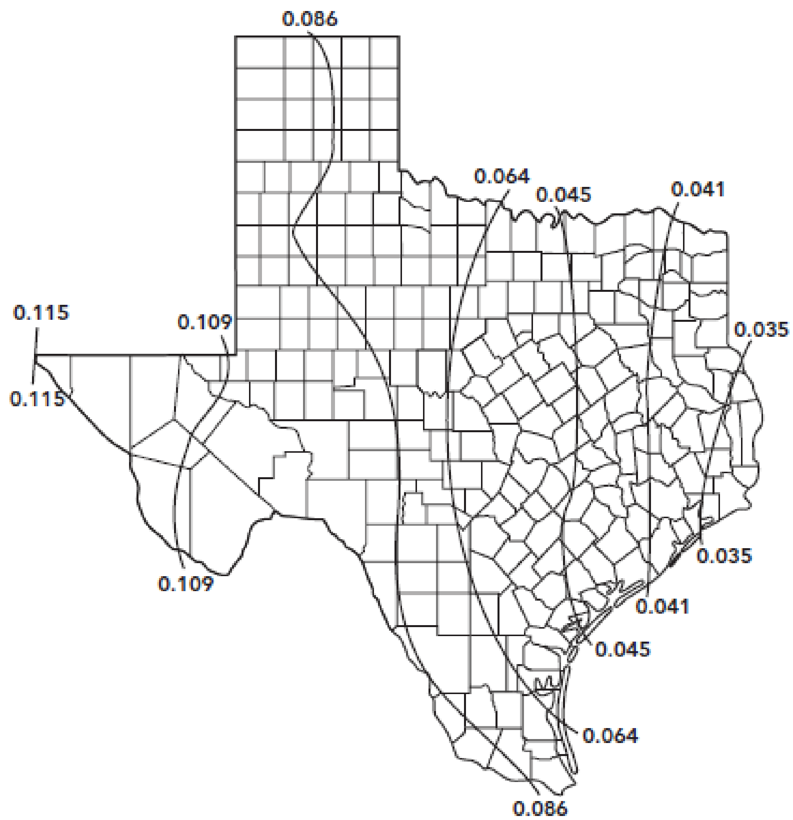

Figure 4. Map of Texas displaying the maximum amount of wastewater that may be surface applied from spray distribution systems in each region with values given in g/ft^{2}d (Lesikar, 2008c)**.**

A constant cover of vegetation such as grass or other landscaping needs to be maintained for spray distribution. Areas used to grow crops for human consumption and bare soils cannot be used. Further, land slopes must be less than or equal to 15% to reduce runoff. A uniform application of effluent is required, so a continuous land area without structures is recommended to simplify the spray distribution area. Areas away with minimal human contact are recommended (Lesikar, 2008c).

To meet qualifications, the aerobic treatment unit must treat wastewater to have concentrations less than 20 parts per million (ppm) of biochemical oxygen demand (BOD), 30 ppm of total suspended solids, and 200 colony-forming units of fecal coliforms per 100 ml (Lesikar, 2008c). In using chlorination as the disinfectant process, the chlorine level must be at least 0.1 ppm for pathogen removal (Lesikar, 2008c).

The major disadvantage associated with aerobic treatment units is that they require much more regular maintenance than conventional systems. The trash tank should be pumped every two to three years, a similar timeframe as emptying a septic tank. Solids also need to be periodically removed from the aeration and settling chamber. The air pump needs to be checked for continuous electrical supply, a clean air filter to the inlet, leaks in the system, and that the dissolved oxygen concentration is at least 1 milligram per liter but preferably 2 milligrams per liter (Lesikar, 2008a). A certified company must check the treatment unit every four months. Spray heads need to be checked as they are easily broken by lawn mowers or other lawn equipment. When using chlorination, chlorine needs to be regularly added, typically, each month. Finally, the smell of the effluent should be monitored. Foul odors can come from several problems such as overloading with organic matter, injecting substances toxic to microbes, or not sending enough waste for microbes (Lesikar, 2008a). The increase in maintenance and more complex system causes ATUs to be more expensive than conventional systems. The cost of installation is $4,500 to $7,500, and yearly maintenance costs range from $300 to $600 (Lesikar, 2008a).

Mound Systems

Mound systems are another alternative to conventional systems. Mound systems use the same design and treatment processes as conventional septic systems with soil absorption fields, but they are used as alternatives in areas with impermeable soils, high water tables, or a high restrictive boundary. A mound system mitigates these problems by raising the soil absorption field for more separation from the water table and restrictive boundary and adding soil with higher permeability. Figure 5 shows the changed design. A sand layer is placed below the field to meet a minimum of 24 inches separation of drain pipes from the water table and 18 inches of separation from impermeable soils or bedrock. Six inches of sandy loam is then placed on top of the geotextile fabric. The sandy loam is used to facilitate oxygen transport to the absorption area. On top of that layer is six inches of topsoil for plant growth (Lesikar and Weynand, 2002).

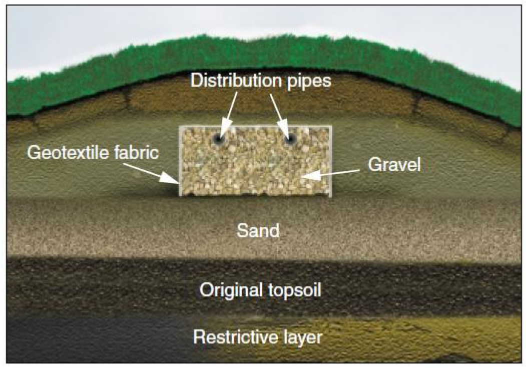

Figure 5. Cross-sectional front view of one drainage trench in a mound system with additional soil layers added (Lesikar and Weynand, 2002)**.**

Dickinson Bayou Watershed

The Dickinson Bayou watershed is the area of land draining into the Dickinson Bayou. The Dickinson Bayou watershed covers approximately 100 square miles located in Galveston and Brazoria counties, containing parts of the smaller cities of Alvin, Dickinson, Friendswood, League City, Santa Fe, and Texas City. The bayou flows from west to east approximately 24 miles and drains into Dickinson Bay which then drains into Galveston Bay. The area covered by the Dickinson Bayou watershed is generally not densely populated with, on average, about 620 people per square mile, and the majority of land use is classified as low intensity or open spaced developed, followed by cultivated land (Texas Commission on Environmental Quality, 2011).

The Dickinson Bayou watershed has a warm and wet climate. Warmer months are from May to September with an average temperature of 80°F while cooler months of November to April have an average temperature of 61°F (Quigg et al., 2009). In a study of the area from 2000-2006, the yearly rainfall average was 64 inches with May to

November receiving higher amounts than other months (Quigg, 2009). The water table for the majority of the watershed is generally within two feet of the ground surface because of the bayou's proximity to the coast. Soil types in the watershed are loams, clay loams, and clays. These soil types are classified in the "moderately well-drained," "somewhat poorly drained," and "poorly drained" drainage classes. The majority of the land classified as somewhat poorly drained (Glaveston County Drainage District Number One, 2007).

Current Issues

Dickinson Bayou is currently classified as "impaired" by the Texas Commission on Environmental Quality (TCEQ) and was first classified as impaired in 1996. An "impaired" body of water, according to the TCEQ, is a body of water that has been measured to consistently have higher than acceptable levels of bacteria as determined by the indicator bacteria *E. coli* in the freshwater portion of the bayou (Texas Coastal Watershed Program, 2010; Texas Coastal Watershed Program, 2010) (Texas Coastal Watershed Program). Indicators are used to show that in their absence, other pathogenic organisms are not present. They do not, however, necessitate the presence of pathogens when indicators are found (Pachepsky et al., 2006). *E. coli* is used as indicator bacteria because it relates to human and animal waste. The criteria for recreational use with *E. coli* in freshwater is that the average of samples taken has a concentration less than 126 colony forming units (cfu) or most probable number (MPN) per 100 ml, and/or less than 394 cfu or MPN per 100 ml in at least 25 percent of individual samples (Texas Commission on Environmental Quality, 2011). In a Total Maximum Daily Load (TMDL) study of the Dickinson Bayou watershed of dry and wet conditions containing over 760 samples for *E. coli*, the single-sample criteria of 394 cfu was exceeded 33% of the time (Texas Commission on Environmental Quality, 2011). Through TMDL studies and other TCEQ and Houston-Galveston area studies, the sources of contamination are predicted to have come from wastewater treatment facilities, stormwater runoff, sanitary sewer overflows, broken sewer lines, and contaminants from failing onsite wastewater treatment systems (OWTSs) reaching the bayou via runoff from surface discharge, stormwater runoff, or other modes (Texas Commission on Environmental Quality, 2011). Contamination due to stormwater runoff could be from permitted and unregulated sources that include both human and livestock wastes.

EPA studies have found 10-20% failure rates for OWTSs in the United States (USEPA, 2002). Reasons for failures can come from age, siting, design, regulation and oversight, compliance, education, and maintenance. In 2001 Reed, Stowe, & Yanke, LLC, performed a study to find the number of failing onsite systems in Texas and the reasons behind the failure (Reed et al., 2001). This study found that approximately 13% of OWTSs in Texas are chronically malfunctioning or 148,573 systems at the time. According to Reed, Stowe, & Yanke, the number of chronically malfunctioning systems poses a serious potential threat to public and environmental health. The report indicated that the three main reasons for failure in Texas are older systems installed in improper soils, in undersized lots, or designed too small for use; newer systems not being properly maintained; and a lack of education for system owners, enforcement programs, and records of OWTSs (Reed et al., 2001).

The cause for concern about malfunctioning OWTSs is that they pose a threat to public and environmental health. Failing OWTSs allow the transmission of disease by allowing bacteria and viruses to reach humans directly or indirectly through water resources. Failing OWTSs also endanger the environment through nutrient overloading and allowing other wastes to reach the environment. According to the EPA, contaminated drinking water is estimated to be the cause of 169,000 viral and 34,000 bacterial illnesses each year, and malfunctioning OWTSs contribute to contaminated drinking water. The EPA also reports malfunctioning OWTSs to be the leading factor to reduced harvests in shellfish growing areas (Reed et al., 2001).

In the Reed, Stowe, & Yanke, LLC, study, the major factors found to be contributing to OWTS system failure in Eastern and Coastal Texas (Region IV) are siting, age, and owner education. The study found 53% of OWTSs in Region IV to be in soils unsuitable for conventional systems, 48% in clayey soils, and that soils were the leading cause for failure (Reed et al., 2001). Another issue with drainage is that the climate in the region had increased rainfall in the winter, coupled with decreased evapotranspiration due to lower temperatures and high water tables. These features lead to saturated soils that are not conducive to removal of pathogens. Another study found that *E. coli* concentrations were higher during winter months and were found deeper in the soil profile (Motz et al., 2011). The reason for this finding is that low temperatures favor survival of bacteria (Kibbey et al., 1978; Zibilske and Weaver, 1978). In addition to these problems, about half of homes with OWTSs are more than 30 years old (USEPA, 2002). These older systems have a higher tendency to break down or face problems, and older systems in Texas are not under current regulations. Finally, the study found that system owners do not receive adequate education for their systems (Reed et al., 2001).

The TCEQ established a policy in 1997 that required permitting of OWTSs in order to improve oversight for OWTS siting, installation, and operation. Based on this permitting data and census data, the number of OWTSs in Dickinson Bayou is 4,857. Out of those systems, 1,546 are estimated to be failing (Texas Commission on Environmental Quality, 2011). In addition to permitting, this legislation changed soil classification to consider soil textures instead of being based on percolation tests and created methods for licensing site evaluators to properly evaluate soils and site characteristics. Soils must be evaluated by licensed evaluators or professional engineers and are classified as the soils in Table 1. These classifications determine what types of OWTSs are suitable for the desired area. Before this legislation soils were only evaluated by percolation tests to find rate at which water moves through the soil and did not consider soil type. These changes have improved siting and design of OWTSs. In the Dickinson Bayou watershed, new installations are typically aerobic treatment units because of soil conditions. However, many households in the watershed still rely on conventional OWTSs (Texas Commission on Environmental Quality, 2011).

The implementation of aerobic treatment units (ATUs), however, has not solved all of the issues related to conventional system failures in the Dickinson Bayou watershed. The Reed, Stowe, & Yanke study found that 92% of correspondents believed aerobic treatment units to function well, versus 67% for septic systems, but operation and maintenance of these systems causes problems. Pumping solids from systems was an issue for both aerobic treatment units and septic systems, but more maintenance problems are associated with ATUs. Disinfectants for the systems were either often incorrectly added or not added at all, and many residents did not renew required maintenance contracts. Without this necessary upkeep, ATUs are not nearly as functional as designed and can create worse problems than conventional systems due to higher contaminant concentrations applied to the ground surface that could then runoff into surface waters. Related to these issues is the lack of education for users (Reed et al., 2001).

HYDRUS Modeling

Modeling OWTSs in the Dickinson Bayou watershed will provide a better understanding of how different systems operate under varying conditions. Through modeling, OWTS selection and design can be optimized by examining the impacts of different parameters, climatic events, system designs, and soil structures. The selected model for this project is HYDRUS-2D. HYDRUS is a finite element model used to simulate subsurface flow of water, solutes, and heat (Šimunek et al., 2011). HYDRUS was selected because of its two-dimensional modeling ability and ability to model complex soil processes including variably saturated flow. HYDRUS does not assume steady state flow and can therefore respond to the rapid changes in soil moisture associated with varying effluent discharges from OWTSs and rainfall events while still being able to respond to gradual changes in the system (NIMSS, 2010). In addition, HYDRUS has already been widely used for solute transport in variably saturated media (Pang and Simunek, 2006). In its use, HYDRUS has also been used for modeling OWTSs (Beach, 2003)(Beal et al., 2008)(Pang et al., 2006) and *E. coli* transport (Bradford et al., 2006)(Foppen et al., 2007a; Foppen et al., 2007b)(Pang et al., 2004) previously. OWTS modeling has been done to display processes and evaluate performance while *E. coli* transport modeling has been done to better understand transport processes of *E. coli* in soils.

Newer modeling programs for colloid transport such as HYDRUS are much improved over earlier models, but limitations still exist for these new programs. Colloid transport models require more parameters than for other solutes, and many of these parameters are either difficult to estimate through experimentation or cannot be estimated at all. Some of these parameters are also tightly coupled, increasing uncertainty (Pang and Simunek, 2006). Facing these limitations, HYDRUS stands apart from other colloid transport models in that it considers different pore velocities and dispersivities for colloids and solutes, irreversible straining and nonlinear blocking, variably saturated water flow, and adjustment of kinetic rates based on the presence of colloids in the system (Šimunek et al., 2011).

The benefit of this project is that the majority of previous studies for OWTSs focused on specific processes in treatment systems and not overall effectiveness of systems or cumulative effects of OWTSs in one area (McCray et al., 2005)(Pang et al., 2006). This study evaluates the effectiveness and benefits of three different systems that are representative of typical systems in the Dickinson Bayou watershed. These results can help determine if OWTSs are contributing to the impairment of the Dickinson Bayou, and the processes can be replicated for use in other areas of concern. Further, the majority of HYDRUS solute transport modeling for OWTSs concerns nitrogen and phosphorous instead of *E. coli* (McCray et al., 2005); (Radcliffe and Bradshaw, 2011). Additional studies concerning *E. coli* transport are beneficial to its understanding as *E. coli* is commonly used as an indicator of OWTS effectiveness.

CHAPTER II

MATERIALS AND METHODS

Study Site and Characteristics

Study Site

The study site is an area between the cities of Dickinson and Santa Fe within the Dickinson Bayou watershed. The area contains two subdivisions, one newer with ATU systems and one older with conventional septic systems, that share a drainage ditch flowing into the Dickinson Bayou. The location was picked because of the inclusion of both types of systems and ability to sample on each side of the ditch for effluent quality for one system or another for future field studies. The simulated conventional system layout was derived from a Galveston County Health District Private Wastewater Disposal System Inspection Report. The report provided design and specifications for an onsite system currently in place within the study site that is considered standard for the area. The simulated system serves a four-person, 1,900 square foot home on a 150 foot by 300 foot lot. The system is composed of two tanks draining into six drainage trenches. The trenches are 75 feet long and 36 inches wide by 18 inches deep with five feet between trenches. Each trench has a four inch diameter PVC pipe surrounded by ½ to 2 ½ inch washed gravel and covered with backfill to ground surface. A cross-sectional front view of one drainage trench and surrounding soil is shown in Figure 6. The simulated mound system used the same design for the drainage trenches but had additional soil for the mound based on design specifications from AgriLife manuals (Lesikar and Weynand, 2002; Lesikar, 2008b). The simulated cross-sectional view is displayed in Figure 7. The ATU system with spray distribution was simulated based on design specifications from AgriLife manuals (Lesikar, 1999b). The cross-sectional view of the ATU system only displays the soil profile because the spray distribution is applied to the surface. The simulated system considered the same size strip as the mound and conventional system in order to compare results in the same manner (Figure 8).

Soil

The onsite treatment systems studied were all located on Mocarey series soils. The composition of soil in the area is 17.8% Mocarey loam (Ma), 44.6% Mocarey-Algoa complex (Mb), and 37.6% Mocarey-Cieno complex (Mc) (National Resource Conservation Service,). Mocarey-Algoa and Mocarey-Cieno are both complexes meaning they are made up of more than one type of soil. Mocarey loam was chosen for the soil profile because it represents the Mocarey series well, and it is not a complex. Complexes are made up of multiple types of soil series with different soil profiles. This variation makes determining the soil profile difficult without field samples. System cross-sectional profiles used for modeling including soil types are shown in Figure 6, Figure 7, and Figure 8. The cross-section of Mocarey loam is as follows: 0-11 inches, loam; 11-22 inches, clay loam; 22-52 inches, loam; and 52-60 inches, clay loam. The depth to the water table is 24 inches (National Resource Conservation Service,).

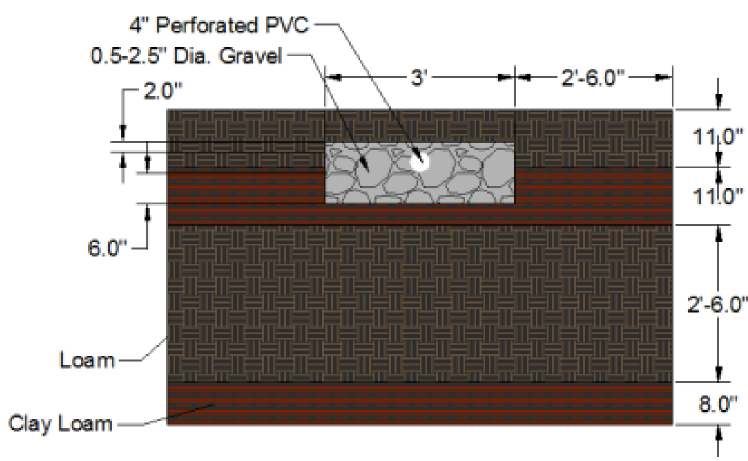

Figure 6. Cross-sectional front view of simulated system profile for a conventional septic system with soil absorption field. The system includes one drainage trench and spans from the mid-point between two trenches to the next mid-point.

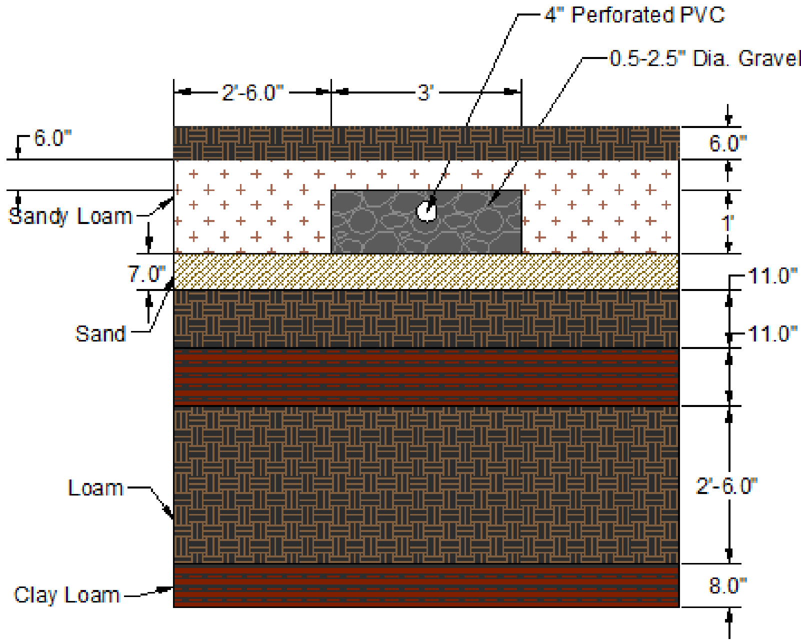

Figure 7. Cross-sectional front view of simulated system profile for a mound system. The system includes one drainage trench and spans from the mid-point between two trenches to the next mid-point.

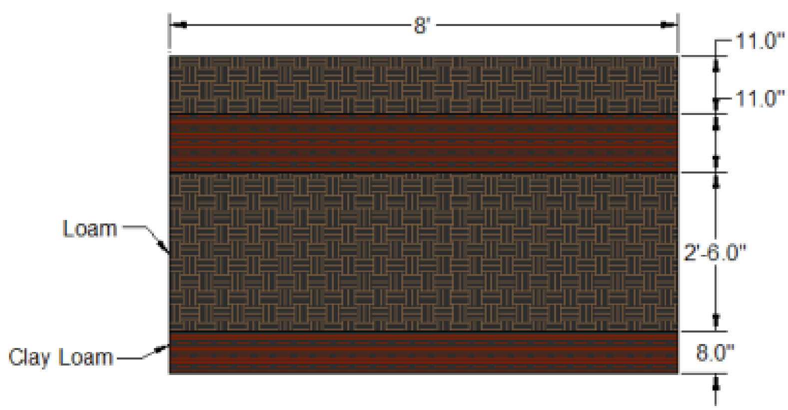

Figure 8. Cross-sectional front view of simulated system profile for an ATU with spray distribution.

