Hydraulic Management of SDI Wastewater Dispersal in an Alabama Black Belt Soil

by

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Abstract

Approximately 52% of the 14000 km^2 Alabama Black Belt region is unsuitable for conventional onsite wastewater treatment systems because of low hydraulic permeability, high water table, or other restrictive soil properties. This research evaluates an experimental subsurface drip irrigation (SDI) wastewater disposal system designed to reduce environmental and health risk of surface ponding and deep percolation in native clay soils by dosing wastewater only when field moisture content is at or below field capacity. A soil moisture control system was linked to a manufacturer's SDI wastewater dosing panel and successfully tested in the laboratory. Subsequent field testing of the system was conducted from September 2006 to June 2008 on an unreplicated 500 m^2 Houston clay soil site in west central Alabama using clean well water (September 2006 to June 2007, year one) and a synthetic wastewater (June 2007 to June 2008, year two). A seasonal cropping rotation of sorghum-sudangrass (*Sorghum bicolor*) and winter wheat (*Triticum aestivum*) with rye (*Secale cereale*) was planted at the site to maximize annual water and nutrient crop uptake and mitigate nutrients offsite transport.

Observed hydraulic dosing rates in the drain field varied from a high of 1.18 cm day^{-1} during summer drought conditions to a low of 0.0 cm day^{-1} during wet winter months. Zero dosing in winter prevented surface ponding from applied wastewater but created requirement for at least a two-month waste storage reservoir. Water percolation is a necessary component of effluent treatment in an OWTS. However, estimated monthly

water balance indicated that water percolated below 45 cm depth accounted for over 30% of dosed water in the warm season of year one which was a 30-year historic drought season, but was negligible in year two during a normal rainfall year. Estimated water percolation was presumably the result of preferential flows stimulated by dry weather clay soil cracking. Over this two-year study, only 4 months out of 12 had an observed water balance in favor of soil adsorption. A minimum a two-month onsite wastewater storage requirement is estimated for the experimental system due to zero or low hydraulic disposal periods during typical wet winter months.

Even though soil moisture controlled wastewater dosing may temporarily provide nutrient loads higher than crop uptake needs in year two, monitored crop uptake and soil nutrient profiles provided no direct evidence of drain field nitrogen and phosphorus accumulation or percolation below 100 cm depth. However, the field observation cannot exclude the possibility that nutrients may have passed through the top 100 cm soil and accumulated at deeper soils. Furthermore, it is anticipated that nutrients would be transported into deeper soils in year one had synthetic wastewater been used. . Although leaching of wastewater is not an environmental issue in the majority of the Alabama Black Belt region, improved monitoring of percolation loss to local groundwater is recommended for wastewater dispersal in this region.

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Table of Contents

List of Tables

List of Figures

CHAPTER ONE

INTRODUCTION

1.1 RESEARCH BACKGROUND

Conventional onsite septic systems have been used in the Alabama Black Belt area for decades in spite of the fact that many soils in the region have a high smectitic clay content with severe drainage limitations such as low permeability, high seasonal water table, and restrictive layers. Since the 1950s, over 10 million acres of row crops have been abandoned in the state of Alabama (McNider, 2005), much of it in the Black Belt region. As a result, rural economies in this area have suffered and many existing conventional onsite septic systems are in disrepair. The weakened economy in the Alabama Black Belt makes the retrofit of individual system difficult, if not cost-prohibitive for many households (McCoy et al., 2004). In 2002, the Lowndes County, Alabama court ruled against 37 families who discharged raw wastewater into their backyards and nearby ditches, drawing national attention to onsite system deficiencies in the Alabama Black Belt. According to the US EPA (2002), approximately 47% of the houses in Alabama are served by OWTSs with an average system failure rate of 20%.

In order to alleviate the environmental and health threat from conventional onsite septic systems in this area, a subsurface drip irrigation (SDI), soil moisture monitoring/control, cropping system were incorporated into an experimental wastewater disposal and treatment system. This new system uses real-time drain field moisture content to control SDI wastewater disposal while a seasonal cropping system provides water and nutrient uptake. SDI was used to achieve uniform wastewater distribution and to locate water and nutrients favorable for crop root uptake (Phene and Ruskin, 1995). Drain field soil moisture control of wastewater disposal was incorporated since it has been used successfully in agricultural irrigation for water conservation (Dukes and Scholberg, 2005). Cropping systems have been widely used in agricultural to provide enhanced water and nutrient uptake (Askegaard and Eriksen, 2008; Wang et al., 2008) and pesticide leaching control (Ferraro et al., 2003; Giupponi, 1998). A seasonal cropping system of sorghum-sudangrass (*Sorghum bicolor*) and winter wheat (*Triticum aestivum*) with rye (*Secale cereale*) was applied over the SDI wastewater drain field to enhance removal of SDI applied wastewater and nutrients.

1.2 Research Objectives

The goal of this research is to evaluate the hydraulic performance of an experimental soil moisture based wastewater dosing system and resulting soil nutrient profile after one year of synthetic wastewater application. Specific objectives of this research are to:

1) Evaluate the potential environmental and health threat from conventional onsite wastewater treatment systems in the Alabama Black Belt.
2) Develop, test in the laboratory, and field install a soil moisture based SDI wastewater

disposal control system.

3) Evaluate system control over drain field water movement through determination of two-year monthly hydraulic disposal rates and water balances.
4) Evaluate system nutrient impact to the drain field from one-year of synthetic wastewater field application through lysimeter sampling, soil nutrient profile, and numerical simulation.

1.3 DISSERTATION ORGANIZATION

This dissertation includes seven chapters. Except for Chapter One (General Introduction), Chapter Two (Literature Review), and Chapter Six (Conclusions and Suggestions for Future Research), each other chapter of this dissertation is formatted as a stand-alone journal paper.

Chapter One provides a background and introduction for this dissertation. Chapter Two provides a literature review of decentralized systems related to water and nutrient management. Chapter Three presents the current status of rural wastewater disposal systems in the Alabama Black Belt area and provides results of a GIS risk assessment tool developed as justification for this research. Chapter Four describes the laboratory and first year field testing of the experimental wastewater disposal system designed to overcome specific deficiencies of conventional septic systems in the high clay soils of the Black Belt. Chapter Five presents the results of the system evaluation over its water and nutrient management over its drain field using combined experimental results from first year clean water study and the second year synthetic wastewater study. Recommendations are made on system application based on the advantages and disadvantages of the experimental system. Chapter Six provides conclusions as well as recommendations for future research.