Solute

Escherichia coli (E. coli) is one of the most used indicators of pathogen contamination. The benefit of using *E. coli* is that sampling and testing is simple, fast, and reliable. *E.*

coli is more favorable than other fecal indicators in that it is found in the feces of all warm-blooded animals and outnumbers other fecal coliforms in human and animal feces (Medema et al., 2003). *E. coli* is a gram-negative and facultative anaerobic organism. It has a rod-like shape with average dimensions of 2.0-6.0 µm x 1.1-1.5 µm (Whitman et al., 2009). *E. coli* is generally viewed as a threat to health, and the infectious dose of enterohemorragic *E. coli* is estimated to be as few as 10 cells (Pachepsky et al., 2006). However, most strains are harmless and only some cause illness. The illnesses associated with *E. coli* range from minor to severe and include gastrointestinal and diarrheal disease, urinary tract infections, and sepsis and meningitis (Foppen and Schijven, 2006).

The value of $1.2*10^6$ count/100 ml used for *E. coli* concentration comes from a measured value of septic tank effluent (Pang et al., 2004). Once in the soil, *E. coli* concentrations are affected by growth, die-off, and soil attachment and detachment. Microbial growth was not considered in the modeling study because replication processes are unlikely for *E. coli* in temperatures below 30 °C (Havelaar, 1991). Both soil and groundwater are below that temperature in the watershed. Die-off can be due to lack of nutrients or other environmental factors and can be caused by other organisms. Protozoa can ingest *E. coli*, removing it by predation, while other bacteria contribute to die-off by competing with *E. coli* for nutrients in the soil (Foppen and Schijven, 2006). The die-off rate of *E. coli* used for simulations in this project came from average values found sand column experiments associated with conventional OWTSs. The values of 0.193 day^{-1} and 3.53 day^{-1} while in water and adsorbed on soil, respectively, were found by the author to be similar to findings of other projects (Pang et al., 2004). This author also found that removal of *E. coli* from soil is primarily through filtration (87-88%) and secondarily through die-off (12-13%) (Pang et al., 2004).

Microorganisms are able to be transported via water as free cells, attached to particulates, or attached to soil particles (Pachepsky et al., 2006). The density of pathogenic microorganisms is close to the density of water, between 1.01 and 1.07

g/cm^3. This similarity in densities causes organisms to remain in suspension rather than becoming absorbed in the water (Pachepsky et al., 2006). *E. coli* has a hydrophilic cell wall, making the particle more resistant to adhesion to other particles (Foppen and Schijven, 2006). Clay is typically an effective absorption material due to its negative charge, but *E. coli* also has a strong negative charge, offsetting clay's absorption advantages (Pachepsky et al., 2006; Foppen and Schijven, 2006).

Model Simulation

HYDRUS is a finite element model used to simulate subsurface flow of water, solutes, and heat (Šimunek et al., 2011). HYDRUS can be used in 1D, 2D, and 3D versions, but for this project, HYDRUS-2D was used to simulate water and solute transport through the soil profile. Soil and system profiles in the Dickinson Bayou watershed were constructed in HYDRUS-2D and given parameters representative to insitu conditions. Simulations were run to represent a typical year of system use. The initial water and solute conditions of the soil profiles were set to consider the first year of use of the OWTS. Water transport initial conditions were based on pressure heads. Initial pressure heads in the system were set equal the distance to the initial water table depth (2 feet or 60.96 cm below the surface) where positive values were below the water table and negative values were above the water table. Initial solute conditions assumed no *E. coli* was present in the soil before operating the OWTSs. Observation nodes were placed on the surface (Node 1), below the drainage pipe (Node 2), at the depth of the initial water table (Node 3), and at the mid-depth of the bottom layer (Node 4) of each soil profile. Observation nodes for the conventional, mound, and ATU system are shown in Figure 9, Figure 10, and Figure 11, respectively. Even though the ATU system does not have a drainage pipe, the observation node below the drainage pipe was placed at the same depth in the ATU system as for the conventional and mound systems. These observation nodes return measured values for pressure head, water content, concentration, and adsorbed concentration.

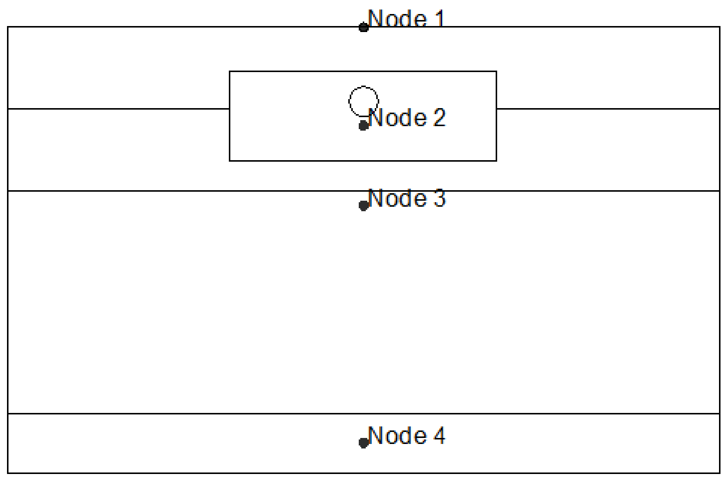

Figure 9. Observation node locations in cross-sectional view of the simulated conventional system.

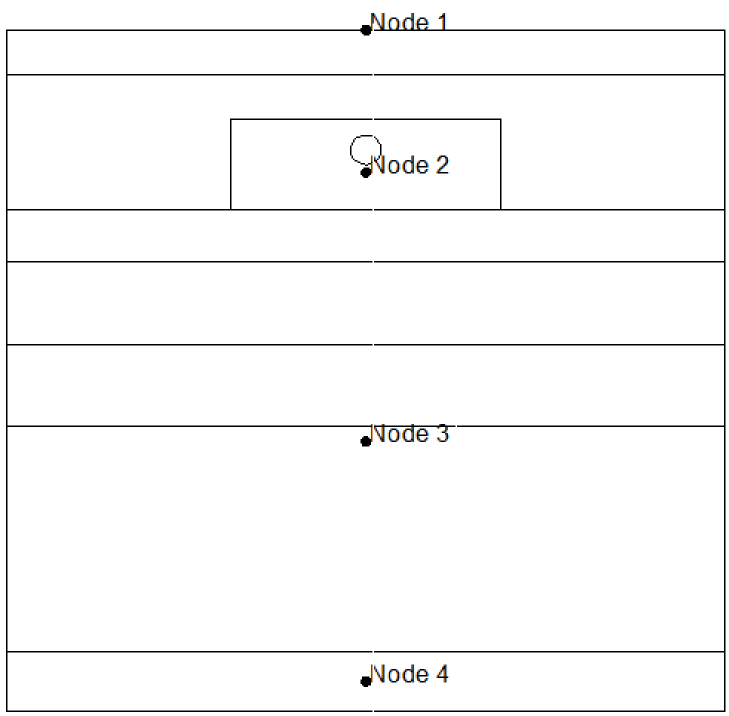

Figure 10. Observation node locations in cross-sectional view of the simulated mound system.

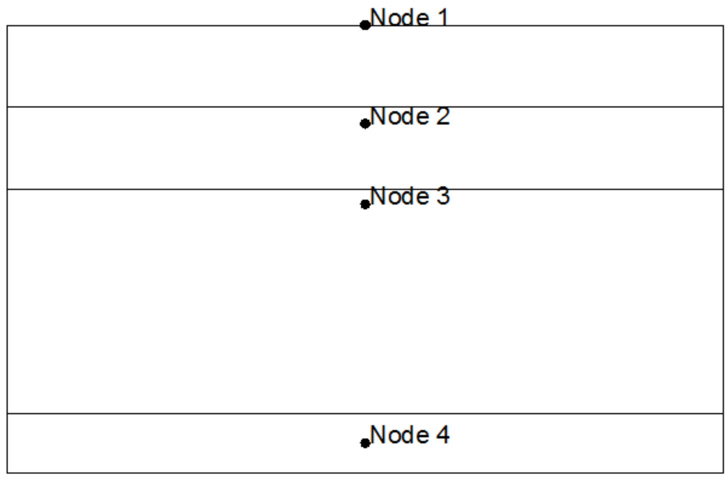

Figure 11. Observation node locations in cross-sectional view of the simulated ATU system.

Water Transport

Water transport in HYDRUS-2D is based on the Richards equation for saturated-unsaturated flow (Šimunek et al., 2011). Water transport fluxes into and out of the system include wastewater flow, precipitation, and evapotranspiration. Precipitation and evapotranspiration are represented by an atmospheric boundary layer on the top surface. A variable flux was specified for the bottom third of the drainage pipe using daily wastewater flow values. The upper portion of the drainage pipe was set as a seepage face in order to simulate that as the drainage pit become saturated, water will flow into the drainage pipe from surrounding saturated soil. The model was constructed with a central drain line and sides halfway between two lines. Therefore, symmetrical water flow on each side of the boundary was assumed and a no flux boundary condition was used on the sides of the soil profile. Establishing the bottom boundary condition required an initial simulation. The bottom of the soil profile is below the water table but is not a restrictive horizon. In this situation, the pressure head at the bottom boundary depends on the fluctuation of the water table within the profile. However, the version of HYDRUS-2D used for this project does not allow system-dependent boundary conditions. An initial simulation was run for one year considering a constant pressure at the bottom of the profile. This constant pressure head was the initial pressure head relative to the distance below the initial water table. Results from this setup showed that the pressure heads in the system fluctuated in accordance to precipitation events. These fluctuations, however, were minimized closer to the bottom boundary and were non-existent at the bottom boundary due to the constant pressure head boundary condition. A variable pressure head boundary condition was then created in order to make the bottom boundary condition more realistic. This variable pressure head was created by using the pressure head results from the node located at the depth of the initial water table. The changes in pressure head at the water table depth were recorded and then used to adjust the pressure head at the bottom of the soil profile. This variable pressure head boundary condition was then used for the system simulations.

The upper and side boundary conditions used for the conventional system were also used for the mound system. The same process to find the variable pressure head boundary condition for the conventional systems was also done to establish the bottom boundary condition for the mound system and ATU system. The ATU system used a different top boundary condition. The surface of this system is affected by precipitation, evapotranspiration, and spray distribution. Using the same atmospheric boundary condition as for the conventional and mound system and also using a variable flux for spray distribution would have been ideal, but HYDRUS only allows one boundary condition at each point. In order to create one boundary condition, spray distribution was modeled as precipitation, and the top boundary was set as an atmospheric boundary condition. Initial precipitation and spray distribution fluxes were added and used together as one precipitation for the atmospheric condition. The ATU system also used no flux boundary conditions for the sides.

Rainfall

Daily Rainfall data was provided by gauges from the Harris County Flood Warning System and the National Weather Service (Harris County Flood Warning System, 2012; National Weather Service, 2012). Data from five gauges located nearest to the study site from 2008 to 2011 were used. The distances of these gauges range from 4.5 to 7.75 miles from the simulated area. Although these gauges are all within 10 miles, rainfall values at each varied significantly. An average rainfall amount from the gauges was created for each day based on their distance to the studied area because of this variance between gauges. The average daily rainfall amounts then displayed wet and dry years from 2008 to 2011. In order to simulate a typical year of rainfall for the studied area, an average year of rainfall was created. Simulations were also run using the driest and wettest year of rainfall during the four year period. Rainfall graphs are shown in Figure 12, Figure 13, and Figure 14.

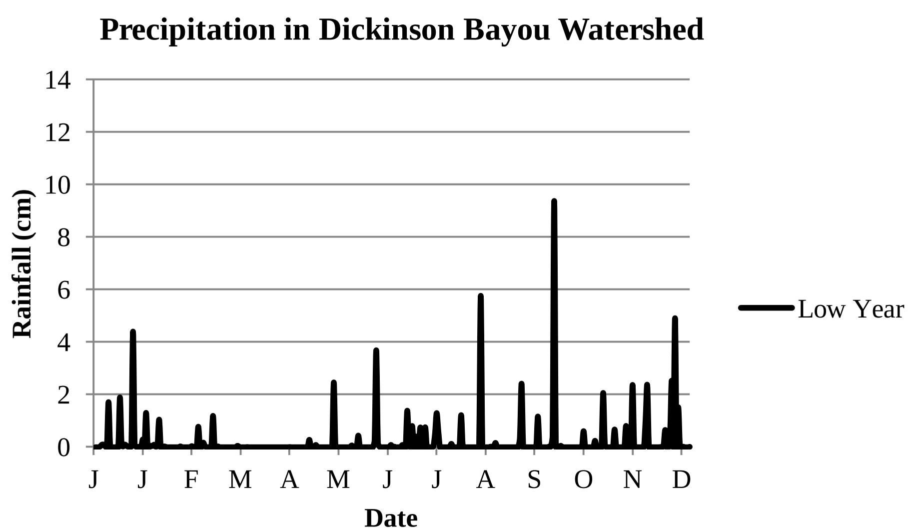

Figure 12. Precipitation in the Dickinson Bayou watershed during a dry year.

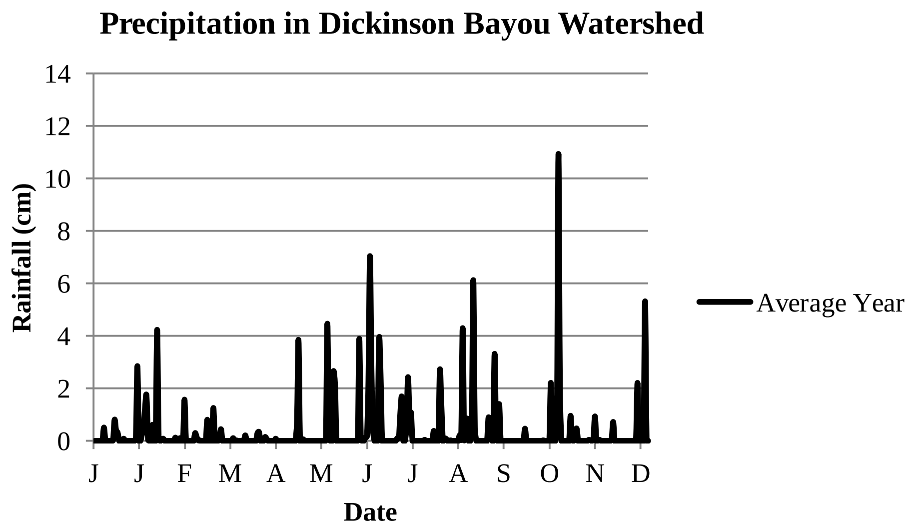

Figure 13. Precipitation values for the in the Dickinson Bayou watershed for a calculated average year of rainfall.

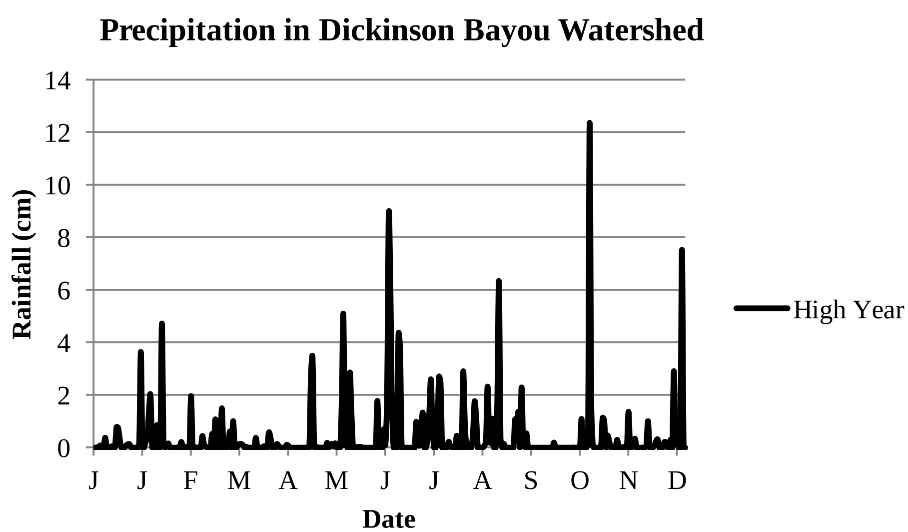

Figure 14. Precipitation in the Dickinson Bayou watershed during a wet year.

Evapotranspiration

Evapotranspiration is the measurement of water removed by evaporation and transpiration. Evapotranspiration was the only water out flux input used in simulations. Values for Galveston and Houston were taken from the Texas A&M System's AgriLife research (AgriLife Extension, 2005). Evapotranspiration values for Houston and Galveston were averaged to represent the Dickinson Bayou watershed which is located halfway between Houston and Galveston. Monthly values are displayed in Table 3.

Wastewater Flow

A typical system design of conventional septic systems with a soil absorption field in the Dickinson Bayou watershed assumes 70 gallons per person per day in a residential dwelling. The study site has a four person home using an OWTS with six drain lines. Therefore, each drain line would have 46.67 gallons per day. To find the flux flowing into the cross-sectional system, equal distribution of flow in the 75 foot long drainage pipe was assumed. Another assumption made was that the drainage pipe flow would be on average one third full. With these assumptions, wastewater flow into the simulated system was 7.26 cm/day out of the bottom third along the full length of the drainage pipe. Values were converted to metric as required by HYDRUS. To simulate this outflow throughout the day in an accurate manner, water distribution was based on a University of Wisconsin study that documented typical household water use throughout a day as shown in Figure 15 (University of Wisconsin, 1978).

Table 3. Evapotranspiration values in cm for each month in Galveston and Houston. These values were averaged for the Dickinson Bayou watershed which is located halfway between Galveston and Houston (AgriLife Extension, 2005)

City	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Galveston	5.59	6.60	10.41	12.70	15.52	16.76	15.75	15.24	13.97	10.67	7.11	5.84	136.17
Houston	5.99	7.19	10.97	12.73	15.52	16.69	16.56	15.44	14.15	10.87	7.37	5.97	139.45
Average	5.79	6.90	10.69	12.71	15.52	16.73	16.15	15.34	14.06	10.77	7.24	5.91	137.81

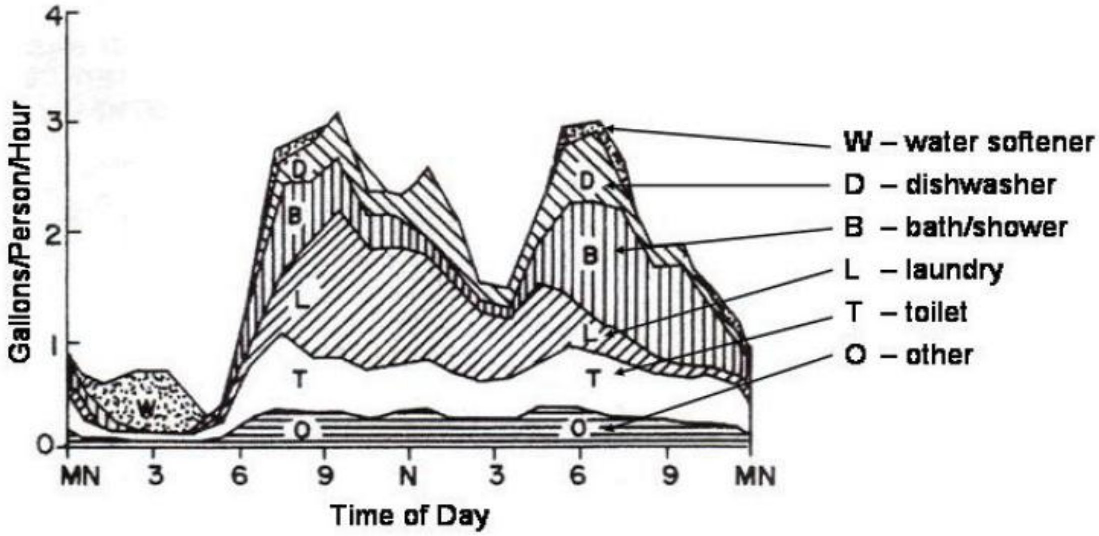

Figure 15. Distribution of wastewater coming from a typical residential home through the duration of one day (University of Wisconsin, 1978).

The daily outflow for systems in the Dickinson Bayou watershed was distributed throughout the day based on the percentage of daily use in Figure 15 to simulate the conventional and mound systems. Figure 16 below displays the calculated variable flux throughout a day for the conventional and mound systems in the study site.

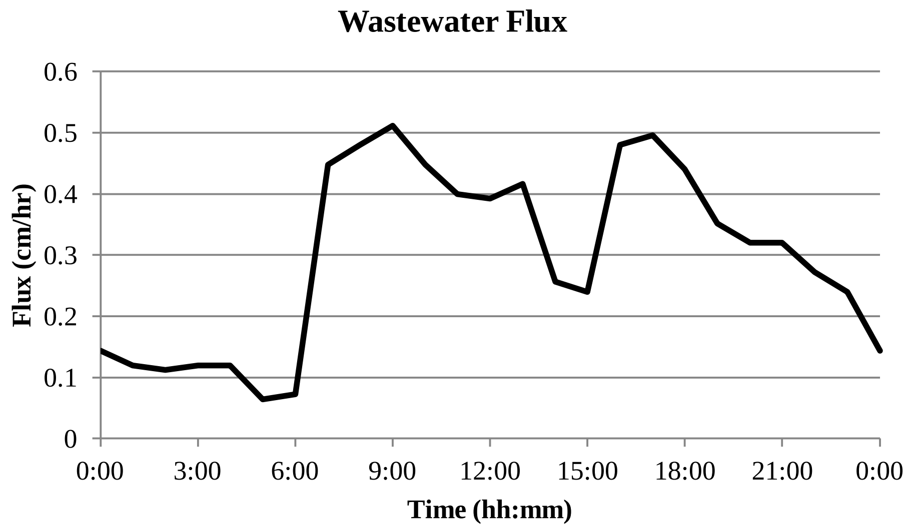

Figure 16. Wastewater flux used for conventional and mound system simulations.

The NSF/ANSI Standard 40/245 for the dispersal of ATU effluent specifies that the percentages of daily waste be distributed throughout the day as follows: 6:00-9:00, 35%; 11:00-14:00, 25%; and 17:00-20:00, 40% (NSF International, 2000). A total of 80 spray doses of effluent were used each day in the simulation, equally divided among the above distribution. Using the assumed 70 gallons per person per day with four people and the allowed 0.04 gallons per square foot per day, a dispersal area of 7,000 ft^2 was needed. Using the prescribed time distribution and application rate requirement, an application rate was created for the 7,000 ft^2 dispersal area (Table 4).