CHAPTER TWO

LITERATURE REVIEW

2.1 DECENTRALIZED WASTEWATER TREATMENT

2.1.1 DECENTRALIZED WASTEWATER TREATMENT IN THE UNITED STATES

The decision to use centralized or decentralized wastewater treatment for a site or region is a cost–benefit analysis (Prihandrijanti et al., 2008). Decentralized wastewater treatment methods are used when connecting isolated households to the existing sewer system is not practical and/or economical (Viraraghavan, 1986; US Census Bureau, 1990).

The Consortium of Institutes for Decentralized Wastewater Treatment, also known as “The Onsite Consortium" defines commonly used terms for decentralized wastewater treatment systems (The Onsite Consortium, 2007). A decentralized wastewater treatment system is defined as a wastewater treatment system for collection, treatment, and dispersal/reuse of wastewater from individual homes, clusters of homes, isolated communities, industries, or institutional facilities, at or near the point of waste generation. Based on the serving population and system size, a decentralized wastewater treatment system can be further divided into a community system, cluster system, onsite system, or individual system (Table 2.1).

A community wastewater treatment system is defined as a publicly owned wastewater treatment system for collection, treatment and dispersal of wastewater from two or more lots, or two or more equivalent dwelling units. A cluster wastewater treatment system is defined as wastewater treatment systems designed to serve two or more sewage-generating dwellings or facilities with multiple ownership; typically to include a comprehensive, sequential land-use planning component and private ownership. An onsite wastewater treatment system (OWTS) is defined as a wastewater treatment system that relies on natural processes and/or mechanical components to collect and treat sewage from one or more dwellings, buildings, or structures and disperse the resulting effluent on property owned by the individual or entity. An individual wastewater treatment system is defined as a wastewater treatment system designed to serve one sewage-generating dwelling or facility.

Table 2.1 Defined decentralized system categorization.

	Community system	Cluster system	Individual system	Onsite system
Serving size	two or more lots, or two or more equivalent dwelling units	two or more sewage-generating dwellings or facilities	one sewage-generating dwelling or facility.	one or more dwellings, buildings, or structures
Ownership	Publicly	Multiple ownership	Individual	Individual

Source: The Onsite Consortium (2007)

In the United States, about 30% of households are using onsite sewage disposal, while in Alabama this number is 47% (AOWTC, 2005). Among current onsite sewage disposal methods, conventional septic tank systems have the longest history and the most common application (US EPA, 2002). In a conventional septic system, sewage goes through primary settling and biological reaction during its retention (around 48 hours) in the septic tank. Upon reaching a preset overflow level, the supernatant from the septic tank is disposed by gravity to the drain field where percolation through an unsaturated soil zone provides various levels of treatment of the effluent (AOWTC, 2005).

US EPA (2002) defines a conventional septic system as a highly efficient, self-contained, underground wastewater treatment system. Because septic systems treat and dispose of household wastewater onsite, they are often more economical than centralized wastewater treatment systems in rural areas where lot sizes are larger and houses are widely spaced. Septic systems are simple in design, which make them less expensive to install and maintain than other systems (US EPA, 2002; Prihandrijanti et al., 2008). By using natural processes to treat wastewater onsite, usually in a homeowner's backyard, septic systems do not require the installation of miles of sewer lines, making them less disruptive to the environment (US EPA, 2002; AOWTC, 2005).

Onsite wastewater treatment industry scientists, engineers, and manufacturers have developed a wide range of alternative technologies such as recirculation sand filters, peat-based systems, mound systems, and package aeration units to address risks from increased hydraulic load and nutrient and pathogen contamination (US EPA, 2002). With proper management and oversight, alternative systems can be installed in areas where soils, bedrock, fluctuating ground water tables, or lot size limit the use of conventional septic systems (US EPA, 2002). According to the EPA Onsite Wastewater Treatment Systems Manual (US EPA, 2002), there are currently thirteen types of advanced onsite wastewater treatment technologies: 1) Continuous-Flow, Suspended-Growth Aerobic Systems (CFSGAS); 2) Fixed-film processes; 3) Sequencing batch reactor systems; 4)

Effluent disinfection processes; 5) Vegetated submerged beds and other high-specific-surface anaerobic reactors; 6) Evapotranspiration and evapotranspiration/infiltration; 7) Stabilization ponds, constructed wetlands, and other aquatic systems; 8) Enhanced nutrient removal--phosphorus; 9) Enhanced nutrient removal; 10) Intermittent sand/media filters; 11) Recirculating sand/media filters; 12) Land treatment systems; and 13) Renovation/restoration of subsurface wastewater infiltration systems (SWIS).

2.1.2 Environmental Challenge for Conventional Onsite Septic System

The environmental challenges for conventional septic systems come from the almost complete reliance on soil properties (Oron, 1996). The soil in a drain field should have a percolation rate that allows wastewater to penetrate into soil at a rate to provide adequate treatment of nutrients and contaminants (Venhuizen, 1995). Unsaturated flow is necessary to obtain efficient aerobic treatment while providing sufficient time for effluent to stay in contact with the soil (Venhuizen, 1995). Tyler et al. (1977) emphasized that it is essential, particularly in coarser soils, that a "clogging mat" form at the top of the gravel/soil interface of a conventional gravity trench to create a zone of partially restricted flow, helping to assure that flow through the soil beyond the clogged zone is unsaturated.

In the United States, by the late 1800s the Massachusetts State Board of Health and other state health agencies had documented the linkage between disease and poorly treated sewage, recommending treatment of wastewater through intermittent sand filtration, with land application of the resulting sludge (US EPA, 2002). Septic tanks for primary treatment (settlement, with limited biological nutrient and contaminant removal) of wastewater appeared in the late 1800s. Discharge of septic tank effluent into gravel-lined subsurface drains became common practice during the middle of the 20th century (Kreissl, 2000). Although the use of septic systems significantly improved public health and water quality in urban areas, the technology is old for homes and businesses not connected to a centralized collection and treatment system (Kreissl, 1982).

The average phosphorus concentrations in a septic tank effluent were reported to be 3 to 20 mg/L (Robertson et al., 1998; Whelan, 1988), with about 85% as orthophosphate (Reneau and Pettry, 1976). Charles et al. (2005) statistically analyzed the results of several intensive septic tank effluent field surveys from 1976 to 1999 in Australia and the United States (Table 2.2), including his own field survey on 200 septic tank effluents in Australia. They compared the results to published regulations in Australia and the United States and concluded that current septic tank effluent quality standards do not ensure safe system design. Charles et al. (2005) observed that the 80th percentile of their effluent survey values, TN (250 mg/L) and TP (36 mg/L), should be used in new regulations to minimize overloading that is associated most with onsite system failure. The more commonly used 50th percentile allows that 50 percent of systems designed are exposed to potential drain field overloading and subsequent failure (Lesikar, 2007).