Table 4. Designed application rate for ATU system for the duration of one day.

Time	Doses	Daily Total %	Flow [g/hr]	Rate [cm/hr]
6:00	7	0.0875	24.5	0.0143
7:00	7	0.0875	24.5	0.0143
8:00	7	0.0875	24.5	0.0143
9:00	7	0.0875	24.5	0.0143
11:00	5	0.0625	17.5	0.0102
12:00	5	0.0625	17.5	0.0102
13:00	5	0.0625	17.5	0.0102
14:00	5	0.0625	17.5	0.0102
17:00	8	0.1000	28	0.0163
18:00	8	0.1000	28	0.0163
19:00	8	0.1000	28	0.0163
20:00	8	0.1000	28	0.0163
Total	80	1	280	0.1630

Soil Properties Used in HYDRUS

Soil properties required for HYDRUS simulations are listed in Table 5. The properties include residual water content (θ_r), saturated water content (θ_s), constants α and n, and saturated hydraulic conductivity (K_s). The constants α and n are empirical coefficients in the soil water retention function of the van Genuchten equation,

$$\boldsymbol{\theta}(\mathbf{h}) = \begin{cases} \boldsymbol{\theta}_\mathbf{r} + \frac{\boldsymbol{\theta}_\mathbf{s} - \boldsymbol{\theta}_\mathbf{r}}{[\mathbf{1}+|\boldsymbol{\alpha}\mathbf{h}|^\mathbf{n}]^\mathbf{m}} & \mathbf{h}<0 \\ \boldsymbol{\theta}_\mathbf{s} & \mathbf{h}\geq\mathbf{0} \end{cases} \qquad [1]$$

where

$$\boldsymbol{m} = \mathbf{1} - \frac{\mathbf{1}}{\boldsymbol{n}}, \boldsymbol{n} > 1 \qquad [2]$$

and where h is pressure the head. HYDRUS provides values for these parameters for the USDA soil types (Figure 2) by selecting a soil type in a drop-down menu. Values used for the van Genuchten equation were taken from (Carsel and Parrish, 1988). These values were chosen for loam and clay loam in all systems and for sand, and sandy loam and in the mound system. Other materials used in the simulation were geotextile fabric and gravel. Values for the geotextile fabric were found in a literature search (Morris, 2000). Saturated hydraulic conductivity for gravel was the only parameter found in a review of literature (Brassington, 1988). Sand properties were used for the remaining parameters. Sand is the soil type with the closest grain sizes and properties to gravel.

Table 5. Water transport properties for soils used in simulations.

Soil	Θr [-]	Θs [-]	Alpha	n	K_s [cm/hr]
Loam[a]	0.078	0.43	0.036	1.56	1.04
Clay Loam[a]	0.095	0.41	0.019	1.31	0.26
Gravel	0.045	0.43	0.145	2.68	114.2[b]
Geotext.	0.009[c]	0.224 [c]	0.008 [c]	1.92 [c]	0.648[d]
Sandy Loam[a]	0.065	0.41	0.075	1.89	4.42083
Sand[a]	0.045	0.43	0.145	2.68	29.7

[a](Carsel and Parrish, 1988), [b](Brassington, 1988), [c](Morris, 2000), [d](Williams and Abouzakhm, 1989)

Solute Transport

Solute transport in HYDRUS-2D is based on the Fick's Law equations for advection-dispersion (Šimunek et al., 2011). Interactions between solid and liquid phases are described by nonlinear non-equilibrium equations. Solutes are assumed to be transported by convection and dispersion in the liquid phase. HYDRUS-2D assumes non-equilibrium interactions between the solution and adsorbed concentration.

HYDRUS-2D has three options for transport: equilibrium, chemical non-equilibrium, and physical non-equilibrium. Simulations for this project used chemical non-equilibrium transport of solute. Solute flow is described by two mechanisms: movement and accumulation. Movement refers to transport in the soil matrix, and accumulation refers to the increase in solute mass in that matrix. During equilibrium no fluid movement takes place because hydraulic heads are static and concentrations are in equilibrium to prevent diffusive flow of mass. However, soils are dynamic systems and are rarely in equilibrium, which is why equilibrium transport was not used. Changes in physical, chemical, and atmospheric conditions continually occur. However, modeling often considers equilibrium transport to limit complications. This assumption can be justified as changes in the unsaturated zone are quickly minimized as the system approaches equilibrium (Tindall et al., 1999). For this assumption, sorption processes would be instantaneous which would not be accurate for this modeling scenario.

Physical non-equilibrium is based on two-region, dual-porosity transport. The two-region concept assumes the liquid phase occurs in mobile and immobile regions. The mobile-immobile regions are caused mainly by macropores. Macropores are channels that often occur in soil profiles from root systems, earthworms, and cracks from freeze-thaw or drying soils (Mawdsley et al., 1995). These macropores increase microbial transport, but the extent of this change is not well understood (Pachepsky et al., 2006). Further, not enough is known about soil structures in the Dickinson Bayou watershed to model physical non-equilibrium, the reason why this option was not used.

For chemical non-equilibrium reactions, HYDRUS considers a two-site sorption model. Type-1 sites are equilibrium sites while type-2 sites are based on a first-order kinetic rate process. Fraction *f* is the amount of sites in equilibrium with the solution phase. Chemical non-equilibrium was chosen for solute transport in this project. HYDRUS recommends the chemical non-equilibrium process based on attachment-detachment for bacteria transport. Therefore, *f* is set to zero so that all sites follow first-order kinetic processes instead of instantaneous sorption. The attachment-detachment model is based on the convection-dispersion equation

$$\boldsymbol{\rho \frac{\partial s}{\partial t} = \Theta k_a \Psi c - k_d \rho s} \qquad [3]$$

where ρ is bulk density [ML^{-3}], s is concentration of solute in the solid phase [#/M], t is time [T], Θ is water content [-], k_a is the first-order deposition (attachment) coefficient

$[T^{-1}]$, Ψ is the dimensionless colloid retention function [-], c is the concentration in the liquid phase $[\#/L^{-3}]$, and k_d is the first-order entrainment (detachment) coefficient $[T^{-1}]$.

Parameters for solute transport were taken from the literature due to the lack of field data for this project. Bulk density values were taken from the NRCS Bulk Density Fact Sheet and are displayed in Table 6 below (National Resource Conservation Service, 1996). A study by (Bradford et al., 2006) using sand columns found the following values based on 710 μm sized sand: longitudinal dispersivity (α_L) 0.486 cm, attachment (k_a) 6.6816 #/d, detachment (k_d) 0.4608 #/d, and maximum amount of solute per site (S_{max}) 1000 1/g. These values were used for all soil types but the gravel and geotextile fabric. For these materials, k_a and k_d were zero while S_{max} was 10. These values were used based on the assumption that *E. coli* would travel freely throughout the gravel drainage pit due to high concentrations, low attachment, and limited surface area. These values are presented in Table 6.

Table 6. Solute transport properties for each soil type in simulations.

Soil	ρ [g/cm³]	α_L [cm]	α_T [cm]	k_a [#/d]	k_d [#/d]	S_{max} [-]
Loam	1.7	0.486	0.0486	6.6816	0.4608	1000
Clay Loam	1.65	0.486	0.0486	6.6816	0.4608	1000
Gravel	2	0.486	0.0486	0.0	0.0	10
Sandy Loam	1.75	0.486	0.0486	6.6816	0.4608	1000
Sand	1.8	0.486	0.0486	6.6816	0.4608	1000

The values for dispersivity and diffusion affect dispersion. Dispersion is solute movement caused by mixing and molecular diffusion. Increased dispersion increases the spread of solute. Dispersion values are given as longitudinal (along flow path) and transverse (perpendicular to flow). The value for transverse dispersivity is assumed to be one tenth of the corresponding longitudinal dispersivity (Pang et al., 2006). Molecular diffusion is the movement of molecules due to kinetic activity in their concentration gradient. The values are related in the following equation

$$D_L = \alpha_L {*} v + D^* \qquad [4]$$

where *v* is pore velocity (Freeze and Cherry, 1979). The diffusion coefficient was found in a study by (Budrene and Berg, 1991) and equals 0.41472 cm^2/d. The final parameters used represent bacterial decay in the liquid and solid phase. A (Pang et al., 2004) study found these values for *E. coli* to be 0.193 and 3.53 #/d, respectively. These parameters for *E. coli* are displayed in Table 7. Microbial growth was not considered in these simulations based on findings from (Pang et al., 2004) that groundwater temperatures are not favorable for bacterial growth.

Table 7. *E. coli* diffusion and decay parameters.

D* [cm^2/s]	**μ_L [#/d]**	**μ_S [#/d]**
0.41472	0.193	3.53

Artificial Dispersion

Attachment and dispersion parameters were taken from column experiments; however, research has found that dispersion is much higher for field scale than for column studies (Freeze and Cherry, 1979). To account for this finding, HYDRUS provides a modeling option for artificial dispersion. HYDRUS uses what is termed as the “stability criterion” that is the product of the Courant and Peclet numbers. A metric to determine if the solute transport simulation is stable is that the product of the Courant and Peclet numbers, the performance index, is less than or equal to two. Finding an accurate value for the stability criterion, however, is another obstacle. Not enough information is known to calculate the Peclet and Courant numbers independently of the simulations, so resulting values of these parameters from the initial simulation were adjusted to find a stability criterion. Through iterations, a stability criterion of 0.096 brings the performance index to equal two.

Sensitivity Analysis

A sensitivity analysis was done to show the effects of critical parameters and the range of possible results because parameters for simulations are not based on field data and cannot be calibrated or validated. The sensitivity analysis focused on parameters used in the solute transport function of HYDRUS. These values are subject to the most uncertainty because they were taken from studies done on sand column experiments and they vary for different soil types and solutes. The water transport inputs and parameters have more certainty in that they were based on field conditions instead of only on a review of literature. The sensitivity analysis was done by evaluating the maximum concentration found in the middle of the clay loam layer below the gravel drain field. Each parameter was changed individually by a factor of 0.5, 0.9, 1.1, and 2.0, and the resulting values for maximum concentration were plotted to find a trend. Additional factors were induced if a strong trend was not created with initial factors. The parameters selected are longitudinal and transverse dispersivity, diffusion coefficient, attachment and detachment rate, and S_{max}, maximum amount of contaminant on sorption sites.

CHAPTER III

RESULTS AND DISCUSSIONS

Water Flow

Water flow results are shown as pressure head values found from HYDRUS simulations. Negative pressure heads represent unsaturated soils, positive pressure heads represent saturated soils, and a pressure head equal to zero is where the water table is located.

Precipitation

Simulations of average yearly rainfall, heavy year of rain fall, and low year of rainfall were all run for a conventional septic system with a soil absorption field. Figure 17 through Figure 22 displays the pressure head values from observation nodes located at the surface of the soil and at the initial water table resulting from simulations based on the average, heavy, and low years of rainfall. These results display how the system reacts to varying amounts of rainfall.

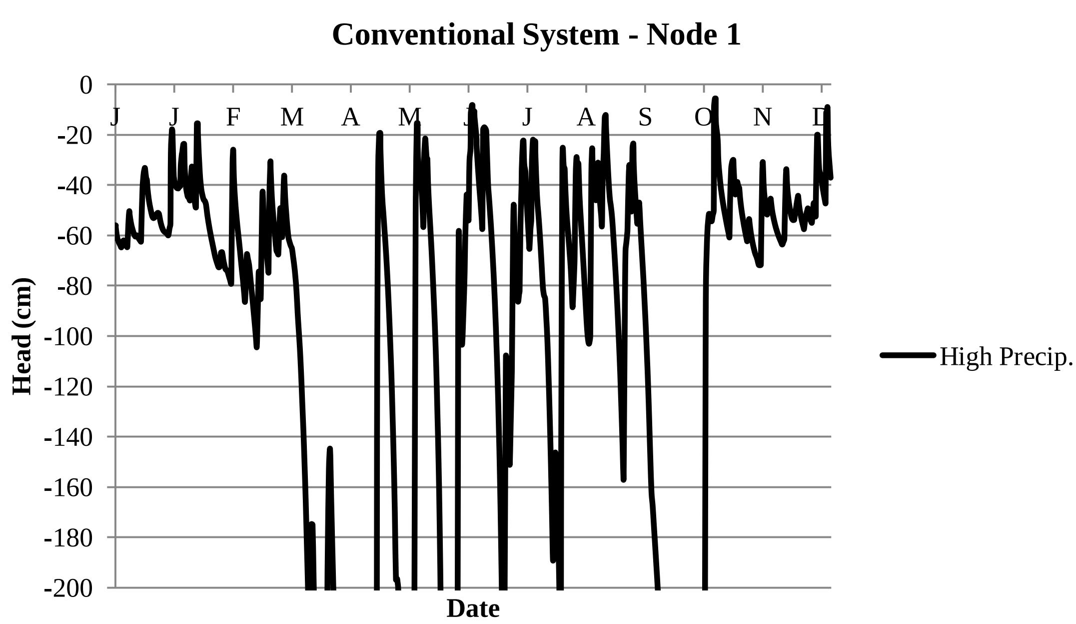

Figure 17. Pressure head values at the surface of the conventional system under a year of heavy rainfall.

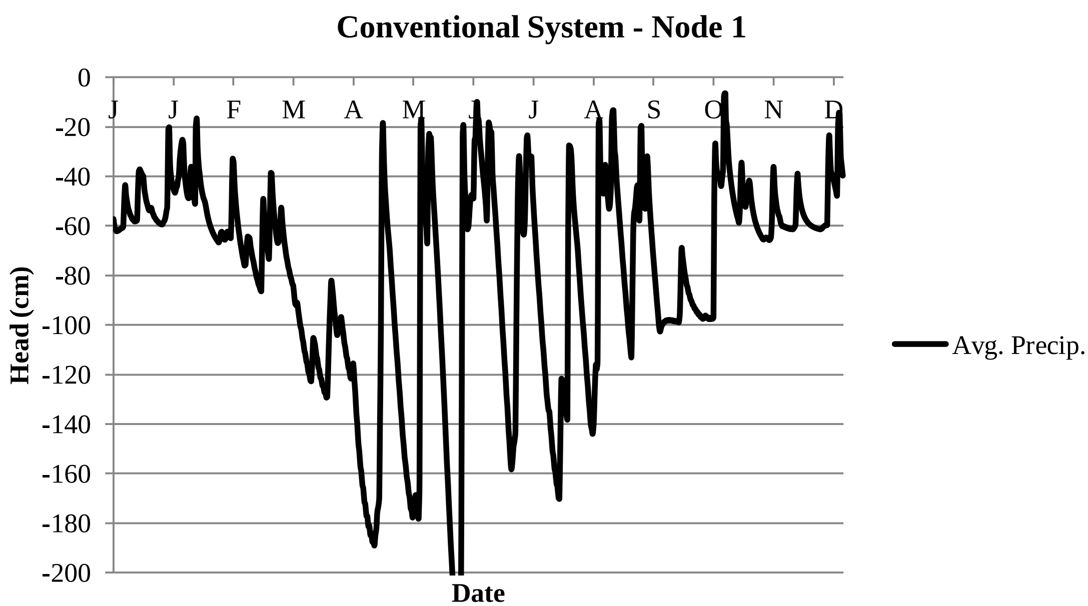

Figure 18. Pressure head values at the surface of the conventional system under an average year of rainfall.

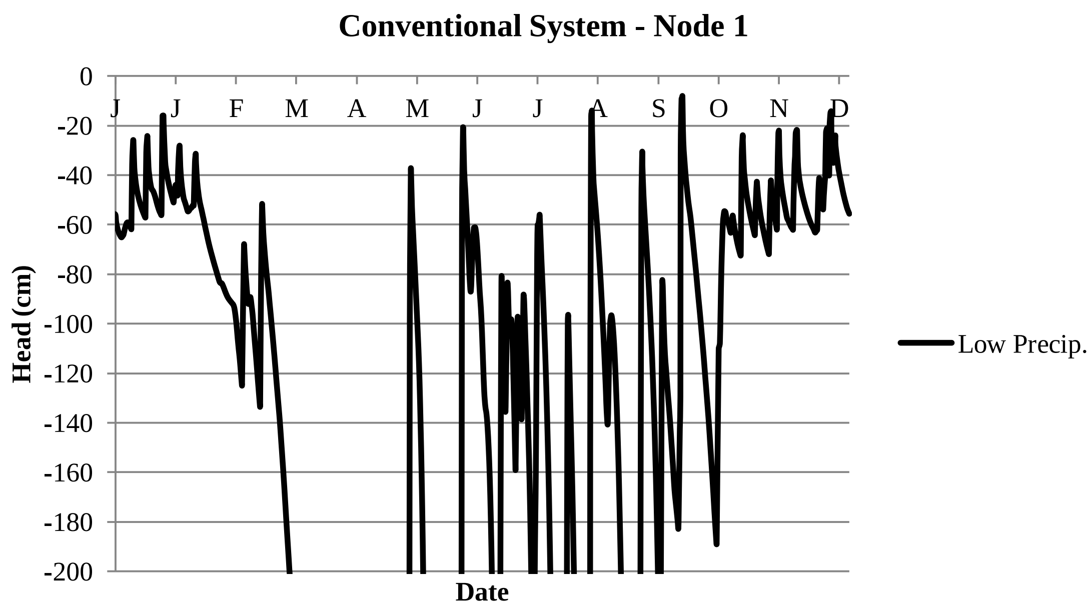

Figure 19. Pressure head values at the surface of the conventional system under a year of low rainfall.

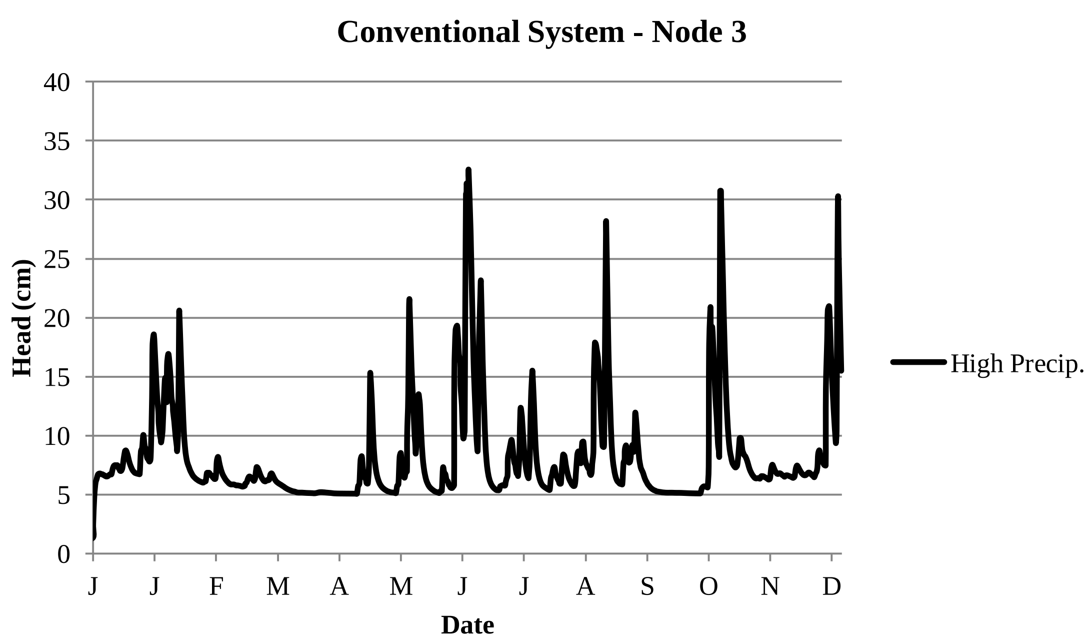

Figure 20. Pressure head values at the initial depth of the water table of the conventional system under a year of heavy rainfall.

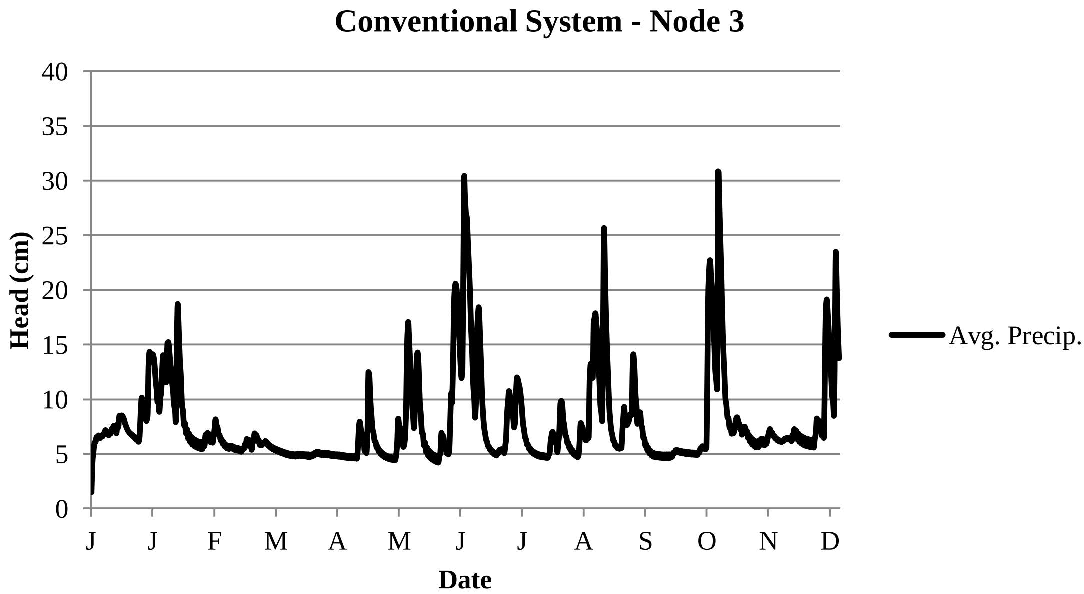

Figure 21. Pressure head values at the initial depth of the water table of the conventional system under an average year of rainfall.