Over 50% of the United States population draws ground water for its potable water supply, while 98% of self-supplied domestic and rural households depend on ground water (US EPA, 1990). During the mid 1940s, infant mortality was reported to be related to NO_3^--N concentrations in drinking water, especially in rural areas of the United States (Hergert, 1986). Public hygiene issues related to onsite septic systems in urban fringes has received attention since 1980 (Boyle and Otis, 1981; DeWalle and Schaff, 1980). Recent medical research has indicated that infant mortality is attributed to NO_3^--N concentrations from improper disposal of human and animal waste (Fare, 1993). Carroll and Goonetilleke (2005) confirmed that a high onsite system density of 290 unit km^{-2} can significantly impact shallow groundwater with increased nutrient and pathogen levels.

Another challenge of conventional onsite septic systems is dependence upon gravity flow through the drain field before assimilation by the soil (Venhuizen, 1995). Typically, gravity dosing alone cannot achieve a uniform wastewater distribution and the result is localized loading rates that are far higher than design loading rates, a circumstance which has been reported in several investigations (Reneau et al., 1989, Harper et al., 1982) and which EPA states is the main cause of system failure (US EPA, 2002).

Table 2.2 Effluent survey values compiled by Charles et al. (2005)

Value	SS mg/L	BOD mg/L	TN mg/L	TP mg/L	Thermotolerant coliforms cfu. 100/ml	Guideline
International						
Average[1]		94	44	8.6	3.7×10^4	USA
Range[1]		46-156	19-53	7-17	4×10^3-3×10^5	USA
Range[2]	50-100	140-200	40-100	5-15	$10^{\wedge5}$-$10^{\wedge8}$	USA
Average[3]	165	280	92	10.5		
85th percentile[3]	250	350	105	14		
Average[4]	54	158	55	15		
Range[4]	11-695	20-480	10-125	4-90		
Range[5]	44-54	129-147	41-49	12-14		
Australia						
Average[6]	448.5	365.7	75.7	21.3		NSW
Range[7]	17-6970	22-2133	30-60	10-20		
Range[8]	40-250	120-280	30-60	10-20		

Note: The number of each superscript represents a field survey on septic effluent.

2.1.3 Subsurface Drip Irrigation Applications With Wastewater

Subsurface drip irrigation first gained popularity in agriculture about 1980 (Beggs et al., 2004) and the uniform hydraulic performance of SDI has been advanced since then (Provenzano et al., 2005; Puig-Bargues et al., 2005; Sahin et al., 2005; Shaviv and Sinai, 2004). SDI application of wastewater was investigated by researchers studying fate and transport of pesticides (Boyd et al., 2003; Jebellie and Prasher, 1999), nutrients (Bakhsn et al., 2000) and viruses (Assadian et al., 2005).

Ruskin (1992) first discussed the potential of using SDI for wastewater reclamation. Phene and Ruskin (1995) present the concept of using SDI to prevent water and nutrients from leaving root zones to maximize nutrient and water utilization. Jnad et al. (2001) studied the impact of SDI wastewater application on soil chemical characteristics over silty clay loam and fine sandy loam and concluded that sodium should be considered if sodium loading to the field is high since sodium accumulation can decrease soil hydraulic conductivity and deteriorate soil physical properties. They also found that phosphorus concentrations were significantly greater near the emitter and close to the soil surface where the drip lateral was installed at a shallow depth. They found no drastic changes along the soil profile in soil TN, Ca, Mg, K, EC, and TOC. Assadian et al. (2005) studied wastewater SDI application on loamy sand and clay loam under arid and semi-arid weather conditions in Texas. They found that virus movement in soil under unsaturated SDI flows was limited to areas around the subsurface emitter. However, bacteriophage persisted in both sandy and loamy soils for a 28-day period after the last irrigation. A successful and safe reclamation of sodic/saline wastewater for subsurface drip irrigation will depend on management strategies that focus on irrigation pretreatment, virus monitoring, field and crop selection, and periodic leaching of salts (Assadian et al., 2005). Li et al. (2007) found that nitrate accumulated toward the boundary of the wetted soil volume for uniform and layered soils (sandy to sandy loam) under drip irrigation.

In general, SDI application of wastewater has the benefit of creating a uniform hydraulic distribution in the drain field (Ruskin, 1992; 1995) and has the potential to restrict nutrient movement by adjusting disposal amount and frequency to a confined wetting front (Jnad et al., 2001). Design dosing is based on the soil texture and specific environmental protection requirement.

2. 2 DRAIN FIELD WATER MOVEMENT

2. 2.1 FORMS OF WATER MOVEMENT

For a drain field, water inputs and outputs are normally composed of precipitation, wastewater application, infiltration, surface runoff, percolation, soil evaporation, and plant transpiration. The forms of these water movements are demonstrated in Figure 2.1.

Infiltration is the process of water entering the soil from the surface. Infiltration under non-saturated flow was first numerically represented by Richards in the 1920s (Raats, 2001), and has been developed into several different classes that hold somewhat different views on the relationship between soil water content, diffusivity and hydraulic conductivity (Raats, 2001). Nevertheless, an appropriate infiltration rate is important to maintain unsaturated flow inside the drain field for surface wastewater application and treatment (Venhuizen, 1995).

Surface runoff is a result of storm water excess above the water holding capacity of the soil (Horton, 1933). The adverse environmental impact from surface runoff is its potential to take with it nutrients and contaminants that were formerly immobilized in the field (Rhode et al., 1981; Arora et al., 1996; Boyd et al., 2003). Soil moisture, surface porosity, and vegetative growth and covering rate affect the runoff from a soil surface (Lowrance et al., 1997). Smith et al. (2001a) found that surface runoff is the main cause of ammonia nitrogen loss during liquid manure application. Storm water applied to the soil where nutrients were recently applied presents a higher risk of nutrient loss (Jarvis et al., 1987; Edwards and Daniel, 1993).

Soil evaporation and plant transpiration are related to local weather conditions, plant growth status, and plant covering rate over the field (Savabi and Williams, 1995; Arnold et al., 1995). Soil evapotranspiration is a major pathway for applied water and is directly related to plant nutrient uptake, thus also an important nutrient pathway out of the soil (Venhuizen, 1995).

Percolation is the process of water movement through soil layers by gravity and capillary forces. Deep percolation occurs when water moves down through the soil profile below the root zone and cannot be utilized by plants (SSSA, 2002). It is important for applied wastewater to pass through soil in order to obtain adequate treatment from soil and plant root uptake (Venhuizen, 1995), but deep percolation is a phenomenon that is not desirable for water conservation in agriculture (Mermound et al., 2005; Ndiaye et al., 2007), nor for controlling contaminant movement (Jebellie and Prasher, 1999; Bakhsh et al., 2000; Boyd et al., 2003). Significant deep percolation indicates a waste of water for agriculture (Patel and Rajput, 2007; Vishnu et al., 2008) as well as potential underground water pollution (US EPA, 2002).

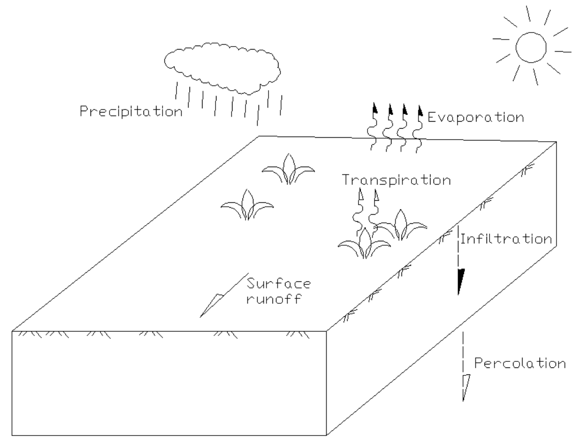

Figure 2.1 Forms of water movement in drain field

For crop agriculture, a water budget that minimizes water to satisfy crop needs is often a target (Ayars et al., 1999; Alam et al., 2002; Wysocki et al., 2005). However, for wastewater land application, the purpose is to dispose as much water as possible without exceeding regulations and endangering the environment (Jnad et al., 2001). For this reason, numerous studies on agriculture irrigation and wastewater land disposal have focused on controlling the fate and movement of nitrogen (Bakhsh et al., 2000; Smith et al. 2000a; Beggs et al., 2004), phosphorus (Reneau et al., 1989, Smith et al., 2000b), pesticide (Jebellie and Prasher, 1999; Boyd et al., 2003), and pathogens (Yavuz-Corapcioglu and Haridas, 1984; Damgaard-Larsen et al., 1977; Foppen et al., 2007).

2.2.2 FIELD CAPACITY MEASUREMENT

Veihmeyer and Hendrickson (1931) first introduce the field capacity (FC) concept and define it as “the amount of water held in the soil after excess gravitational water has drained away and after the rate of the downward movement of water has materially decreased.” Several assumptions not stated in this FC definition are that the soil is deep and permeable, no evaporation occurs from the soil surface, and no water table or slowly permeable barriers occur at shallow depths in the profile (SSSA, 1986).

Gardner (1960) defined FC as that water content below which the hydraulic conductivity is sufficiently small that redistribution of moisture in the soil profile due to a hydraulic head gradient can usually be neglected. However, different soils take different time periods to achieve the “equilibrium” point. FC is not truly an equilibrium water content but instead is that water content at which the soil water flux out of the rooting zone becomes negligible and no significant change in water content occurs with time (SSSA, 1986).

In 1997, the Glossary of Soil Science Terms (SSSA, 1997) defined FC as “the content of water, on a mass or volume basis, remaining in a soil 2 or 3 days after having been wetted with water and after free drainage is negligible”. This definition assumes a uniform soil profile and zero evaporation from the soil surface.

It is emphasized in the two generations of Soil Science Society of American (SSSA) published soil analysis manual (SSSA, 1986; 2002) that field capacity represents a practical and readily understandable coefficient to measure the ability of a soil to retain water and it is of significant importance to identify a suitable procedure to determine its value. If at all possible, the test should be representative of actual conditions. Only a field experiment can take into account the various factors affecting the soil water regime at FC, and preference should be given to field tests for determining the FC (SSSA, 2002).

The principle of an in situ FC test is to add water to a field soil, wetting the soil to a desired depth. After the water has redistributed into the drier underlying soil and drainage from the initial wetted zone becomes negligible, the water content is taken as in situ FC (SSSA, 2002). For practical reasons, an arbitrary drainage time of 48 h has been used in the field capacity definition. However, it is important to emphasize that, in reality, while the hydraulic conductivity of coarser-textured soils may become “negligible” in less than 24 h, the hydraulic conductivity for finer-textured soils may continue to drain at a “non-negligible rate” for weeks (SSSA, 1986; 2002). Furthermore, field capacity can be determined for the same plot during different times of the year to allow for possible temporal variations since wetting history and changes in rooting depth with time can affect the FC (SSSA, 2002).

There are attempts (Cassel and Sweeney, 1974; Jamison and Kroth, 1958) to relate FC to water retention characteristics. It is worth noting that water retention is a soil property, whereas FC is a process-dependent value of the water content of the profile (SSSA, 2002). Jamison and Kroth (1958) selected the equilibrium water content at 33 kPa to estimate FC for their soils even though the pressure values ranged from > 10 kPa to >100 kPa. Over time, the use of the 33-kPa water content value has been adopted as field capacity (The value of FC is much affected by field conditions, so lab tests must be considered only as approximate methods (SSSA, 2002). Gardner (1968) pointed out that FC should be more related to the hydraulic conductivity of a soil than to the soil water matrix potential.

Even though there is disagreement about the laboratory approximation methods at appropriate pressures for specific soil textures, the selection of the appropriate pressure for desorption of a particular soil sample is also not straightforward (Cassel and Sweeney, 1974). The 33-kPa water content may correlate well with a particular coarse-textured soil rather than the 10-kPa water content. However, for coarse-textured soils as a group, the 10-kPa water content is usually considered to be the better choice. Some other reasons for this observed deviation are difference in organic matter content, soil structure, degree of compaction, degree of wetting, and percent sand, silt and clay (Reeve et al., 1973). FC values for different soils were found to vary from -50 to less than -350 cm (SSSA, 2002). Actually, -330 cm of water pressure (~1/3 bar) represents somewhat of a compromise between these laboratory results (McIntyre, 1974; Addiscott and Whitmotr, 1991; SSSA, 1986).