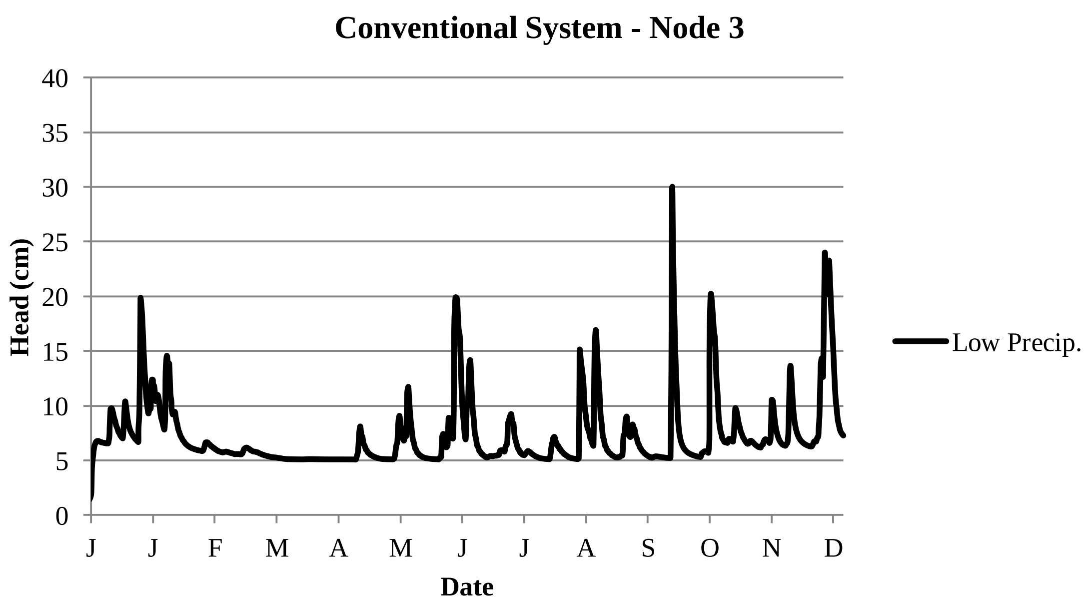

Figure 22. Pressure head values at the initial depth of the water table of the conventional system under a year of low rainfall

The difference between precipitation events is primarily displayed through the amount and severity of peaks in pressure head. The typical pressure head level remained nearly the same for each simulation. The changes in pressure head are associated with rainfall events. Both the average year and year of heavy rainfall had more rainfall events than the year of low precipitation. The peaks in pressure head values were higher in the year of heavy rainfall than for the other years. The severity of the peaks is most related to system failure. Higher peak values lead to saturated systems and effluent reaching the surface. Based on these results, years with higher levels of precipitation will cause an increase in system failures. The remaining simulations and results are based on an average of yearly rainfall.

Soil Types

To show the effect of clay on water and solute transport, system profiles were simulated with different soil profiles. One simulation replaced the clay loam layers in the initial simulation to be loam so that the entire profile was loam. The other simulation replaced the clay loam layers with clay. These alternative simulations were compared to the simulation of the initial setup of a typical conventional system located in the Dickinson Bayou watershed.

Pressure head results in Node 2 located below the drainage pipe show that increasing clay content increases failure occurrences with the drainage pipe becoming saturated (Figure 23, Figure 24, and Figure 25). The simulation with all loam soil only experienced saturation at Node 2 one time while the initial system and system with clay had three and four occurrences, respectively. These results show the negative hydraulic effects clay has on OWTSs and why conventional septic tanks for with soil absorption fields are not to be installed in clay soils. However, these results also show that the system failed with all loam soil, also. The reason for this failure could be due to the high water table.

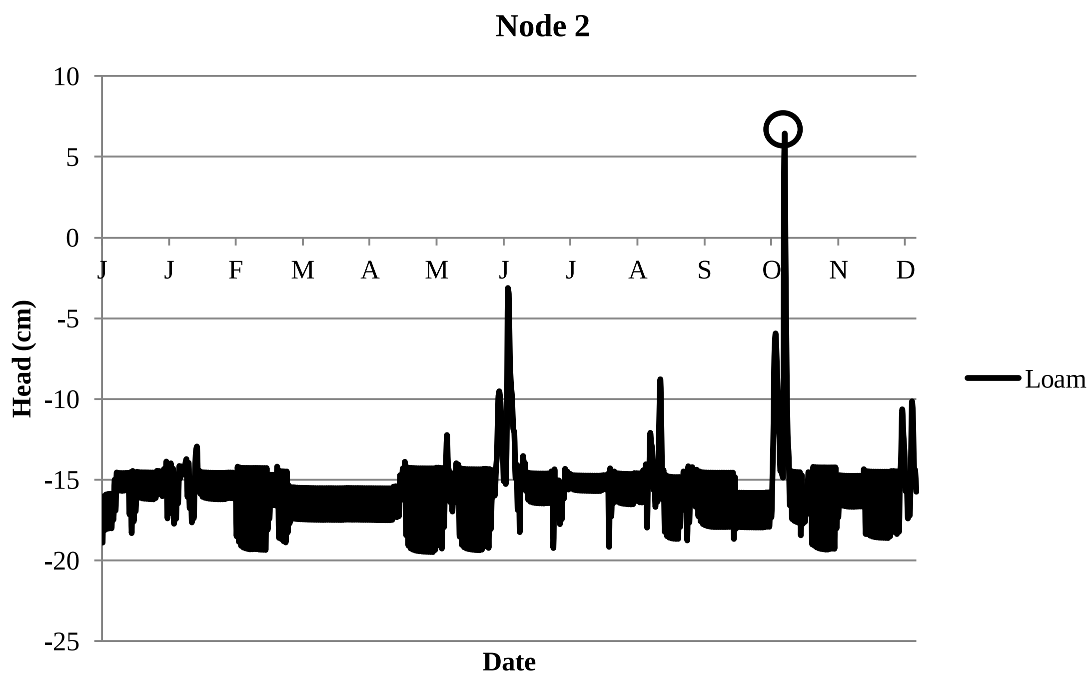

Figure 23. Pressure head values at Node 2 in the conventional system with all loam soil.

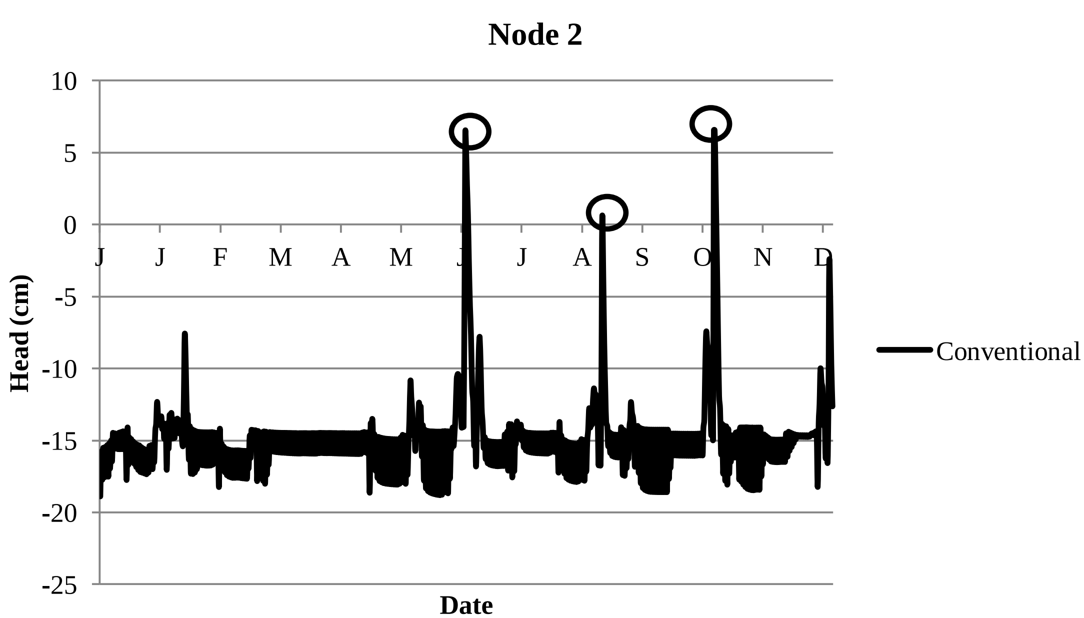

Figure 24. Pressure head values at Node 2 in the conventional system.

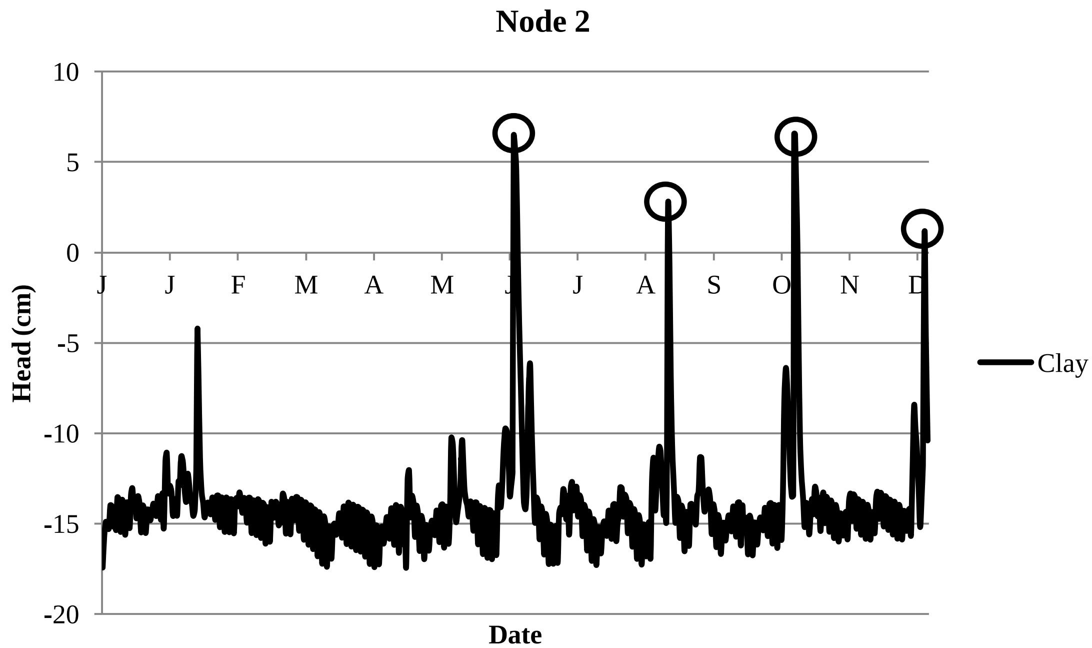

Figure 25. Pressure head values at Node 2 in the conventional system with clay soil.

Results at Node 3 also display that increased clay content increases pressure head levels. Node 3's depth below the water table increased in simulations with higher clay content. Figure 26, Figure 27, and Figure 28 display these results.

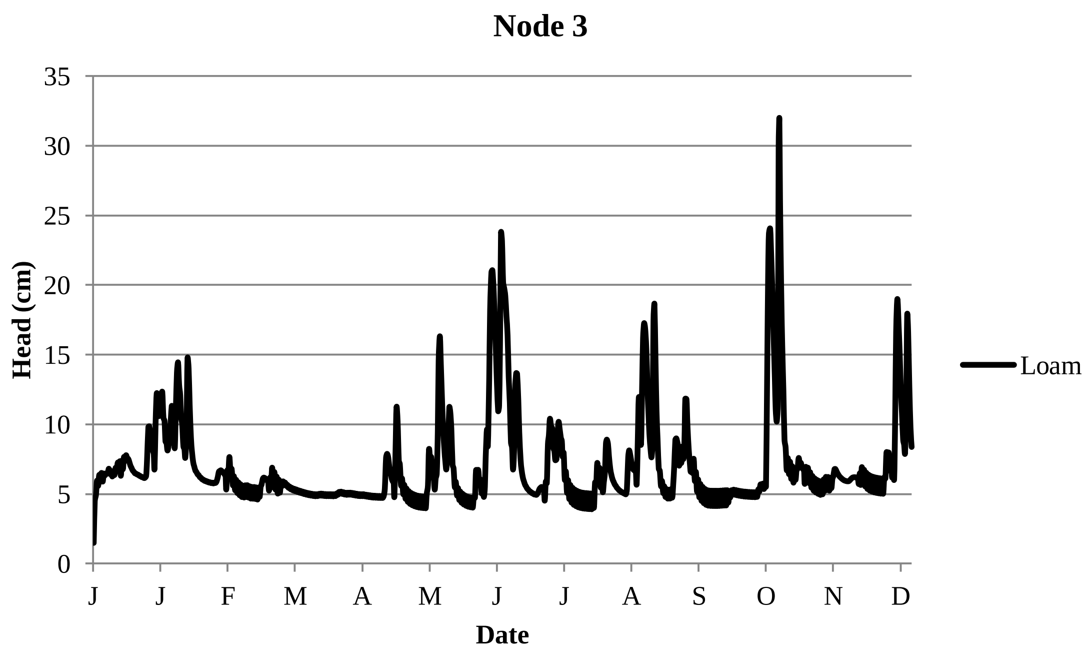

Figure 26. Pressure head values at Node 3 in the conventional system with all loam soil.

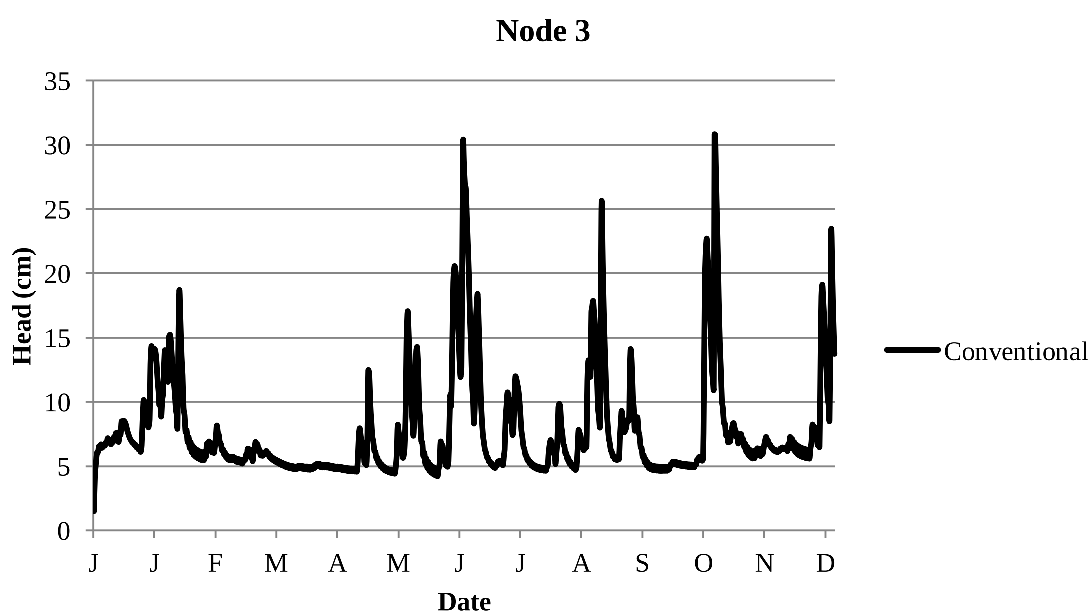

Figure 27. Pressure head values at Node 3 in the conventional system.

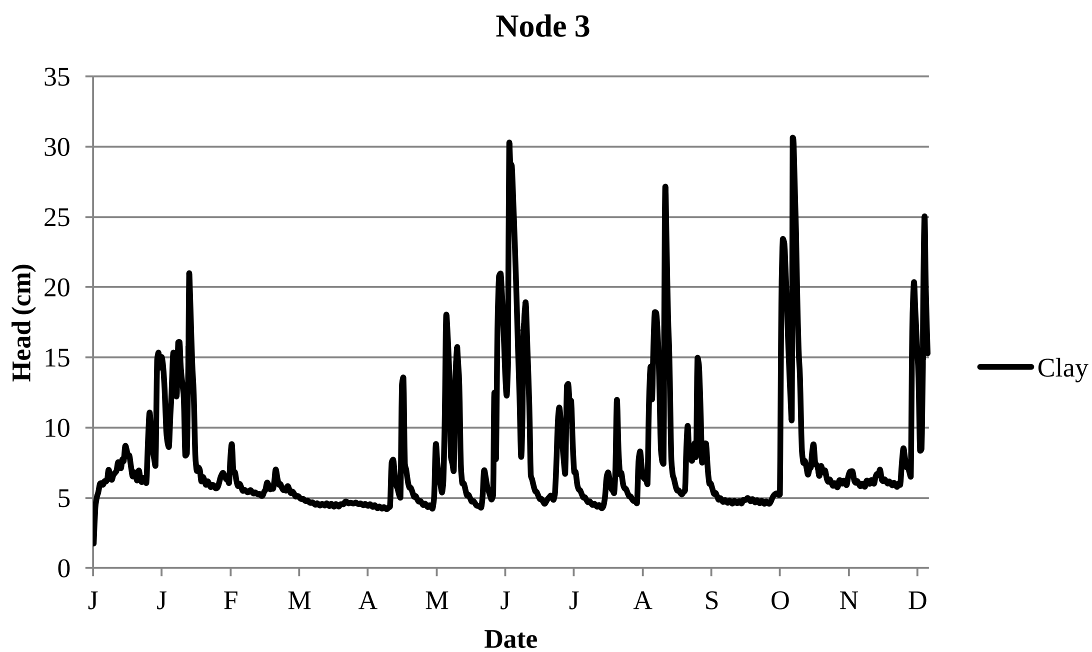

Figure 28. Pressure head values at Node 3 in the conventional system with clay soil.

The results show that clay is detrimental to soil treatment for OWTSs. As seen in Figure 23, Figure 24, and Figure 25, failure occurrences are more prevalent with increasing clay content.

Conventional vs. Mound

This section compares the results in pressure head values between the conventional system and the mound system. As seen in Figure 29, the pressure head at the node located below the drain pipe for the mound system remained within a constant range throughout the year with values ranging between -20 and -15 from daily outflow variations. The pressure head at Node 2 for the mound system never become positive meaning that saturated conditions never existed directly below the pipe. The pressure head for the conventional system shows a similar range for most of the year but had several peaks in pressure head (Figure 30). These peaks cause concern as three peaks reach positive pressure head values, circled in Figure 30. A positive pressure head at Node 2 means that the water table has risen to the drainage pipe. Under this condition, effluent is flowing directly into the water table without treatment, and the drainage field is saturated.

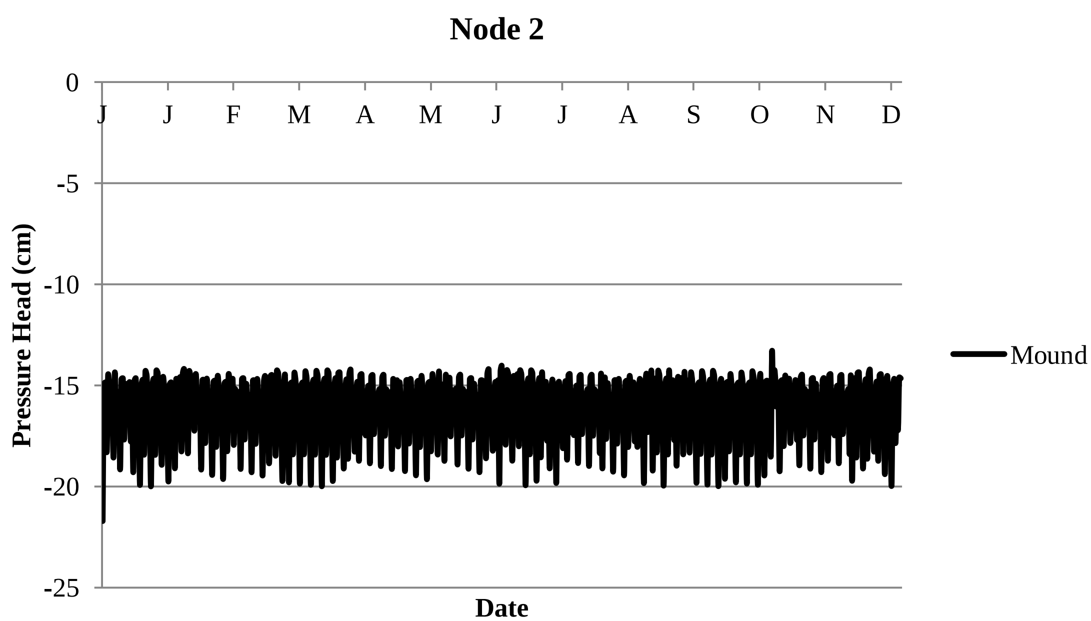

Figure 29. Pressure head values at Node 2 located below drainage pipe for the mound system.

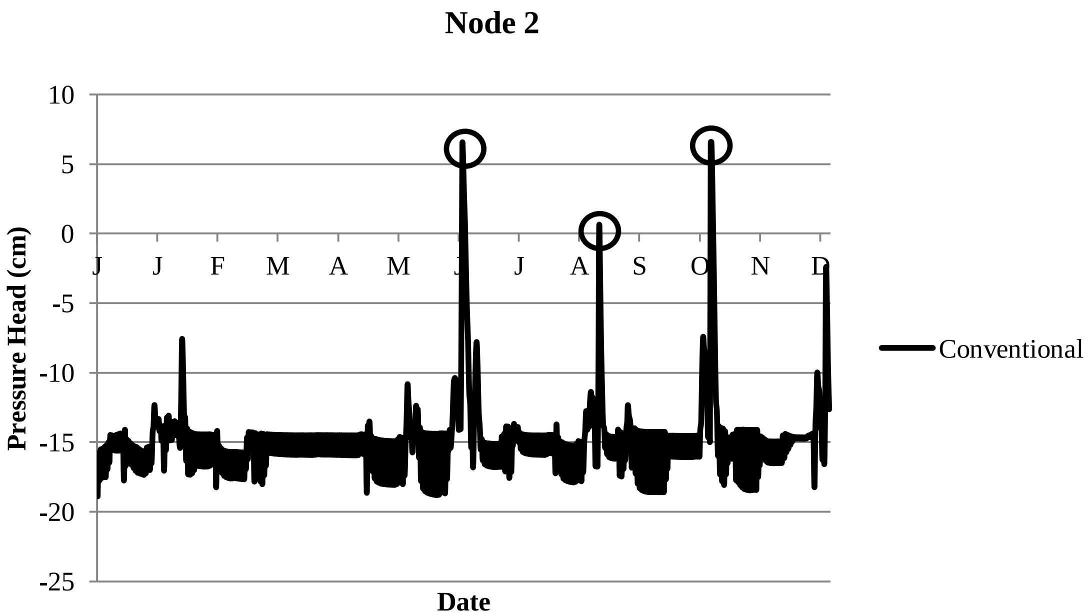

Figure 30. Pressure head values at Node 2 located below drainage pipe for the conventional system.