Numerous studies found that FC changes with seasons as field condition changes. However, typically, the FC for a given soil is treated as a “constant,” as a specific value which never changes. The concept of a constant value of FC is criticized as “people who do not really understand the soil physics” from a scientific perspective (SSSA, 2002). Due to the difficulty in applying a defined FC to field measurement and the complications associated with selecting a lab approximation (SSSA, 1986; 2002), FC can be either field or lab measured based on available literature to determine a certain working range. However, the resulting soil moisture thresholds should be tested over time to evaluate the influence of FC on the specific engineering application.

2.2.3 Soil Water Movement Modeling

Current multi-dimensional vadose zone solute transport numerical models are BioF&T, Chemflux, SUTRA, and HYDRUS (Table 2.3). HYDRUS is notable among them in that it incorporates much of the power of the multi-dimensional general pollutant transport models while having features oriented specifically to modeling irrigation and drainage (Beggs et al., 2004).

HYDRUS has been used to simulate and predict water movement (Skaggs et al., 2004; Mermound, 2005; Stillman et al., 2006) and solute movement (Hanson et al., 2006; Ndiaye, 2007; Hu et al., 2007) in field and laboratory experiments. Due to the heterogeneity in field conditions, validating a model is not always feasible (Hopmans and Šimůnek, 1997; Deurer et al., 2003; Haws et al., 2005).

Table 2.3 Common multi-dimensional vadose zone solute transport numerical models

Model	Maximum dimensions	Organization	Major features or applications
BioF&T	3	Scientific Software Sandy, UT	Multi-component biodegradation and transport including oxygen limited and Monod-type.
Chemflux	3	Rockware, Inc. Golden, CO	Chemical transport, including decay; self-adjusting element mesh.
SUTRA	2	USGS	Saturated and vadose transport
HYDRUS	3	USDA Salinity Lab, Riverside, CA	Multi-component chain reactions, irrigation and drainage features.
RZQM	2	USDA-ARS	Chemical transport over a wide range of topographies, soil types, climatic conditions, and management practices.

Note: Adapted from Beggs et al. (2004)

Royer et al. (2002) mentioned that because it is typically infeasible to make detailed distributed measurements of soil properties at a study site, parameters must be estimated based on one hydrograph, which leads to a higher likelihood of non-unique parameter sets and an ill-posed inverse calibration. In a deterministic model, representative values of soil properties are often accomplished by assuming a representative elementary volume (REV) of the study site, where that medium is represented by homogenized parameters. The REV assumption holds if there is a separation of scales such that the volume of the spatial heterogeneities is significantly smaller than the macroscopic transportation domain being modeled (Haws et al., 2005). Successful HYDRUS calibrations are either carefully controlled laboratory soil column tests (Pernyeszi et al., 2006; Dousset et al., 2007; Suarez et al., 2007) or field tests that treat the whole field as a single unit (Moradi et al., 2005; Pot et al., 2005; Fernandez-Galvez and Simmonds, 2006; Gardenas et al., 2006; Ajdary et al., 2007; Foppen et al., 2007).

2.3 DRAIN FIELD NUTRIENT MOVEMENT

2.3.1 NITROGEN MOVEMENT

Nitrogen removal in the soil is mainly through the pathways of adsorption, fixation, volatilization, biological uptake, and denitrification (Figure 2.2) (Broadbent and Reisenauer, 1988; Laak, 1982; Lance, 1972; Petrovic, 1990; Tyler et al., 1977; US EPA, 2002). Venhuizen (1995) did an extensive review of nitrogen fate in the soil and concluded that much of the ammonium will be nitrified after entering the soil. Even though nitrification most readily occurs when the soil is warm, few sections of Alabama stays cool enough to prevent nitrification and subsequent loss of N through leaching and nitrification (Reiter, 2003; Scarsbrook and Cope, 1958). Unless conditions are favorable for denitrification, such as saturated soil or no air circulation, nitrate will tend to stay in the soil and become a potential nitrate pollution source. Nitrate is of negative charge which makes it more mobile in soils than ammonia (US EPA, 2002), Furthermore, since most ammonia nitrogen can be easily nitrified into nitrate, the chances of observing nitrate leaching for a drain field is greater than that of ammonia, (US EPA, 2002).

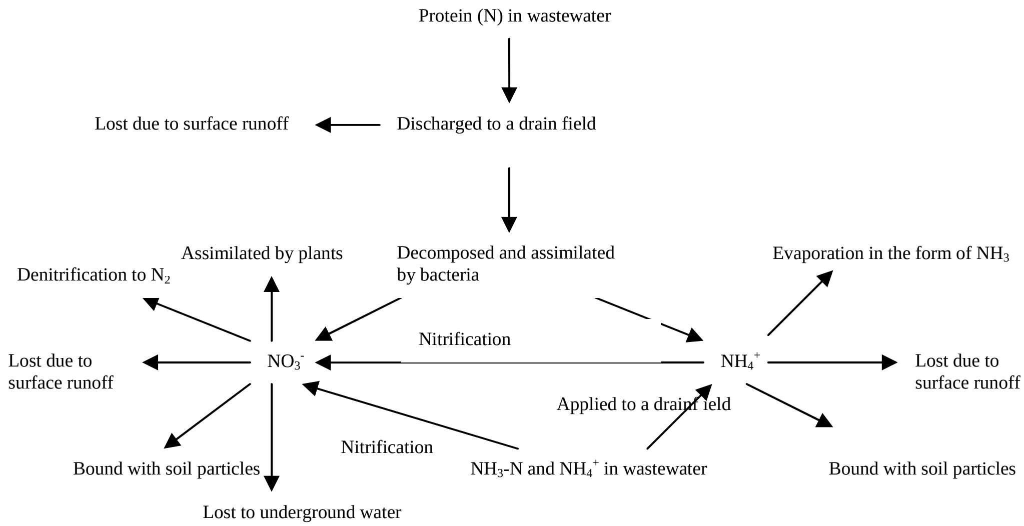

Figure 2.2 Nitrogen movement and fate in wastewater drain field.

Ammonia fixation occurs when ammonium ions become trapped in the intermicellar layers of clay minerals and fixation by organic components of the soil may also occur (Laak, 1982; Lance, 1972; Tyler et al., 1977). However, the potential for ammonium fixation is limited and not likely to be important in the long-term nitrogen budget of a soil system (Tyler et al., 1977; Broadbent and Reisenauer, 1988).

Tyler et al. (1977) and Lance (1972) both mentioned that significant volatilization losses require considerable air and water contact, which is unlikely during subsurface wastewater disposal, since wastewater will be adsorbed quickly by soil particles for subsequent nitrification. Also, the equilibrium between ammonium and ammonia gas is pH dependent. Significant volatilization can only be expected at a fairly high pH. Comparably low pH in the soil will not work in favor of volatilization of ammonium to ammonia gas (Lance, 1972). Consequently, nitrogen loss through non-biological volatilization of ammonium to ammonia gas is believed to be an insignificant factor for subsurface wastewater disposal systems (Venhuizen, 1995).