At Node 3, the pressure heads for the conventional system vary much more than those of the mound system. The mound system displays a consistent water table level around six centimeters above Node 3 and only rises above 10 centimeters from Node 3 in five occurrences (Figure 31). The conventional system does not display a consistent water table level but fluctuates often (Figure 32). The head for the conventional system does spend much of the year below the head of the mound system, but the peaks for the conventional system are much higher than the mound's with only one exception. A possible explanation for the water table becoming lower in the conventional system than the mound system between rainfall events is that evapotranspiration can have a greater effect on the water table in the conventional system because the water table in the conventional system is closer to the surface.

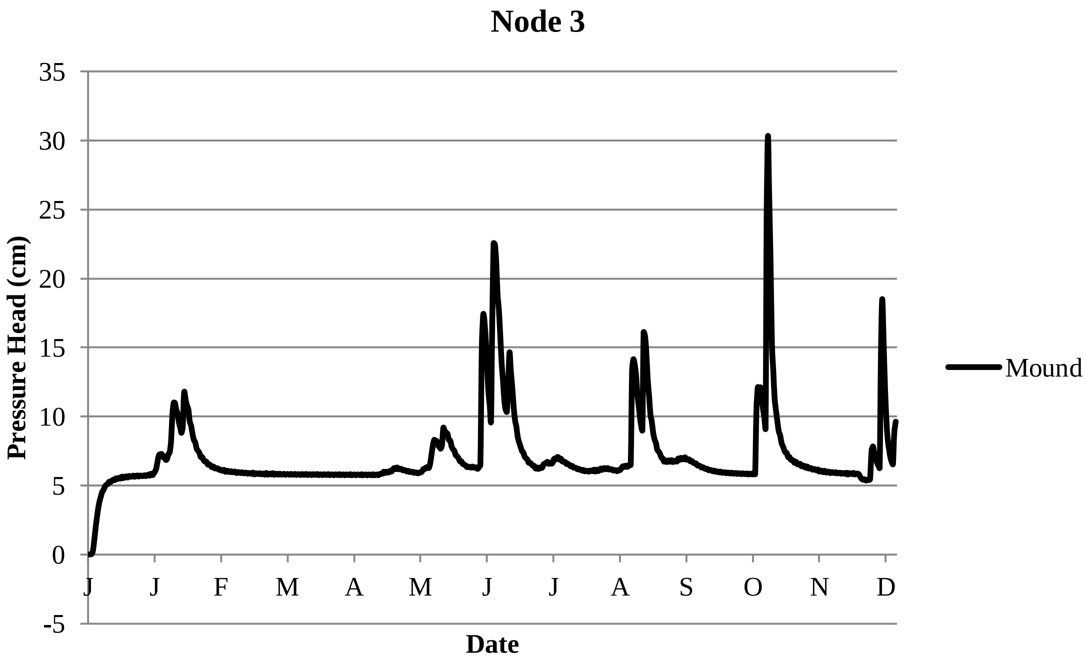

Figure 31. Pressure head values at Node 3 of the mound system.

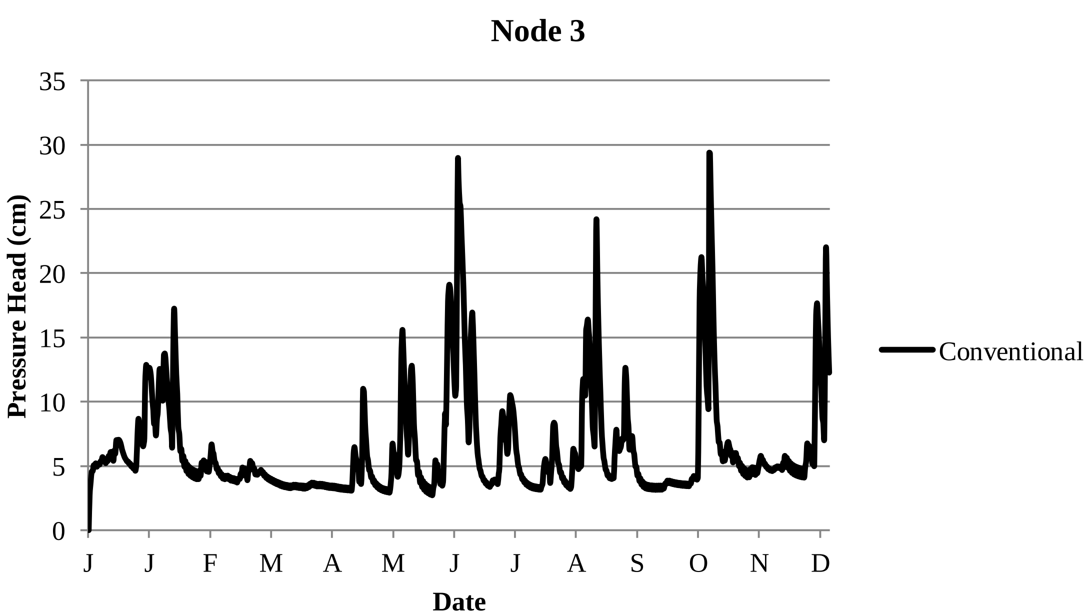

Figure 32. Pressure head values at Node 3 of the conventional system.

The results display that the mound system lessens the impact of rainfall on the drain field. During non-peak times, the pressure heads from the mound system are close to those of the conventional system. However, during rainfall events the pressure head increases in the conventional system are much more dramatic than those for the mound system. The mound system often experiences no peaks due to rainfall. A clear example of this distinction occurs in Node 2 in Figure 29 and Figure 30. The pressure head remains fairly constant for the mound system but varies greatly in the conventional system. Results at this node also show that the water table rises up to the drain line. Under this condition soil treatment of the effluent is severely decreased. System saturation reduces attachment capabilities of the soil and decreases retention times. With the mound system, the water table never comes near the drain line. These results display that a mound system will improve hydraulic and treatment conditions in the studied area. Problems associated with the high water table and clayey soils are mitigated with the added soil layers.

Conventional vs. ATU

This following section compares the results in pressure head values between the conventional system and the ATU system. The variation for pressure head in Node 2 is much greater in the ATU system. While the head remains around -15 for most of the year for the conventional system, the head for the ATU system constantly fluctuates and ranges from -25 to 25 cm (Figure 33). The peak values for the ATU are much lower and much higher than those for the conventional system. Figure 34 displays the values of pressure head at the node located the depth below the drain pipe in the conventional system. The reason that the pressure head values at Node 2 for the ATU system become less than those for the conventional system is that the ATU system does not have a constant influx of effluent occurring above the node. A possible explanation for why the pressure head values for the ATU system become greater than those for the conventional system is that the effects of evapotranspiration can be negated by a constant spray distribution. The constant spray distribution decreases the soil storage capacity that would be created by evapotranspiration, and with this decreased storage, rainfall events would have a greater impact on pressure heads.

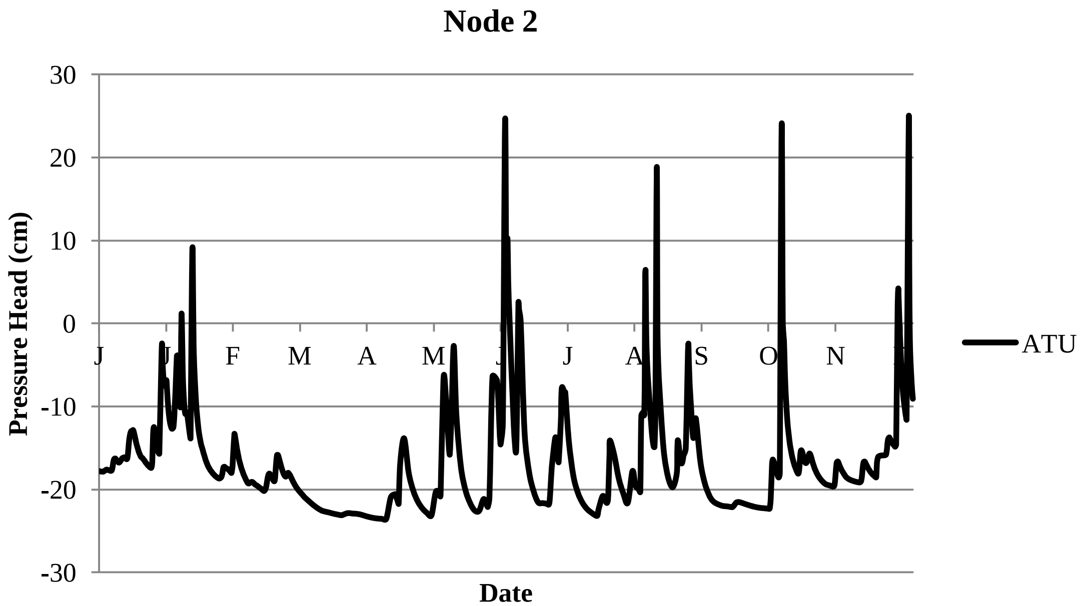

Figure 33. Pressure head values for Node 2 in the ATU system.

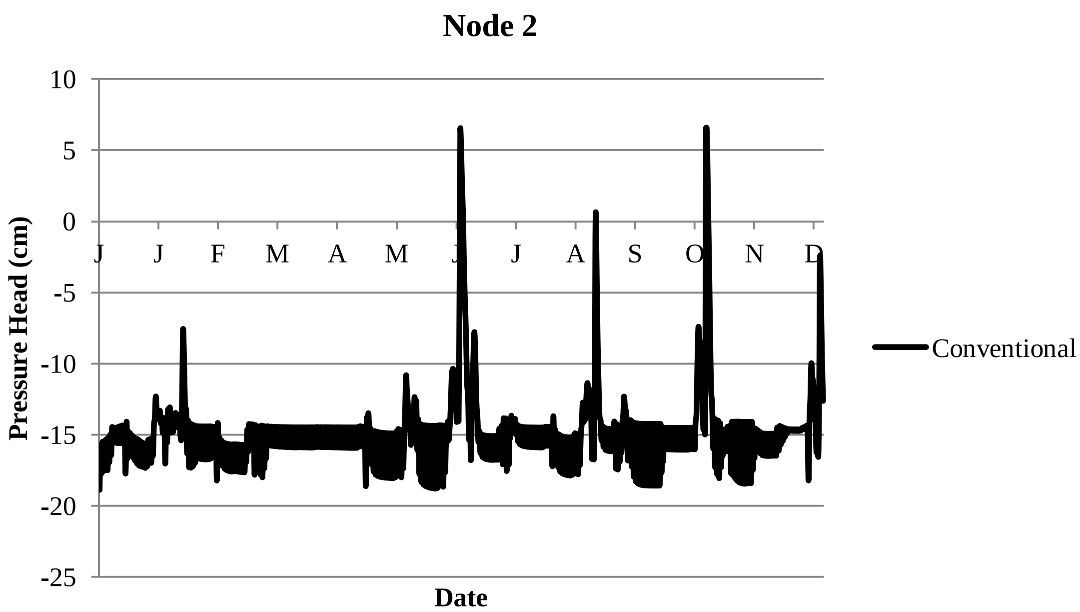

Figure 34. Pressure head values for Node 2 in the conventional system.

The water table pressure head results in Figure 36 for the conventional and ATU systems display similar patterns with the ATU remaining about 5 cm below the conventional system for almost all of the year. However, the peaks associated with rainfall are much more dramatic for ATU than for conventional. While the head is typically 5 cm below the conventional system, ATU spikes go above conventional peaks in several occurrences.

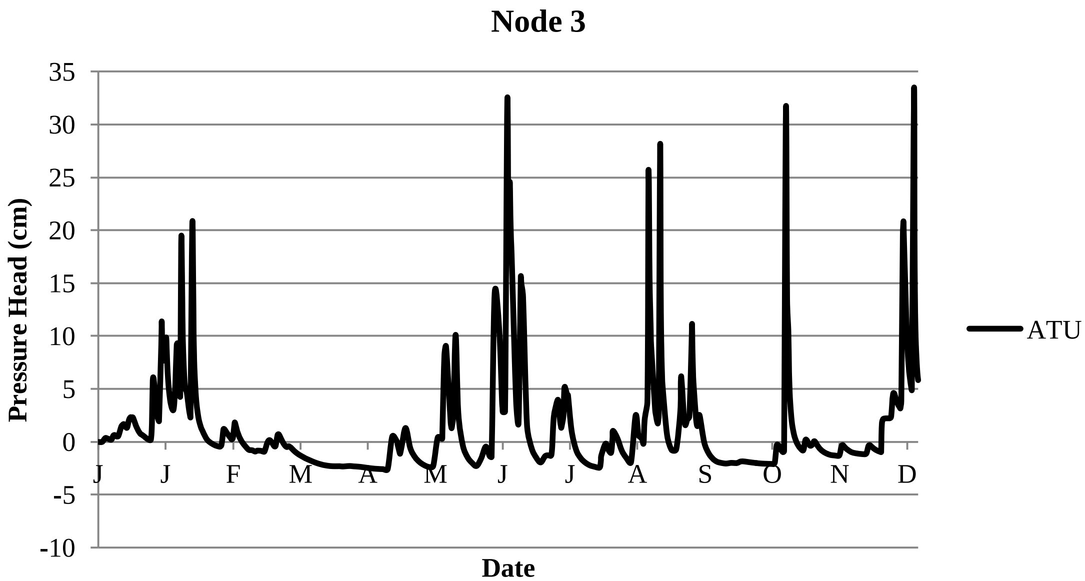

Figure 35. Pressure head values for Node 3 in the ATU system.

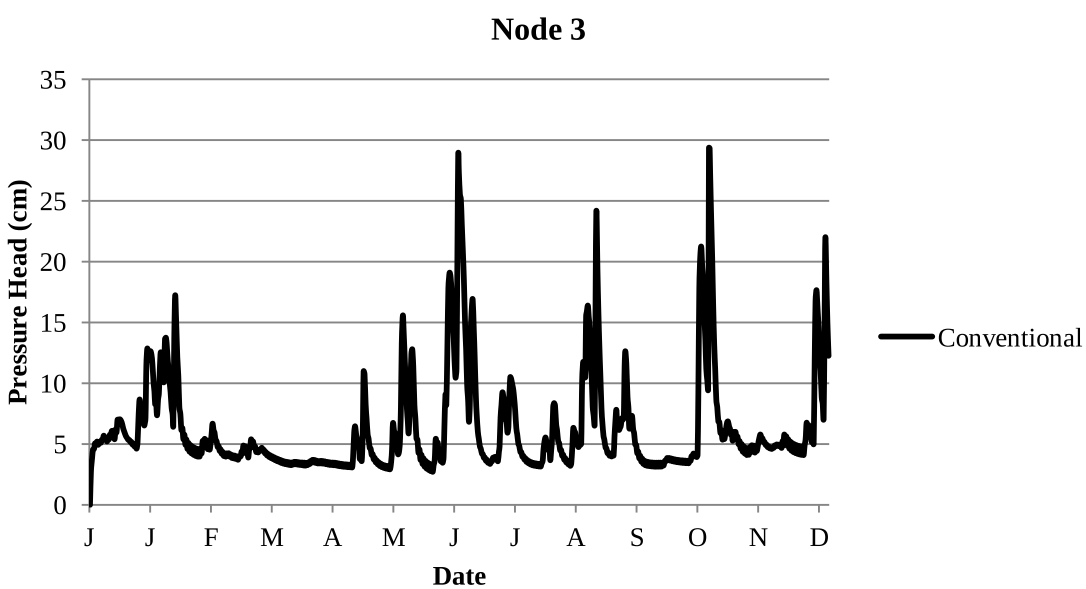

Figure 36. Pressure head values for Node 3 in the conventional system.

Pressure head results at the surface show that the pressure head for the ATU system approaches zero in two cases while the conventional system reaches a peak at -5 cm (Figure 37. Pressure head values for Node 1 of the ATU system and Figure 38). With the ATU system, the surface becomes fully saturated. Under these conditions in the ATU system, soil treatment of *E. coli* is severely limited.

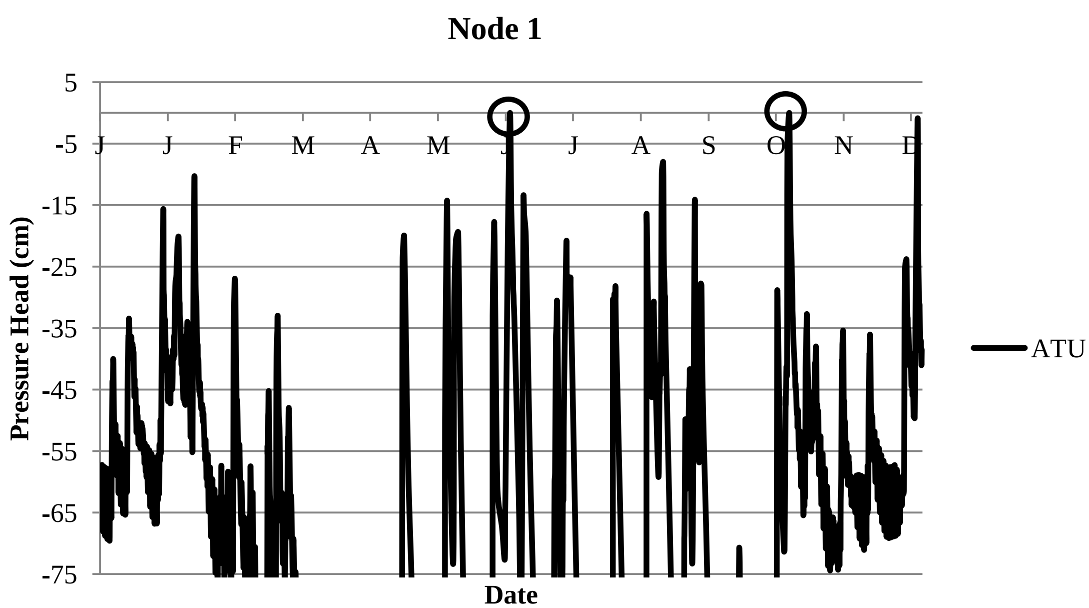

Figure 37. Pressure head values for Node 1 of the ATU system

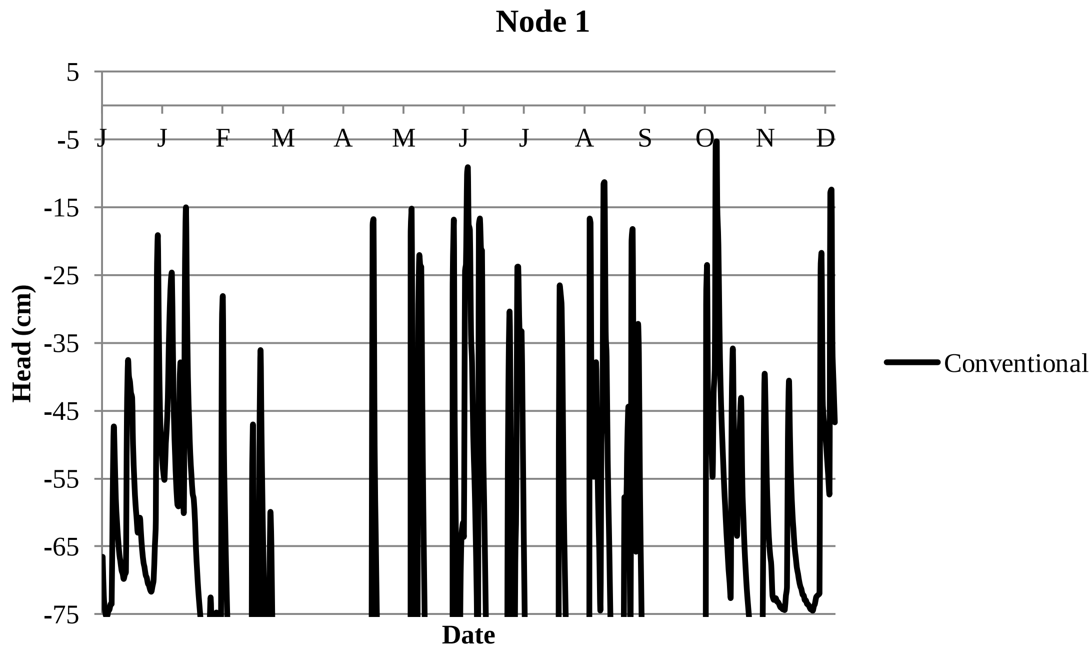

Figure 38. Pressure head values for Node 1 of the conventional system.

Pressure head levels are consistently lower for ATUs than with conventional systems. However, peaks associated with heavy rainfall are higher with ATUs. The highest water table levels are associated with ATUs. Also, in two instances the surface becomes fully saturated, which does not once occur in the other systems (Figure 37. Pressure head values for Node 1 of the ATU system). A possible explanation for these results is that the flux out of the system through evapotranspiration is negated from the surface application of the spray distribution system. The top of the soil profile is not able to achieve the same storage capacity that the conventional and mound system are able to achieve. The effects of precipitation are then magnified. ATUs do not solve hydraulic problems associated with conventional systems; however, ATU treatment processes operate differently and do not depend on soil properties.

Solute Transport

Solute transport results are discussed more qualitatively than quantitatively because the solute transport parameters are taken from a review of literature and are not associated with a field study. Concentrations should not be taken as accurately representing field conditions, but are used for comparison to display the effectiveness of each system. Differences in *E. coli* removal based on system design are shown because the same solute transport parameters were used for each system.

The solute transport for an effective ATU system is not a concern because a fully effective ATU system will only release 2 count/ml in its effluent. However, the concern for ATUs is improper maintenance leading to not fully functioning ATUs. Solute transport simulations for ATUs were also run to represent these not fully functioning systems by using the same effluent concentration as the conventional and mound systems, 12,000 count/ml. Figure 39, Figure 40, and Figure 41 display concentration results for Node 1. Figure 42, Figure 43, and Figure 44 display concentration results at Node 3.

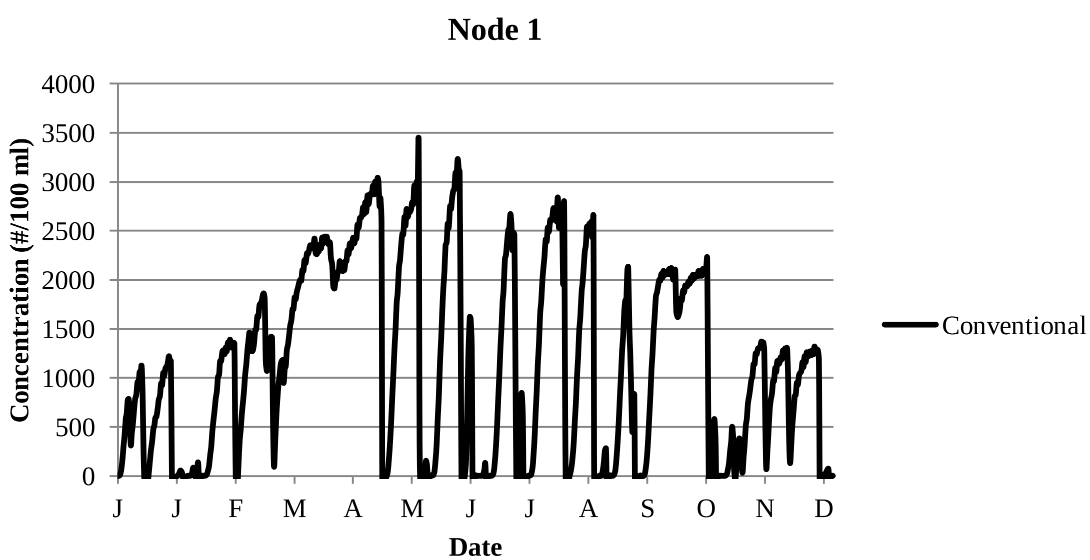

Figure 39. ***E. coli*** **concentrations at Node 1 of the conventional system.**

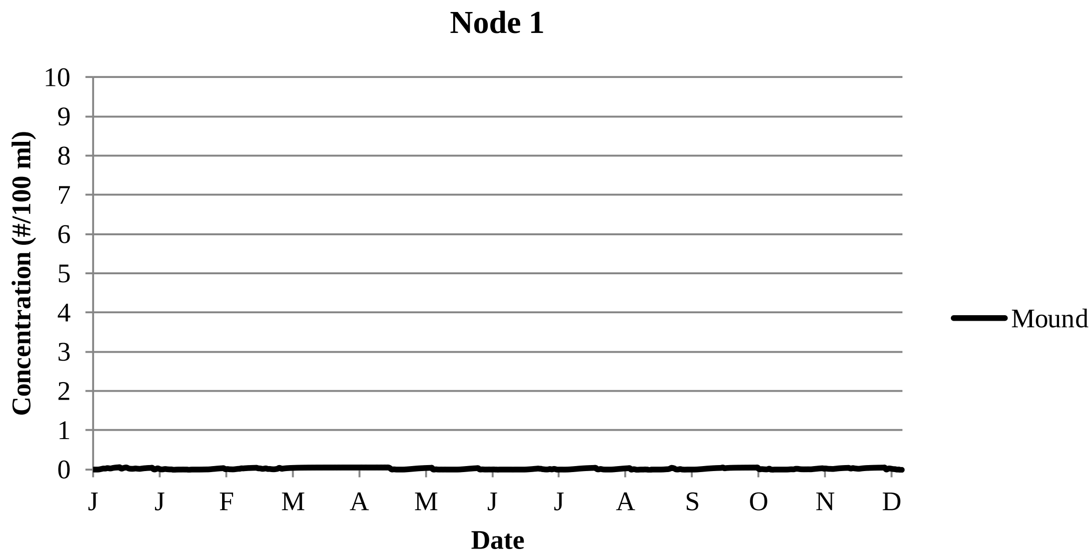

Figure 40. ***E. coli*** **concentrations at Node 1 of the mound system.**

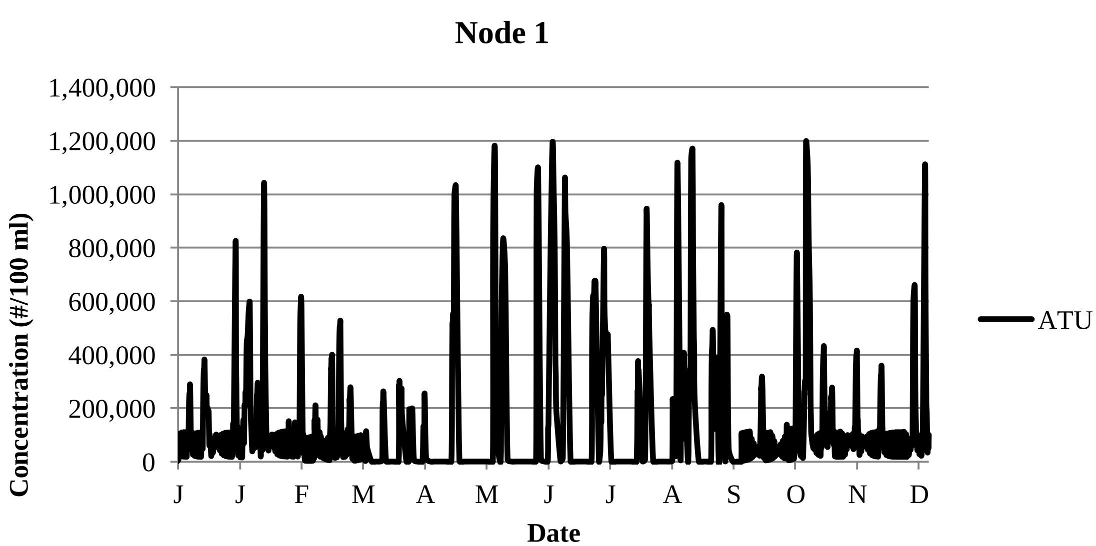

Figure 41. ***E. coli*** **concentrations at Node 1 of the ATU system.**

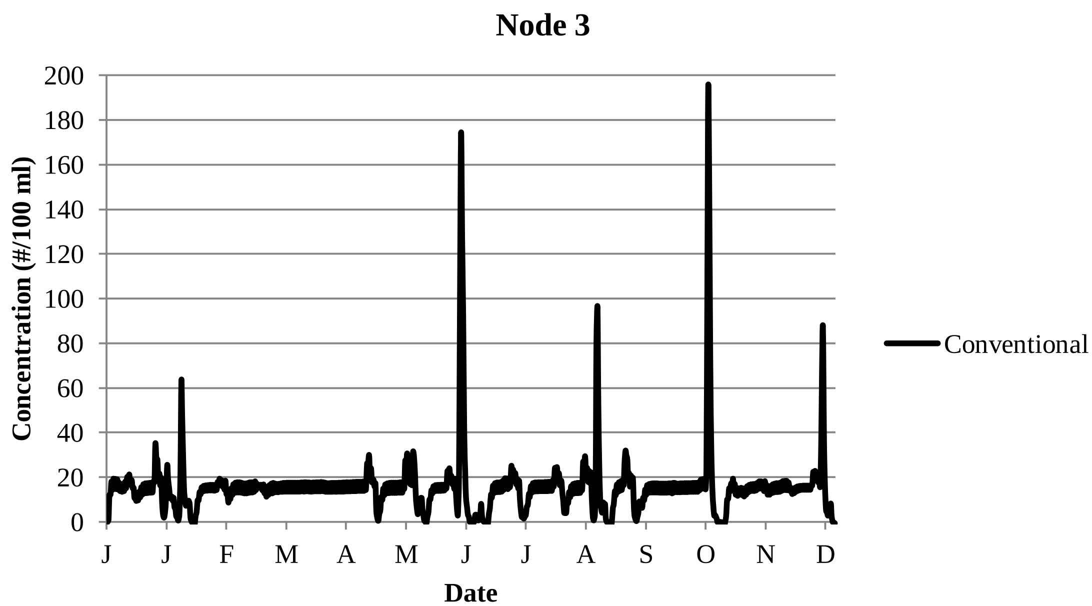

Figure 42. ***E. coli*** **concentrations at Node 3 of the conventional system.**

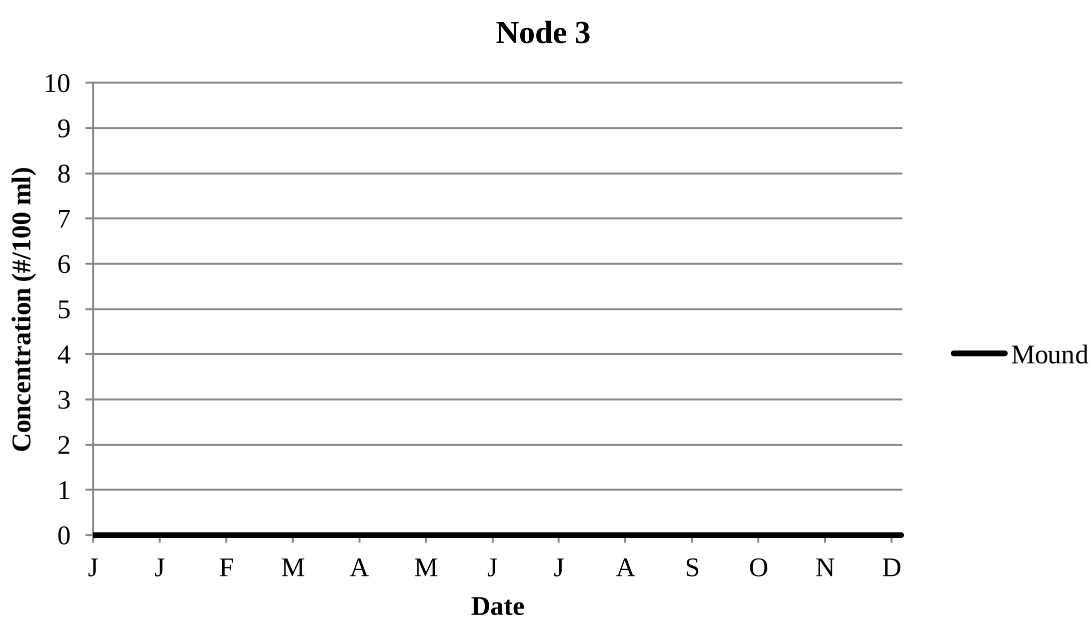

Figure 43. ***E. coli*** **concentrations at Node 3 of the mound system.**

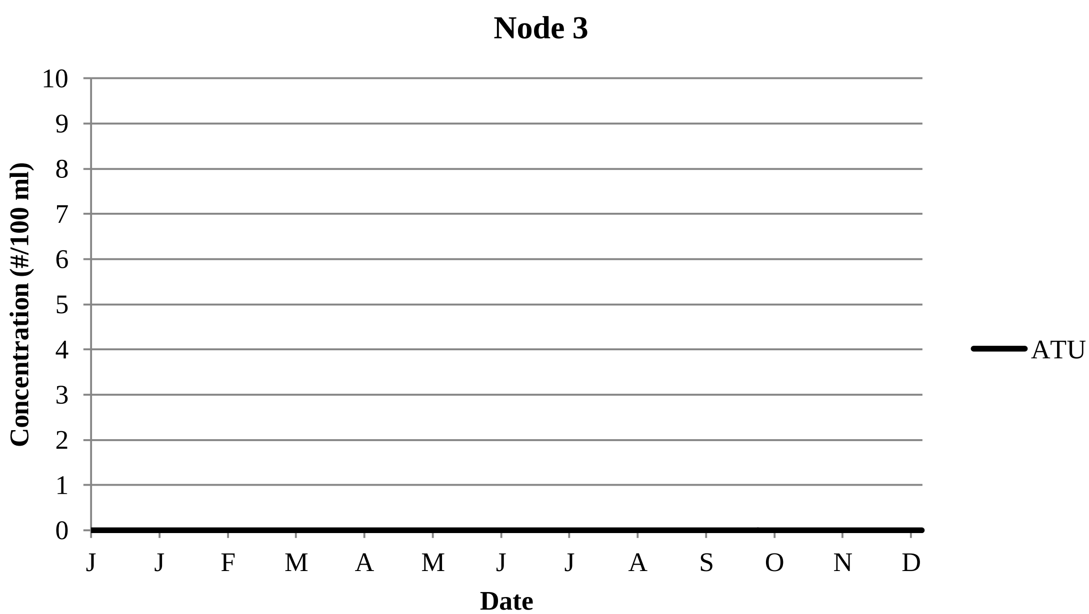

Figure 44. ***E. coli*** **concentrations at Node 3 of the ATU system.**

E. coli concentrations at Node 1 and Node 3 for the conventional system have an inverse relationship. At Node 1 concentrations gradually rise during dry periods and quickly drop during rainfall events. At Node 3 concentrations have sharp peaks mimicking rainfall events. The difference is due to water content and saturated-unsaturated conditions. Concentration values are based on the count of *E. coli* and volume of water. Concentrations increase if the count is increased or if the volume is decreased. Node 1 is unsaturated, and its corresponding water content varies according to rainfall and evapotranspiration. Node 3 is saturated during the entirety of the simulation; therefore, its corresponding water content remains constant as pressure head values change. Changes in concentration values at Node 1are more related to water content changes. Concentrations increase as dryness increases. Evapotranspiration removes water from the soil and decreases the volume of water in the concentration term. During precipitation the concentration is quickly diluted and falls toward zero. Concentrations at Node 3 are more related to change in count of *E. coli*. Rainfall events push concentrations towards the node, but concentration values do not experience dilution because the water content remains the same.

The water table never rises to the surface of the conventional system, but concentrations of *E. coli* are still found at the surface. These results appear to be contradictory; if contaminated water from the system does not reach the surface, then it may be assumed that *E. coli* in the contaminated water would also not reach the surface. Evapotranspiration, the hydraulic gradient, and dispersion could be the hydraulic processes that cause *E. coli* to reach the surface. Precipitation flowing from the surface to the water table mixes with effluent from the septic system. After rainfall events evapotranspiration removes this precipitation and it to the surface. However, evapotranspiration is associated with water vapor that would likely not pull *E. coli* with it. The hydraulic gradient also forces some water to the surface that could then also bring *E. coli* concentrations. The hydraulic gradient causes the system to push high pressure to low pressure, located at the surface. Pressure heads for the conventional system come within 15 cm of the surface. With this distance the hydraulic gradient along with dispersion may be enough to account for *E. coli* at surface. With that said, the presence of *E. coli* at the surface is not well understood. Further research into this phenomenon would be beneficial.

With the given parameters, *E. coli* concentrations reached the soil surface and the water table in the conventional system (Figure 39 and Figure 42). Both are unacceptable situations. *E. coli* reaches the depth of the initial ground water table, but the worst case scenario for groundwater contamination occurs when the water table rises to the outlet (Figure 24). In this case the full concentration from the effluent reaches a saturated soil profile and can be easily tranported through the ground water. *E. coli* concentrations reaching the surface is also a major concern. The Dickinson Bayou is classified as impaired, and OWTSs allowing *E. coli* to reach the ground surface that can then be transported through runoff can be a contributor of that impairment. The simulated mound system does not allow *E. coli* to reach either the surface or the initial water table depth (Figure 40 and Figure 43). With a functional ATU, the surface concentration of *E. coli* is minimal and does not go beyond the top loam surface. However, one of the main issues associated with ATUs is the lack of owner maintenance resulting in inadequate treatment before surface application. If no treatment is added, the surface concentration reaches 1,200,000 count/100 ml. When the soil profiles becomes saturated, that amount of *E. coli* can quickly travel in runoff across the soil surface and reach surface waters with little to no treatment.

Artificial Dispersion

The conventional system was also simulated considering artificial dispersion to account for dispersion losses that could be experience from applying column experiments to field-scale experiments. Figure 45 through Figure 50 below compares the results of the initial simulations without artificial dispersion and results from simulations with artificial dispersion.

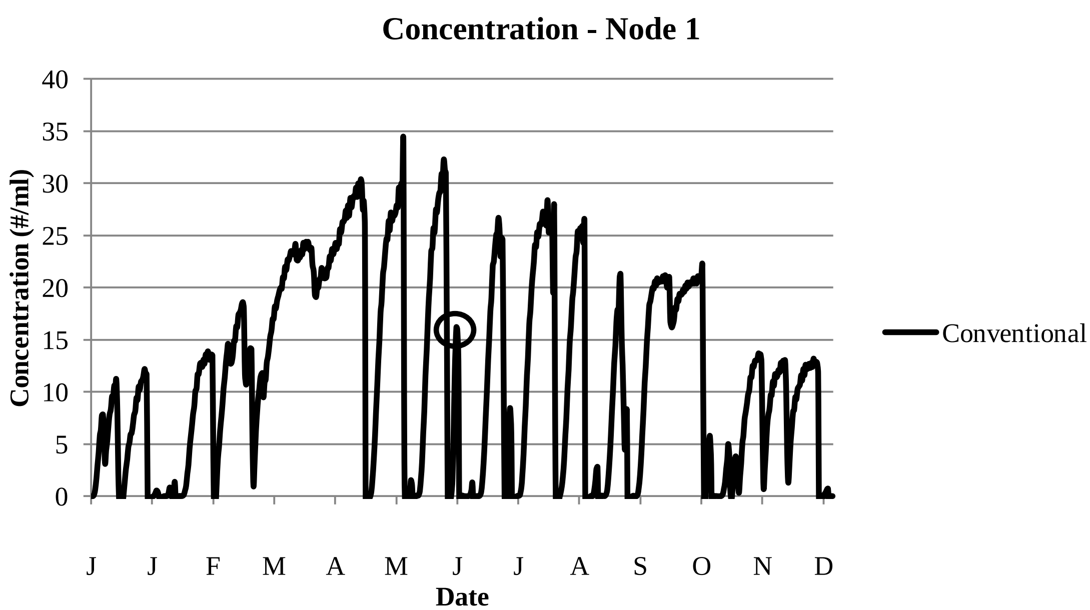

Figure 45. *E. coli* concentrations at Node 1 in conventional system without artificial dispersion.

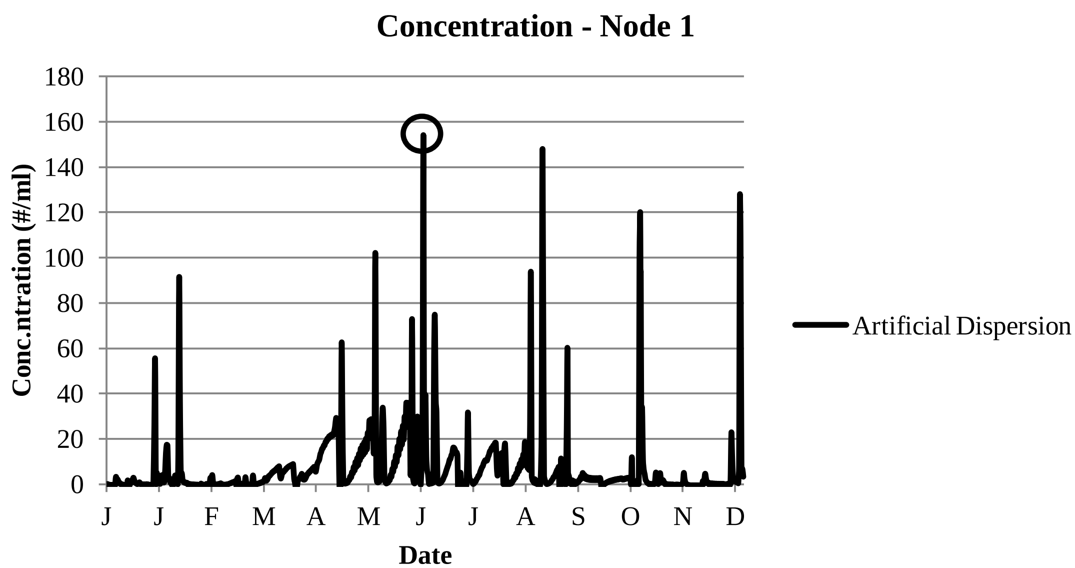

Figure 46. E. coli concentrations at Node 1 in conventional system with artificial dispersion.

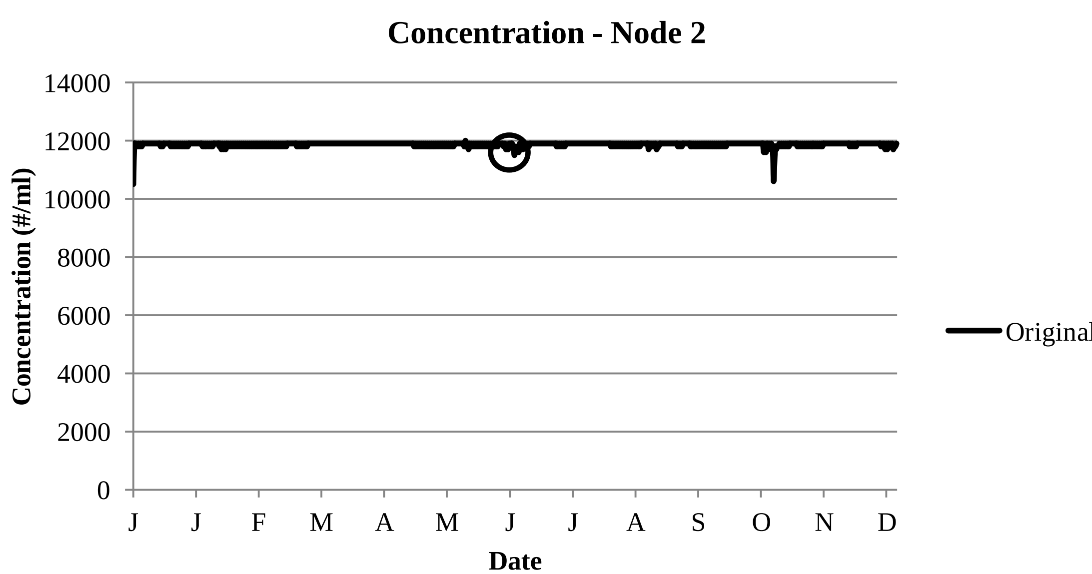

Figure 47. *E. coli* concentrations at Node 2 in conventional system without artificial dispersion.