Soil adsorption has been proven to be the major mechanism in the removal of large quantities of ammonium from solution, (Broadbent and Reisenauer, 1988; Lance, 1972; Tyler et al., 1977). However, since ammonium is most easily nitrified under aerobic conditions, significant adsorption will not typically occur after nitrification due to the weak affinity between nitrate and soil particles (Venhuizen, 1995).

Biological uptake and denitrification are also important pathways for applied or disposed nitrogen to be removed from a soil system (Venhuizen, 1995). Nitrogen is absorbed by plants primarily as ammonium and nitrate, but can also be absorbed as urea ($Co(CN_2)_2$) (Scarsbrook and Cope, 1958; Tisdale et al.,1993). Tyler et al. (1977) mentioned that extensive root systems are not present in effluent drain fields and therefore only limited amounts of N would be taken up. Also, in commenting on the general uptake potential of plants, Laak (1982) noted that wastewater from subsurface effluent drain fields is not as available to plants due to rapid leaching loss. However, some leached nitrogen can be scavenged if roots are deep. Although not all of the nitrogen applied to the soil can be removed by plant uptake, 50% or less nitrogen uptake rate by perennial grasses is generally expected (Broadbent and Reisenauer, 1988).

Lance (1972) mentioned three conditions that are necessary for denitrification to proceed: (1) oxidation of ammonium to nitrate, since ammonia will not be transformed directly to nitrogen gas; (2) passage through an anaerobic zone after nitrification has occurred, when nitrate is ready for denitrification; and (3) provision of an adequate energy source for denitrifying bacteria in the anaerobic zone, which means a carbon source is required. The potential total nitrogen reduction by denitrification in land applied wastewater effluent has been estimated at 10-15% (Lance, 1972) and 15-25% (US EPA, 1977). Broadbent and Reisenauer (1988) noted that temporary anoxia can be created in the soil by a passing wetting front. Venhuizen (1995) stated that since effluent is intermittently dosed in a low pressure disposal system, alternating cycles of higher saturation (anoxia) and lower saturation (oxic) will be created. As a result, in an intermittent wastewater disposal strategy, alternating nitrification and denitrification will occur (Beggs et al., 2004; Venhuizen, 1995).

2.3.2 PHOSPHORUS MOVEMENT

The mechanisms for removing phosphorus removal from the soil include plant uptake, biological immobilization, adsorption, chemical precipitation, surface and subsurface runoff (Reneau et al., 1989). These mechanisms are complex and have not been completely understood (Venhuizen, 1995). Crops uptake of N: P is approximately 8:1 (Sharpley, 1996) which often leaves P in surplus in animal and human waste land disposal (Faulkner, 2001).

For wastewater applied phosphorus, it is uncertain whether phosphorus is adsorbed or precipitated in any given instance unless the soil is analyzed (US EPA, 1977). Removal and immobilization of phosphorus is dependent on availability of adsorption sites to bind phosphorus, with the majority of sorption sites provided by clay and organic soil fractions (Venhuizen, 1995). The phosphorus removal process typically starts with a relatively fast adsorption reaction, followed by slower immobilization due to the formation of low solubility precipitates (Sawhney and Hill, 1975). Roose and Fowler (2004) found that phosphorus remains relatively immobile in the soil once it is adsorbed onto soil particles, even under large scale water movement. Although adsorption is theoretically limited, soils generally have a greater capacity for phosphorus retention than is predicted by adsorption theories, since adsorption sites "regenerate" as precipitation proceeds (Broadbent and Reisenauer, 1988; Canter and Knox, 1985; Sawhney and Hill, 1975; Tyler et al., 1977; US EPA, 1977; Venhuizen, 1995).

Phosphorus sorption capacities vary in different soils and disposal systems which will impact the length of time required to exhaust the phosphorus adsorption capacity to any given depth (Sawhney and Hill, 1975; Venhuizen, 1995). The contact area of soil and wastewater should be increased in order to fully utilize the adsorption capacity of the applied drain field (Sawhney and Hill, 1975). Organic matter also has the potential to reduce phosphorus loss by surface runoff by offering additional adsorption sites for phosphorus (Roberts and Clanton, 1991). Sawhney and Hill (1975) stated that deeper soil layers generally have a lower phosphorus sorption capacity due to decreased cation exchange capacity (CEC) and organic matter content. Consequently, phosphorus removal should be completed within a certain depth of wastewater dosing locations, otherwise phosphorus leaching may occur.

2.4 Integrated Cropping System

2.4.1 Water and Nutrient Management of Cropping Systems

Depending on the specific objective, a well planned cropping system can either reduce nutrient loss between seasons (Askegaard and Eriksen, 2008), increase long term soil fertility (McNair Bostick et al., 2007; Jagadamma et al., 2007), or control negative environment impact from fertilizer application (Ferraro et al., 2003; Giupponi, 1997; Zhu et al., 2005).

A cropping system may be defined as a community of plants which is managed by a farm unit to achieve various human goals, which include food, fibre, other raw materials, wealth and satisfaction (FAO, 1995). The international Rice Research Institute (IRRI) (1978) gives a more detailed definition of a cropping system: "....the crop production activity of a farm. It comprises all cropping patterns grown on the farm and their interaction with farm resources, other household enterprises and the physical, biological, technological and sociological factors or environments".

Besides providing agronomic value, research in cropping systems also contributes to irrigation management (Wang et al., 2008), nutrient control (Askegaard and Eriksen, 2008; Colomb et al., 2007; Jagadamma et al., 2007; Ju et al., 2006; Liu et al., 2003; Ohno et al., 2005; Verma and Sharma, 2008; Wright et al., 2007; Zhu et al., 2005; Zougmore et al., 2004), micronutrient control (Wright et al., 2007), and increased soil organic matter (Hamza and Anderson, 2005; McNair Bostick et al., 2007; Jagadamma et al., 2008). The environmental impact of cropping systems with regard to management of pesticides control has also been studied (Ferraro et al., 2003; Giupponi, 1998).

Wang et al. (2008) studied summer forage cropping as an effective way to control deep drainage in a sandy clay loam in southern Australia. Summer forage cropping increased evapotranspiration (ET) 40% and lowered soil water content 53%. Drainage loss was reduced 62% at the cost of a 13% yield loss in the next winter wheat yield.