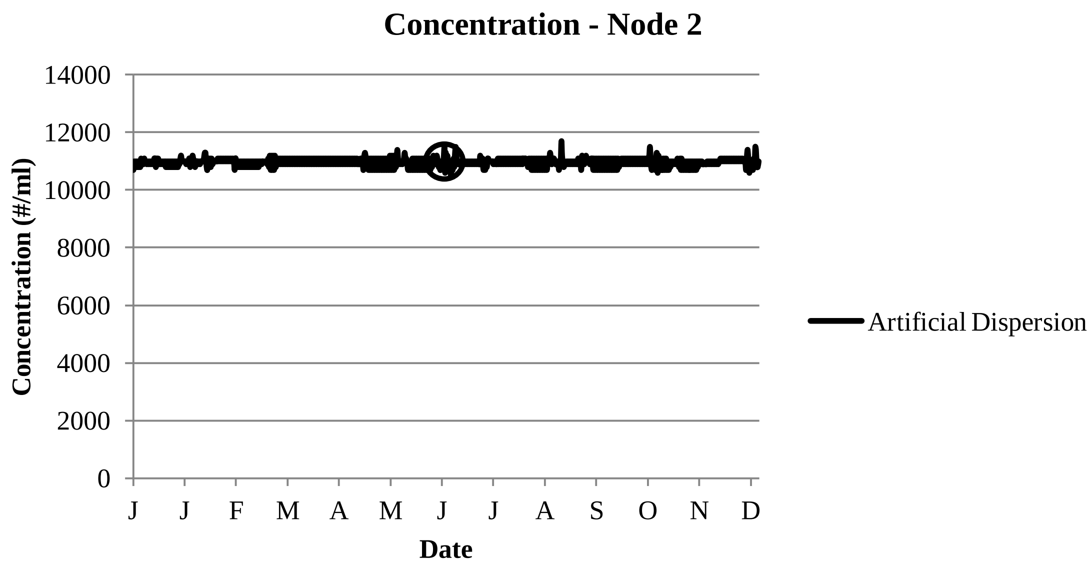

Figure 48. ***E. coli*** **concentrations at Node 2 in conventional system with artificial dispersion.**

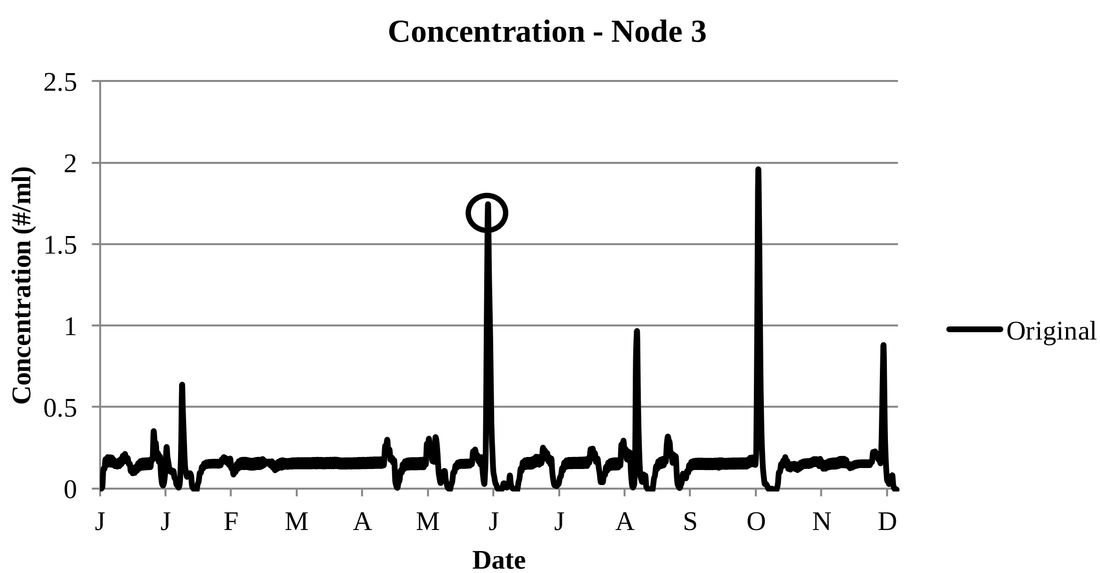

Figure 49. ***E. coli*** **concentrations at Node 3 in conventional system without artificial dispersion.**

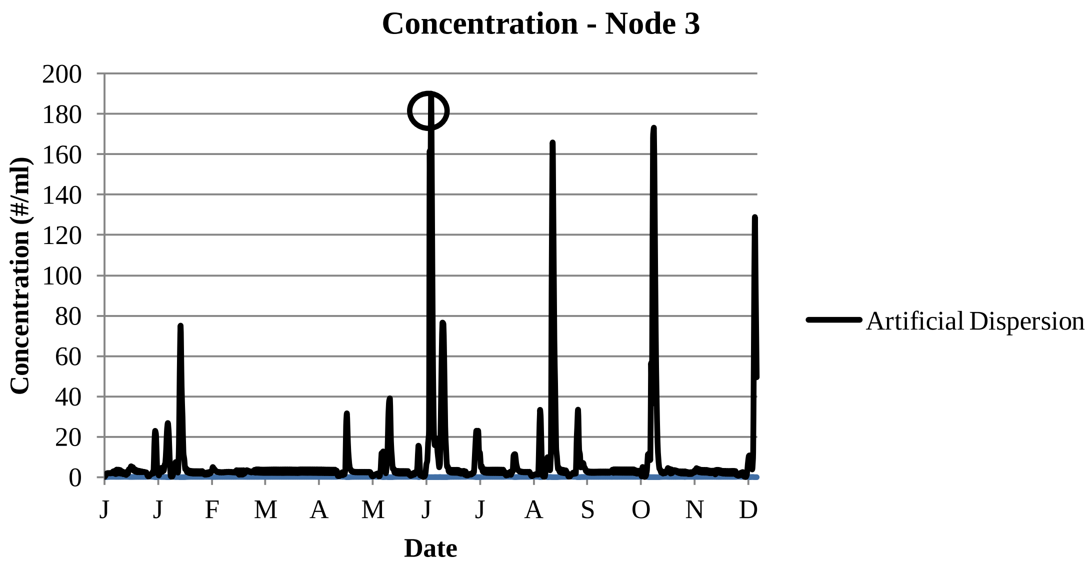

Figure 50. ***E. coli*** **concentrations at Node 3 in conventional system with artificial dispersion.**

Figure 51 through Figure 54 displays the spread in the solute plume through the conventional system with and without artificial dispersion. The cross-sectional images are graphical displays taken from HYDRUS of the simulated conventional system taken at June 1, circled in Figure 45 through Figure 50. Different colors represent the *E. coli* concentration in each area with the scale provided to the right of the figures. Figure 51 and Figure 52 display the results with a scale that includes all *E. coli* levels. The range for these figures is large, from -750 to 13,000 #/ml, so Figure 53 and Figure 54 display the results with a smaller range, -100 to 500 #/ml. With this scale, white areas represent areas with concentrations lower or higher than the minimum and maximum on the scale

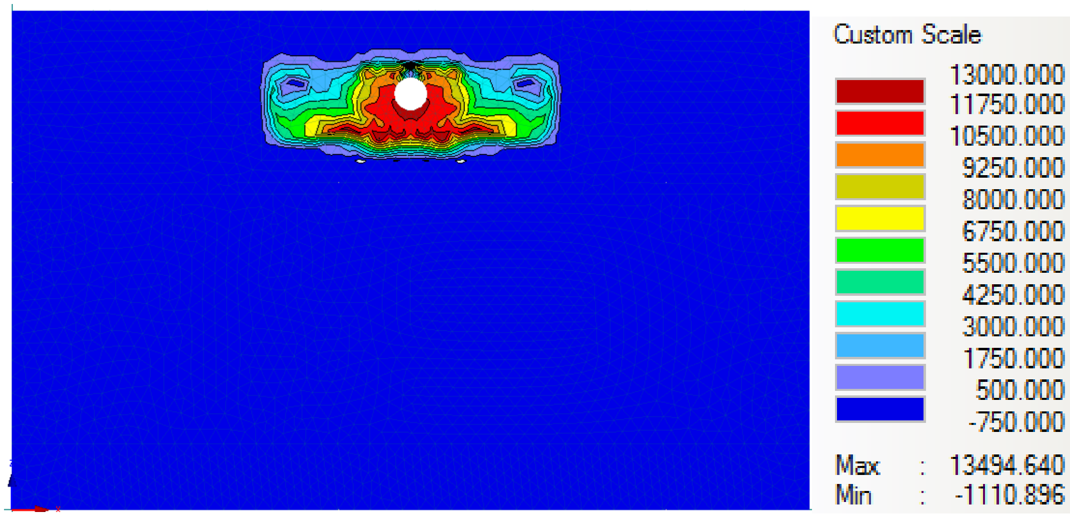

Figure 51. ***E. coli*** **concentrations in the simulated conventional system without artificial dispersion.**

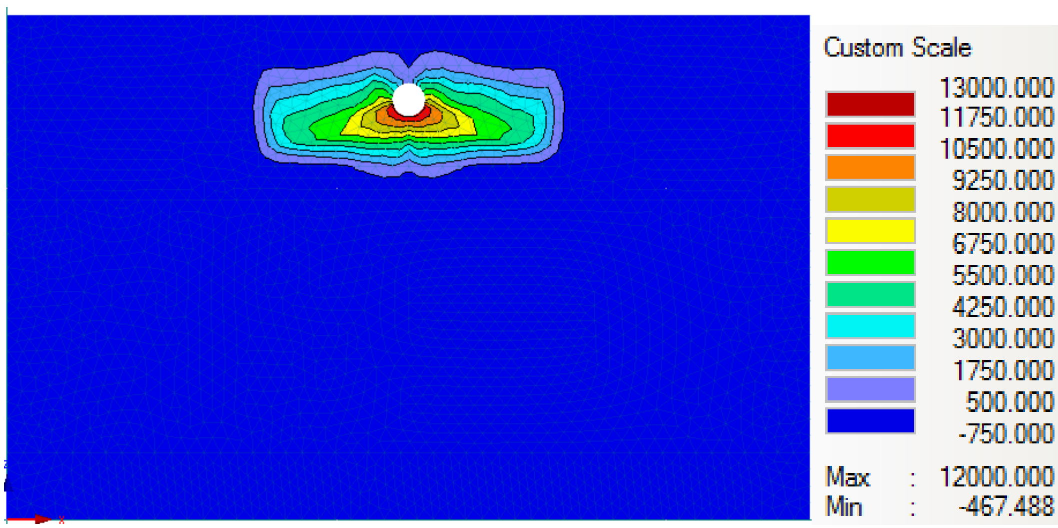

Figure 52. ***E. coli*** **concentrations in the simulated conventional system with artificial dispersion.**

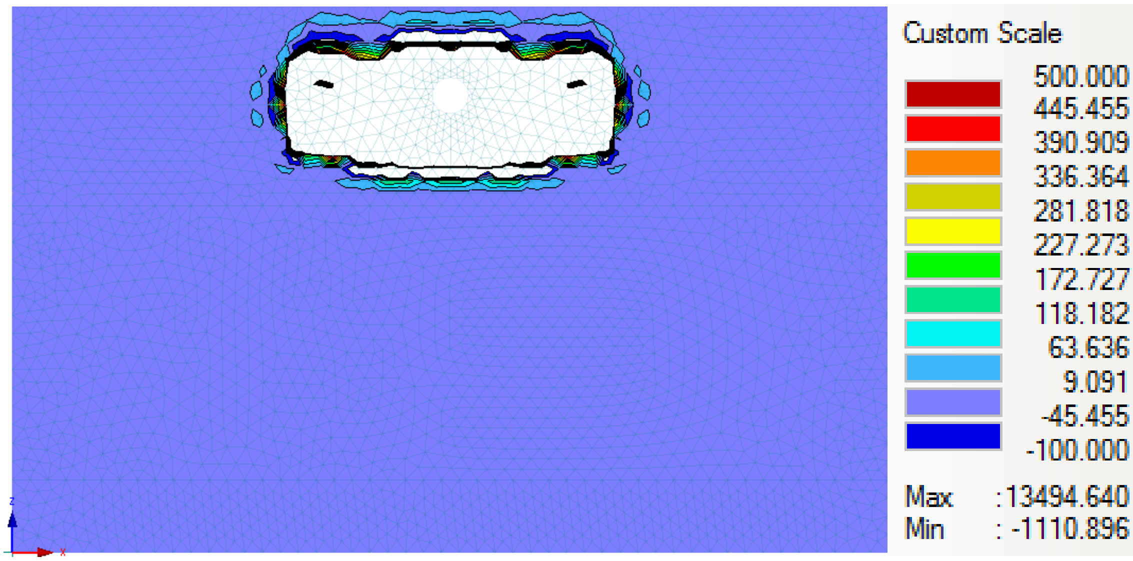

Figure 53. ***E. coli*** **concentrations in the simulated conventional system without artificial dispersion.**

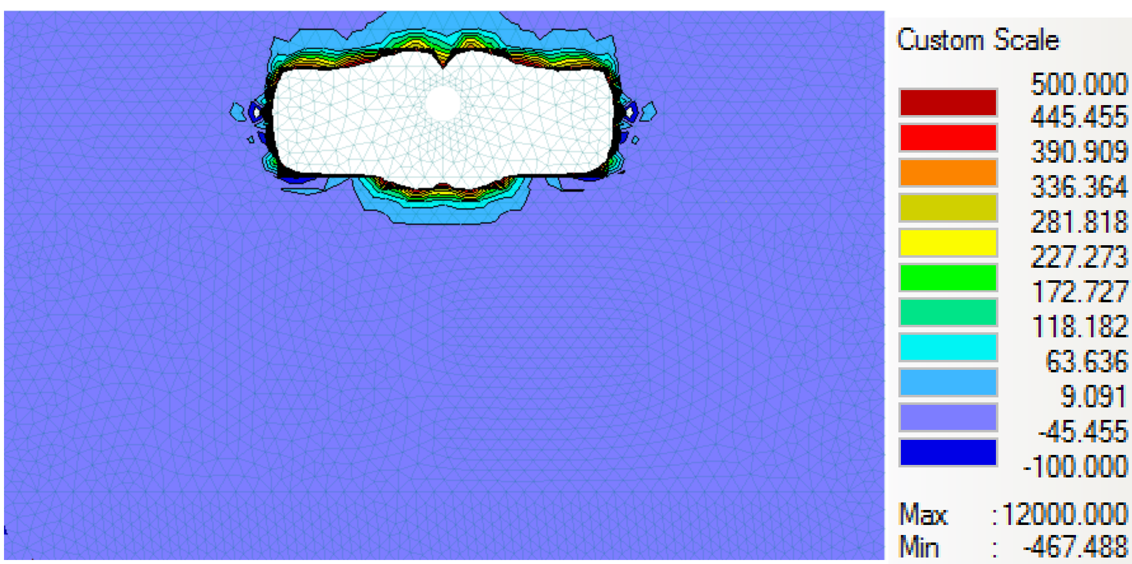

Figure 54. ***E. coli*** **concentrations in the simulated conventional system with artificial dispersion.**

E. coli concentrations in simulation with artificial dispersion were lower than initial concentrations during times of lower rainfall but had much higher peaks during rainfall events. A possible explanation for this difference is that artificial dispersion increased the spread of the concentration plume, decreasing the density of the affected area but allowing the reach of the plume to go much farther. This spread is best seen in Figure 51 and Figure 52. The concentrations around the drainage pipe in the simulation without artificial dispersion were higher, and the area with more than 12,000 #/ml was larger. The simulation with artificial dispersion, however, had higher concentrations reaching farther away from the drainage pipe. Concentrations at Node 1 (Figure 45 and Figure 46) experienced increases in concentration up to doubled or tripled initial values during rainfall events. Concentrations at Node 3 (Figure 49 and Figure 50) experienced the greatest increase. Values from the initial simulation were often increased a one order of magnitude. Considering artificial dispersion, *E. coli* concentrations are less confined to the drainage trench, and ground and surface water contamination due to OWTSs is much worse than the initial simulations would suggest.

Runoff

Runoff results from the HYDRUS simulations were also considered because a main concern of onsite wastewater treatment systems in the Dickinson Bayou watershed is the creation and contamination of runoff from OWTSs. The following systems were simulated: conventional system, conventional system with all loam soil, conventional system with clay soil, malfunctioning ATU system, and mound system. Simulations were also run for to represent a year with heavy rainfall, artificial dispersion, and heavy rainfall and artificial dispersion. Runoff results are displayed in Table 8. Results are given for cumulative runoff volume for one year, total runoff per household, and the peak *E. coli* concentration at the surface for each simulation. Total runoff per household was found by taking the two-dimensional results for one drainage line and distributing along the length of the drainage line (75 feet) and multiplying to equal the amount of drainage lines for the studied house (6). Differences in runoff due to soil type in the conventional system are displayed. Results from a year of high rainfall and for artificial dispersion are also given to represent what would be considered a worst case scenario for runoff and concentration.

Table 8. Runoff volume and concentration for possible OWTS design and scenarios in the Dickinson Bayou watershed.

System	Cumulative Runoff at Surface [cm^3]	Total Runoff Per House [cm^3]	Peak Concentration at Surface [#/100 ml]
Conventional - Loam	0	0	3430
Conventional	138.45	62,302.5	3400
Conventional - Clay	211.77	95,296.5	2840
Conventional - High Rainfall	220.5	99,225	3730
Conventional - Artificial Dispersion	138.45	62,302.5	11,600
Conventional - High Rainfall & Artificial Dispersion	220.5	99,225	33,500
Malfunctioning ATU	1165	1,019,375	1,200,000
Malfunctioning ATU - High Rainfall	2548.3	2,229,762.5	1,200,000
Malfunctioning ATU - Artificial Dispersion	1165	1,019,375	1,200,000
Malfunctioning ATU - High Rainfall & Artificial Dispersion	2548.3	2,229,762.5	1,200,000
Mound	0	0	0
Mound - High Rainfall	0	0	30
Mound - Artificial Dispersion	0	0	4600
Mound - High Rainfall & Artificial Dispersion	0	0	5220

Runoff was generated in the conventional and ATU systems. At the surface of the conventional system, pressure head never reaches zero, and water content never reaches saturation. The ATU system only reaches saturation at the surface twice. Even under these conditions, runoff is still generated. Evapotranspiration is a factor preventing the surface from becoming fully saturated. The surface is affected by evapotranspiration the most as water is constantly removed by this process and prevents full saturation even when runoff is generated. However, if runoff is being generated, it would be assumed that the pressure head at the surface would be positive during that time. A few explanations for this contraction are available. HYDRUS measures pressure head from bottom-up and not top-down. The pressure head located at the top node is in relation to the water table and not to water conditions above the node. Another possibility for this contradiction is that runoff is still generated under these conditions because the water content at the surface is increased, and therefore, infiltration capacity and rate is decreased. Even when the surface is not fully saturated, with a high enough water content infiltration is too slow, and runoff is generated.

The effect of soil type on runoff is seen in the top three lines of the chart. A soil profile with all loam generated no runoff while the initial profile and profile with clay instead of clay loam generated 138.45 and 211.77 cm^3 of runoff, respectively (Table 8). Increased clay content increased runoff. The slow transport through clay keeps water content values above the clay layer higher, increasing runoff potential.

The amount of runoff was much greater for the ATU system than for the conventional system. Having more runoff from ATU systems than from conventional systems makes sense because with ATUs effluent is surface applied while conventional systems have soil sorption of water before it reaches the surface. The surface application minimizes the effects of evapotranspiration, decreasing soil storage capacity and increasing water content. Contrarily, the mound system had no runoff. The increased distance to the water table provided more space for water absorption, allowing water content in soil layers close to the surface to decrease much more between storm periods. The increase in storage capacity prevented runoff from occurring in all simulations of the mound system.

Runoff alone is not the major concern, but the *E. coli* concentration associated with the runoff is. Results for the conventional system show that, under initial conditions, the maximum concentration at the surface was 3400 #/100 ml (Table 8). This concentration, however, could be much higher with more rainfall and assuming artificial dispersion, 33,500 #100/ml (Table 8). Again, these concentration values cannot be taken as fully accurate for the in place systems due to the lack of field data, but they do display that runoff and *E. coli* concentration in runoff from conventional systems is a concern for the Dickinson Bayou watershed. The runoff from an effectively operating ATU system is of no concern because little to no contamination is in the runoff. However, an ineffectively operating ATU system is much more detrimental than conventional systems. The amount of runoff and level of *E. coli* concentration from an ineffective ATU system is much higher that what could come from a conventional system, 2,230,00 cm^3 and 1,200,000 #/100 ml (Table 8), demonstrating how a failing ATU system can cause much more harm to public health than a failing conventional system. Mound systems, however, would not harm the Dickinson Bayou due to runoff because no runoff was generated and the concentration reaching the surface in the worst case scenario was one order of magnitude less than the conventional system (Table 8). These results show that conventional and ineffective ATU systems can contribute to the impairment of the Dickinson Bayou due to contaminated runoff.

Sensitivity Analysis

Results from the sensitivity analysis for longitudinal and transverse dispersivity, diffusion coefficient, attachment and detachment rate, and S_{max}, maximum amount of contaminant on sorption sites are shown in Table 9. Results are also shown in Figure 55 through Figure 60 to find a trend for each parameter.

Table 9. Sensitivity Analysis

Parameter	Value	Factor Changed	C_{max} [#/ml]	ΔC_{max}
Dispersivity, Longitudinal [cm]	0.243	0.50	11.90	-0.21
	0.4374	0.90	10.10	-0.03
	0.486	*1.00*	*9.80*	*0.00*
	0.5346	1.10	9.45	0.04
	0.972	2.00	7.00	0.29
Dispersivity, Transverse [cm]	0.0243	0.50	9.19	0.06
	0.04374	0.90	9.68	0.01
	0.0486	*1.00*	*9.80*	*0.00*
	0.05346	1.10	9.91	-0.01
	0.0972	2.00	10.90	-0.11
Diffusion [cm^2/d]	0.20736	0.50	9.87	-0.01
	0.373248	0.90	9.81	0.00
	0.41472	*1.00*	*9.80*	*0.00*
	0.456192	1.10	9.78	0.00
	0.82944	2.00	9.57	0.02
Attachment [d^{-1}]	3.3408	0.50	144.00	-13.69
	4.4544	0.67	28.10	-1.87
	6.01344	0.90	12.60	-0.29
	6.6816	*1.00*	*9.80*	*0.00*
	7.34976	1.10	7.51	0.23
	10.0224	1.50	1.49E-18	1.00
	13.3632	2.00	3.48E-18	1.00
Detachment [d^{-1}]	0.2304	0.50	10.70	-0.09
	0.41472	0.90	9.98	-0.02
	0.4608	*1.00*	*9.80*	*0.00*
	0.50688	1.10	9.62	0.02
	0.9216	2.00	8.12	0.17

Table 9. Continued.

Parameter	Value	Factor Changed	C_{max} [#/ml]	ΔC_{max}
S_{max} [-]	400	0.40	1.13	0.88
	500	0.50	1.14	0.88
	525	0.53	1.08	0.89
	550	0.55	6.15	0.37
	575	0.58	21.1	-1.15
	600	0.60	24.40	-1.49
	667	0.67	22.80	-1.33
	900	0.90	12.20	-0.24
	1000	*1.00*	*9.80*	*0.00*
	1100	1.10	8.22	0.16
	2000	2.00	4.98	0.49

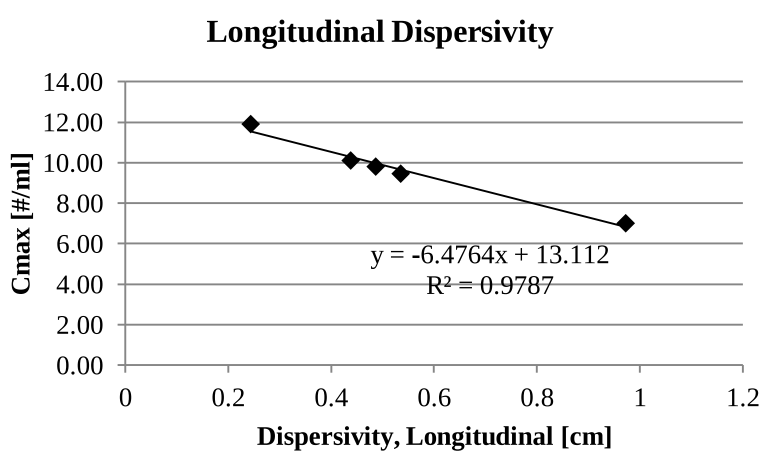

Figure 55. Sensitivity analysis results for the effect of longitudinal dispersivity on concentration in the conventional OWTS.

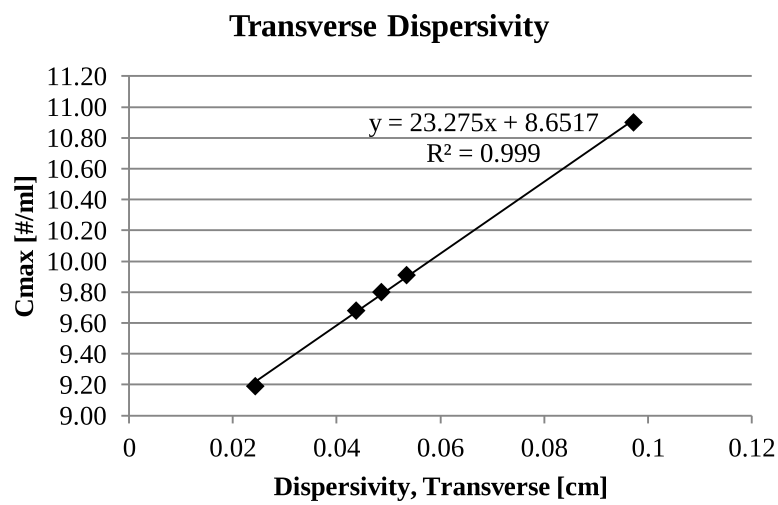

Figure 56. Sensitivity analysis results for the effect of transverse dispersivity on concentration in the conventional OWTS.

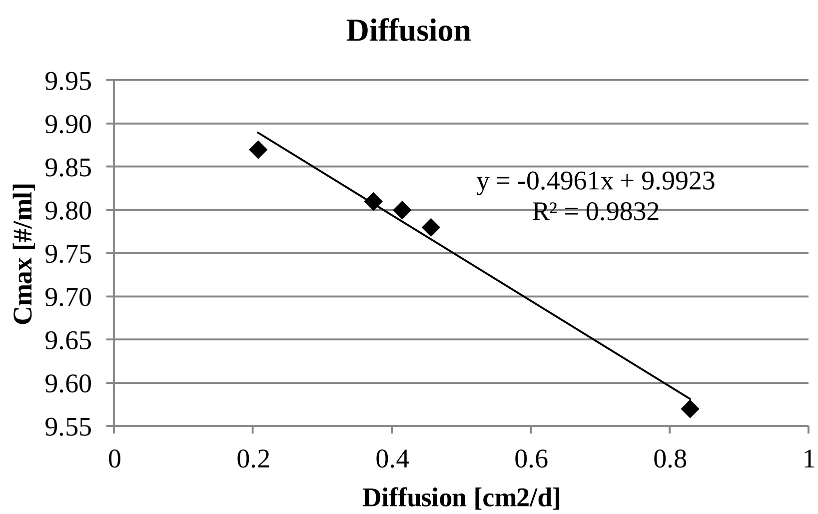

Figure 57. Sensitivity analysis results for the effect of diffusion coefficient on concentration in the conventional OWTS.

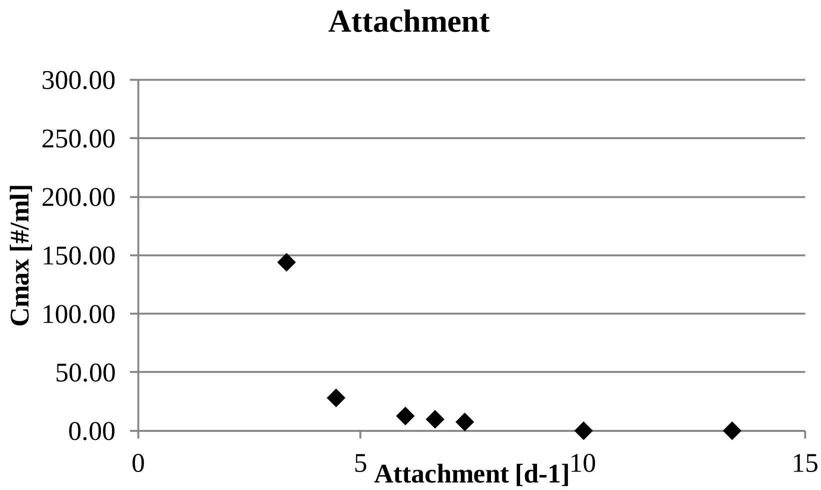

Figure 58. Sensitivity analysis results for the effect of attachment rate on concentration in the conventional OWTS.

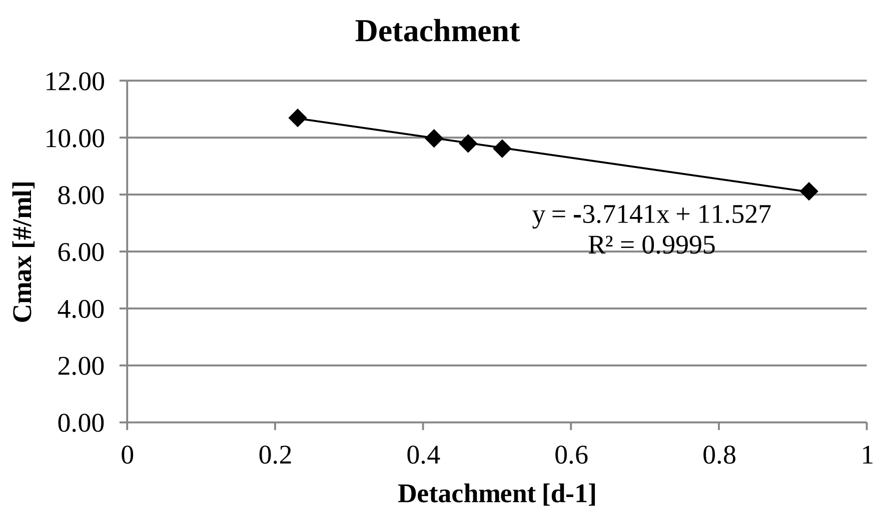

Figure 59. Sensitivity analysis results for the effect of detachment rate on concentration in the conventional OWTS.

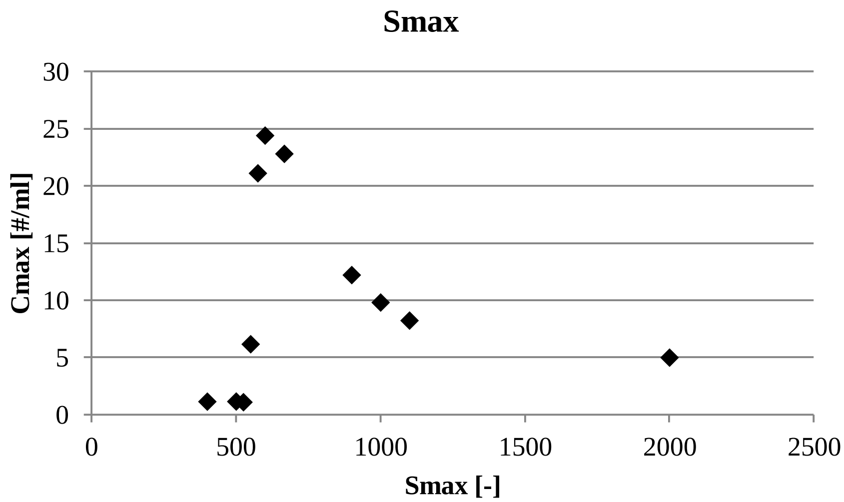

Figure 60. Sensitivity analysis results for the effect of the maximum number of solute on sorption sites on concentration in the conventional OWTS.

Parameters with the greatest effect on measured concentration are longitudinal dispersivity, attachment, and S_{max} (the maximum amount of solute on sorption sites). Dispersivity values are some of the most difficult values to find for solute transport. Values are typically found through tracer tests in column experiments. However, these values are not indicative of field conditions which have larger dispersivity values (Freeze and Cherry, 1979). Attachment values for this project were taken from sand column experiments done by (Bradford et al., 2006). Field values for attachment are likely higher because clay soils have increased attachment rates which would reduce observed concentrations. Another factor making attachment more difficult is that attachment is not believed to be the dominant mechanism for *E. coli* deposition in finer-textured media (Bradford et al., 2006). In this case, clay loam soils may be more likely to remove *E. coli* under straining than by the attachment model used.

CHAPTER IV

CONCLUSIONS

Results from the simulations of conventional septic systems with soil absorption fields show that they would fail in the Dickinson Bayou watershed based on the used parameters. The simulated systems show hydraulic and treatment failure. Hydraulic failure is displayed by system saturation up to the drainage line. Under saturated conditions, contaminants move much more quickly through the soil profile to the surface or groundwater. Hydraulic failure is also displayed by the contribution of the system to runoff. The high water table and clay content were the largest contributors to hydraulic failure. Treatment failure in the simulated systems was shown by allowing *E. coli* to reach the surface and to reach the initial depth of the water table. Runoff from conventional systems is the greatest concern to Dickinson Bayou. The simulated system backs up to a drainage ditch not far from the bayou. Runoff does not have to travel far before reaching this ditch and will be able to be transported quickly without much more treatment before entering the bayou. The Dickinson Bayou is classified as "impaired," and one speculated reason is runoff from conventional systems. This assumption appears to have some legitimacy from observations in this research. However, actual system failure in the Dickinson Bayou watershed cannot be certain in this research. Hydraulic parameters used in simulations come from average values associated with common soil types. These values can vary in different locations. Additionally, solute transport parameters are based on sand column experiments. These parameters could be much different than insitu conditions.

Aerobic treatment units with spray distribution prevent contamination of both ground and surface waters when fully functional. When the system is used and maintained properly, *E. coli* concentrations are kept a low level. These systems, however, are often not properly maintained. When these systems are not maintained, the resulting contamination to surface waters would be greater than that of the conventional systems. More runoff is generated in ATU systems than conventional systems, and compounding greater runoff with a much higher concentration at the surface makes malfunctioning ATU systems much more detrimental to surface waters than conventional systems. Based on the Reed, Stowe, & Yanke, LLC study, malfunctioning ATU systems due to lack of maintenance is a likely scenario (Reed et al., 2001).

Mound systems showed improvements in water and solute transport. The increased amount of soil between the drain field and water table allows for increased removal of *E. coli* before reaching the water table and also prevents the drainage line from becoming saturated. *E. coli* did not reach the surface or the initial depth of the water table in the simulated system. Mound systems were as effective as or more effective than ATU systems in terms of water and solute transport and do not have the same problem in maintenance.

Major limitations for this project are associated with solute transport. The HYDRUS modeling software has limitations in solute transport in that it does not have a biological transport modeling function. The program treats bacteria as a solute and does not consider all processes that would affect the transport of *E. coli*. The greatest detriment to solute transport modeling in this project stems from the lack of field data and research on various types of soils. Research including clay loam, loam, and clay attachment and detachment of *E. coli* along with field data would greatly contribute to these HYDRUS simulations. The vast majority of research concerning *E. coli* transport is done through sand column experiments. Increased research with different soil types would be beneficial to future projects. Most detrimental to quantifying actual *E. coli* concentrations and contamination in the Dickinson Bayou watershed would be field data. Finding pressure head or water content values in the soil profile along with soil and runoff *E. coli* concentrations would allow the modeling to be calibrated and validated.

Without validation of parameters and results for the simulated systems, systems in the Dickinson Bayou watershed cannot be evaluated with certainty. Future research to find field data to be used for model calibration and validation would be able to better address these issues. However, comparing the results of the different systems in uniform conditions shows that mound systems may be the best OWTS option in the Dickinson Bayou watershed. Mound systems operated more effectively than conventional systems and are not associated with as many maintenance issues as ATU systems.

REFERENCES

AgriLife Extension. 2005. Texas average evapotranspiration. 2012.

Beach, D.N.H. 2003. Numerical modeling of unsaturated flow in wastewater soil absorption systems. Ground Water Monitoring Remediation 23:64-72.

Beal, C.D., D. Rassam, E. Gardner, G. Kirchhof, and N. Menzies. 2008. Influence of hydraulic loading and effluent flux on surface surcharging in soil absorption systems. J. Hydrol. Eng. 13:681-692.

Bitton, G., N. Lahav, and Y. Henis. 1974. Movement and retention of klebsiella aerogenes in soil columns. Plant Soil 40:373-380.

Bradford, S.A., J. Simunek and S.L. Walker. 2006. Transport and straining of E. coli O157: H7 in saturated porous media [Online]. Water Resources Research 42.

Brassington, R. 1988. Field hydrogeology. geological society of london handbook series. 41-43.

Budrene, E.O., and H.C. Berg. 1991. Complex patterns formed by motile cells of Escherichia coli. Nature 349:630-633.

Carsel, R.F., and R.S. Parrish. 1988. Developing joint probability distributions of soil water retention characteristics. Water Resour. Res. 24:755-769.

Clapp, R.B., and G.M. Hornberger. 1978. Empirical equations for some soil hydraulic properties. Water Resour. Res. 14:601-604.

Foppen, J.W., M. van Herwerden and J. Schijven. 2007a. Transport of Escherichia coli in saturated porous media: Dual mode deposition and intra-population heterogeneity. Water Res. 41:1743-1753.

Foppen, J.W., M. van Herwerden, and J. Schijven. 2007b. Measuring and modelling straining of Escherichia coli in saturated porous media. Journal of Contam. Hydrol. 93:236-254.

Foppen, J.W.A., and J.F. Schijven. 2006. Evaluation of data from the literature on the transport and survival of Escherichia coli and thermotolerant coliforms in aquifers under saturated conditions. Water Res. 40:401-426.

Freeze, R.A., and J.A. Cherry. 1979. Groundwater. Prentice Hall, Englewood Cliffs, NJ.

Glaveston County Drainage District Number One. 2007. Drainage criteria manual.

Harris County Flood Warning System. 2012. Rainfall gauge data: Gauges 110, 120, 125, & 130; January 2008-December 2011. 2012.

Havelaar, A.H. 1991. Bacteriophages as model viruses in water quality controlag. Water Res. 25:529-545.

Huysman, F., and W. Verstraete. 1993. Water-facilitated transport of bacteria in unsaturated soil columns: Influence of cell surface hydrophobicity and soil properties. Soil. Biol. Biochem. 25:83-90.

Kibbey, H.J., C. Hagedorn, and E.L. McCoy. 1978. Use of fecal streptococci as indicators of pollution in soil. Appl. Environ. Microbiol. 35:711-717.

Lesikar, B. 2008a. Onsite wastewater treatment systems: Aerobic treatment unit. Rep. L-5302.

Lesikar, B. 2008b. Onsite wastewater treatment systems: Septic Tank/Soil absorption field. Rep. L-5227.

Lesikar, B. 2008c. Onsite wastewater treatment systems: Spray distribution system. Rep. L-5303.

Lesikar, B. 1999a. Onsite wastewater treatment systems: Conventional septic Tank/Drain field. Rep. L-5234.

Lesikar, B. 1999b. Onsite wastewater treatment systems: Spray distribution. Rep. L-5236.

Lesikar, B. and V. Weynand. 2002. On-site wastewater treatment systems: Mound system. Texas A&M AgriLife Extension L-5414.

Mawdsley, J.L., R. D. Bardgett, R.J. Merry, B.F. Pain, and M.K. Theodorou. 1995. Pathogens in livestock waste, their potential for movement through soil and environmental pollution. Applied Soil Ecology 2:1-15.

McCray, J.E., S.L. Kirkland, R.L. Siegrist, and G.D. Thyne. 2005. Model parameters for simulating fate and transport of On-Site wastewater nutrients. Ground Water 43:628-639.

Medema, G.J., P. Payment, A. Dufour, W. Robertson, M. Waite, P. Hunter, R. Kirby, and Y. Andersson. 2003. Safe drinking water: An ongoing challenge. p. 11-45. *In* World Health Organization (ed.) Assessing microbial safety of drinking water: Improving approaches and methods. IWA, London, UK.

Morris, C. 2000. Unsaturated flow in unwoven geotextiles. Proceedings of GeoEng.

Motz, E.C., E. Cey, and M.C. Ryan. 2011. Vadose zone microbial transport below at-grade distribution of wastewater effluent. Water Air Soil Pollut.1-15.

National Resource Conservation Service. 1996. Soil quality resource concerns: Compaction.

National Resource Conservation Service. Web soil survey. 2011.

National Weather Service. 2012. Preliminary monthly climate data: League city, January 2008-December 2011. 2012.

NIMSS. 2010. NE1045: Design, assessment, and management of onsite wastewater treatment systems: Addressing the challenges of climate change.

NSF International. 2000. Residential wastewater treatment systems. NSF/ANSI.

Pachepsky, Y.A., A.M. Sadeghi, S. A. Bradford, D.R. Shelton, A.K. Guber, and T. Dao. 2006. Transport and fate of manure-borne pathogens: Modeling perspective. Agric. Water Manage. 86:81-92.

Pang, L., and J. Simunek. 2006. Evaluation of bacteria-facilitated cadmium transport in gravel columns using the HYDRUS colloid-facilitated solute transport model. Water Resour. Res. 42:W12S10.

Pang, L., C. Nokes, J. Simunek, H. Kikkert, and R. Hector. 2006. Modeling the impact of clustered septic tank systems on groundwater quality. Vadose Zone J. 5:599-609.

Pang, L., M. Close, M. Goltz, L. Sinton, H. Davies, C. Hall, and G. Stanton. 2004. Estimation of septic tank setback distances based on transport of E. coli and F-RNA phages. Environ. Int. 29:907-921.

Quigg, A., L. Broach, W. Denton, and R. Miranda. 2009. Water quality in the Dickinson Bayou watershed (Texas, Gulf of Mexico) and health issues. Marine Pollut. Bull. 58:896-904.

Radcliffe, D.E., and J.K. Bradshaw. 2011. A nitrogen model for onsite wastewater systems. *In* A nitrogen model for onsite wastewater systems. April 11-13, 2011 2011.

Reed, Stowe, and Yanke. 2001. Study to determine the magnitude of, and reasons for, chronically malfunctioning on-site sewage facility systems in Texas.

Šimunek, J., M.T. van Genuchten, and M. Sejna. 2011. The HYDRUS software package for simulating the two- and three-dimensional movement of water, heat, and multiple solutes in variably-saturated media: Technical manual. 2.

Soil Conservation Service. 1975. Soil taxonomy: A basic system of soil classificiation for making and interpreting soil surveys. Agricultural Handbook 436:.

Tan Y., Bond W.J., Rovira A.D., Brisbane P.G. and Griffin D.M. 1991. Movement through soil of a biological control agent, pseudomonas fluorescens. Soil Biology Biochemistry 23:821-825.

Texas Coastal Watershed Program. 2010. Bacteria implementation plan for Dickinson Bayou watershed. Rep. Fact Sheet 7.

Texas Commission on Environmental Quality. 2011. Eight total maximum daily loads for indicator bacteria in Dickinson Bayou and three tidal tributaries.

Texas Commission on Environmental Quality. 2001. On-site sewage facilities. TAC Chapter 285.

Tindall, J.A., J.R. Kunkel, and D.E. Anderson. 1999. Contaminant transport. p. 345. *In* Contaminant transport. Unsaturated zone hydrology for scientists and engineers. Prentice Hall, Upper Saddle River, New Jersey.

University of Wisconsin M. 1978. Management of small waste flows. Rep. 600/2-78-73.

USEPA. 2002. Onsite wastewater treatment systems manual.

USEPA. 1997. Response to congress on use of decentralized wastewater treatment systems.

Whitman, W.B., P. De Vos, and D.H. Bergey. 2009. Bergey's manual of systematic bacteriology, volume 3 : The firmicutes (2nd edition). Springer, New York, NY.

Williams, N.D., and M.A. Abouzakhm. 1989. Evaluation of geotextile/soil filtration characteristics using the hydraulic conductivity ratio analysis. Geotextiles Geomembranes 8:1-26.

Zibilske, L.M., and R.W. Weaver. 1978. Effect of environmental factors on survival of salmonella typhimurium in soil. J. Environ. Qual. 7:593-597.

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