Shan et al. (2005) studied nutrient control and management in cropping systems through a 40-year rice growing test in a silty sandy clay loam site in China. They found a persistent downward movement of phosphorus , but did not show a correlation between phosphorus leachate and phosphorus fertilizer application. Higher phosphorus fertilization application rates did result in accumulation of phosphorus in the top soil. Ram et al. (2006) showed that adequate soil moisture maintained during summer provided an effective environment for efficient water and nutrient uptake for menthol mint in a sandy loam soil in a semi-arid, subtropical climate. Fuentes et al. (2003) was surprised to observe nitrate accumulated in a deep soil at an arid test site. The expectation was that water consumption was relatively high so nutrient movement downward would be relatively low. Zougmore et al. (2004) showed that water and nutrient management can help to control nitrogen loss from a field in a semiarid sandy loam site, where soil erosion and surface runoff were the main causes of nutrient loss. Zhu et al. (2005) studied nitrogen transport under an excessively fertilized sandy loam with hot peppers, a crop that provides low nitrogen use efficiency. They indicated that unplanned fertilizer applications creates the potential for atmosphere and underground water pollution. Further emphasis was laid on the importance of choosing an appropriate cropping system to match fertilizer application in order to minimize the adverse environmental impact from fertilizer over application.

Jagadamma et al. (2007) studied the effects of nitrogen fertilization and cropping systems on soil organic carbon and total nitrogen pools in a 32-year field experiment. The results showed the top 30 cm of soil organic carbon was significantly increased by fertilization and cropping systems, while soil bulk density was decreased. However, no significant correlation was found between soil properties, fertilization rate, and cropping systems below 30 cm soil depth. Wright et al. (2007) studied nutrient stratification for 5 years on a silty clay loam site in Texas. Results indicate that crop rotation increased micronutrient levels relative to monoculture cropping in unfertilized soils. Phosphorus stratification was found to be significant with the top soil at higher phosphorus levels than lower layers. Decreasing plant-available nutrient concentrations with increasing N fertilizer rates was also found as a result of increased crop yields that ultimately led to higher nutrient removal by crops.

McNair Bostick et al. (2007) demonstrated that soil organic carbon sequestration in the top 20 cm was increased through a 10-year cropping system on loamy sand. Rotation cropping systems tested were Sorghum-Sorghum-Sorghum, Groundnut-Sorghum-Cotton, Groundnut-Cotton-Sorghum, Cotton-Cotton-Cotton, Groundnut-Groundnut -Groundnut, Cotton-Maize-Sorghum, Sorghum-Fallow, and Continuous fallow. Colomb et al. (2007) studied phosphorus management under different crop rotation systems in a 36-year field experiment on a silty-clay, clay soil. The site was used for a variety of other studies including crop response to water availability, irrigation, and nitrogen. Their conclusion was that selective cropping systems can help to minimize the transport of phosphorus from soil to water thus preserving global resources of phosphorus.

Ju et al. (2006) studied nitrogen balance and groundwater nitrate contamination on a loamy silty alluvial soil under three intensive cropping systems (wheat-maize, greenhouse vegetable, and apple orchards). They concluded that farm management should be improved from time to time based on field nutrient requirements so as to lower the accumulation of nitrate in soils and groundwater while maintaining or improving agricultural productivity. Tonitto et al. (2006) studied 36 published metal and/or nutrient related agricultural studies and concluded that diversified cropping systems have the potential to provide adequate nutrition for an increased population, as well as insure that agricultural practices do not contaminate potable water supplies or other food sources such as regional fisheries.

2.4.2 SOIL MOISTURE CONTROLLED IRRIGATION

Soil moisture controlled irrigation supplies plant water requirement indirectly by sensing water status of the soil matrix. Soil moisture control systems manipulate irrigation frequency and amount according to actual soil conditions throughout the growing season. This automated technology provides not only a convenience for the operator but also substantial water savings compared to conventional irrigation management that is based on current or historical weather (Dukes and Scholberg, 2005). Phene and Howell (1984) first used a custom made soil matric potential sensor to control SDI for processing tomatoes. Phene et al. (1992) used an evaporation pan to estimate ET and used the information as an irrigation scheduling tool for cotton. As electronic technology has developed, Granular Matric Sensors (GMS) and Time Domain Reflectometry (TDR) sensors have come into use for automated irrigation scheduling (Meron et al., 1996; Muñoz-Carpena et al., 2003). By using TDR to limit the soil moisture level to around field capacity, Dukes and Scholberg (2005) reported at least 11% water savings and 24% reduction in deep percolation. Furthermore, they demonstrated that a smaller high/low soil moisture operation window further increased the percentage of applied water taken up by crop.

Beggs et al. (2004) modeled water and nitrogen movement under a time controlled SDI and suggested the potential for improved nitrogen removal if soil moisture content and nitrification/denitrification reaction rates were coupled. Their proposed system would create an optimized wastewater disposal strategy based on a synchronized aerobic/anaerobic cycle for maximum nitrogen reduction (Venhuizen, 1995).

2.4.3 WATER AND NUTRIENT BALANCE STUDIES

Water provides the medium for transport of plant nutrients and other growth regulatory compounds and is fundamental to the maintenance of normal physiological activity and membrane transport process (Jones and Tardieu, 1998). Water budgets during agricultural water management have been extensively studied for both nutrient management (Alva et al., 2006; Guo et al., 2008; Kibe et al., 2006; Ram et al., 2006; Rego et al., 2008) and water conservation (Abid Karray et al., 2008; Berg and Driessen, 2002; Cameira et al., 2005; Eitzinger et al., 2003; Ji et al., 2007; Jones and Tardieu, 1998; Loos et al., 2007;).

Berg and Driessen (2002) reviewed approaches that relate soil water potential to crop yield. They reported that current approaches to estimate or calculate crop water needs are focused on soil water potential, relative soil water content, and evaporative demand. Water balance components proposed by several studies are listed in Table 2.4. To quantify each component of the water balance, focus was placed on determining soil evaporation and crop transpiration or combined as evapotranspiration (ET) since the other components of the water balance can be directly measured with adequate resources (Abid Karray et al., 2008; Cameira et al., 2005; Jones and Tardieu, 1998; Loos et al., 2007).

Nutrient budget calculations are often coupled to a water budget in the calculation of nutrient input, runoff, and leaching loss. (Abid Karray et al., 2008 ; Berg and Driessen, 2002; Cameira et al., 2005; Eitzinger et al., 2003; Ji et al., 2007; Jones and Tardieu, 1998; Loos et al., 2007). Nutrient budget components taken from selected studies are listed in Table 2.5. Due to the fact that sampling leaching sample is labor intensive (Alva et al., 2006; Rego et al., 2003), prediction methods were used when no direct measurement is available. The prediction method used to obtain the nutrient leaching loss in literature is by Darcy’s law to estimate of water flux, multiplied by soil extractable nutrient concentration (Alva et al., 2006; Rego et al., 2003).

Table 2.4 Water budget components. (Based on selected studies)

	Rain fall	Irrigation	Soil evaporation	Crop transpiration	Deep percolation	Surface runoff	Soil water content change
Arnold et al. (1995)	X	X	X	X	X	X	X
Loos et al. (2007)	X	X	X	X	X		X
Karry et al. (2008)	X	X	X	X	X		X

Table 2.5 Nutrient budget components. (Based on selected studies)

	Fertilizer	Mineralization	Atmospheric deposition	Non-symb. N fixation	Plant uptake	Surface runoff	Soil nutrient change	Gaseous loss	Water erosion	Deep percolation
Alva et al. (2006)	X	X	X	X	X		X			X
Guo et al. (2008)	X				X		X			
Rego et al. (2008)	X	X	X	X	X	X	X	X	X	X

CHAPTER THREE

POTENTIAL ENVIRONMENTAL AND PUBLIC HEALTH RISK FROM ONSITE WASTEWATER TREATMENT SYSTEMS IN THE ALABAMA BLACKLAND PRAIRIE SOIL AREA

3.1 ABSTRACT

The Alabama Black Belt, a 14000 km^2 area of widespread clayey soils that make up part of the larger Blackland Prairie soil area in central Alabama and eastern Mississippi, presents an environmental challenge for onsite wastewater treatment systems (OWTS) that rely on soil for contaminant dispersal. In this study, a new OWTS soil suitability rating system (OWTS-SSRS) was developed as an interpretation of the current Alabama OWTS regulations. The new soil rating system was applied to the Alabama Black Belt area using soil information extracted from USDA-NRCS digital soil survey data (SSURGO). The existing NRCS soil limitation rating system (NRCS-SLRS) was provided as an available nationwide assessment tool of the study area. The OWTS-SSRS rated approximately 52% of the Alabama Black Belt area as "Unsuitable" for OWTS, while the existing NRCS soil limitation rating system rated 89% of the study area as "Limiting" for OWTS. Spatial analysis of results indicates that the new OWTS-SSRS is less conservative than the NRCS rating system. Demographic analysis based on US Census 2000 data reveals that rural areas represent approximately 95% of this region with

an average house age of over 20 years. Subsequent raster-based analysis was used to prioritize the environmental and health risk in the region using a weighted combination of OWTS-SSRS results and US Census derived demographics. City fringes were found to be generally at higher risk from OWTS than rural areas, suggesting the following two mitigation strategies in the Alabama Black Belt region. For isolated rural households outside the range of municipal sewer service, retrofit or replacement of aged OWTS with more advanced dispersal systems is recommended. For certain city fringe communities at high risk, proactive extension of municipal sewer service is recommended in advance of widespread OWTS malfunction. Findings of this study indicate the need for continued development of planning and assessment tools to protect vital soil and water resources and public health in the Alabama Blackland Prairie soil area.

3.2 INTRODUCTION

Onsite wastewater treatment systems (OWTS) currently disperse 15 million metric tons of wastewater per day, serving about 30% of households in the US (Spicer, 2002; AOWTC, 2005) and functioning as an important supplement to centralized public wastewater treatment plants (WWTP). Despite wide adoption, conventional OWTS consisting of a septic tank and a gravity fed effluent dispersal field (where effluent is treated) can pose a significant threat to the environment (US EPA, 2002).

Drain field nutrient overload leading to nonpoint source pollution is a recognized source of environmental and health risk from OWTS (US EPA, 2002). Typical nitrogen concentrations in septic tank effluent range from 40-80 mg L^{-1} (Walker et al., 1973), of which approximately 75% is ammonium nitrogen and 25% is organic nitrogen (Otis et al., 1974). Reported total phosphorus concentrations in septic tank effluent range from 3 to 20 mg L^{-1} (Robertson et al., 1998; Whelan, 1988), with about 85% as orthophosphate (Reneau and Pettry, 1976). Charles et al. (2005) analyzed the results of several intensive septic tank effluent field surveys from 1976 to 1999 in Australia and the US, including their own field survey of 200 convential OWTS in Australia. They compared results to published regulations in both countries and concluded that prevailing septic tank effluent quality estimates of drain field nutrient load (50-60 mg TN L^{-1} and 10-15 mg TP L^{-1}) are too low to ensure safe environmental performance and may underestimate actual drain field nutrient loads. Charles et al. (2005) concluded that the 80th percentile of their effluent survey values, 250 mg TN L^{-1} and 36 mg TP L^{-1} should be used to estimate the nutrient loads associated with conventional OWTS operation. Their research concluded that nutrient overloading may be occurring in a significant number of conventional OWTS designed under current Australian and US regulations (Charles et al., 2005).

Septic tank effluent can contain a significant number of pathogens that are a potential threat to local and regional environmental health if not properly controlled. Although conventional OWTS can provide 50-90% removal of pathogens (Gerba and Goyal, 1985; Siegrist, 2007), concerns exist on how far those pathogens can move and how long they can survive after they have entered the soil-water system (Jamieson et al. 2002). It is recognized that physical filtration of soil media is the main mechanism limiting pathogen travel (Canter and Knox, 1985; Hagedorn et al., 1981) and that removal efficiency is typically inversely proportional to soil particle size (Tanik and Comakoglu, 1997; Venhuizen, 1995). Nevertheless, despite reports that significant pathogen removal in soil can be achieved within a relatively short distance under proper OWTS management (Tyler et al., 1977; Converse et al., 1991), pathogen translocation up to 830 m in malfunctioning OWTS is not rare (Venhuizen, 1995; Dowd et al., 1998; Schijven and Hassanizadeh, 2000; Charles et al., 2004). Furthermore, since soil moisture sustains pathogen survival, even pathogens retained by soil media still remain a threat until they die off (Venhuizen, 1995). In the worst case, as a malfunctioning OWTS loses its nutrient and pathogen removal capability, the environmental threat gradually expands beyond the drain field (Carroll and Goonetilleke, 2005; Charles et al., 2007).

It is currently understood that although the lot size of an OWTS is designed to achieve adequate removal of nutrients and pathogens within a fairly short length of soil (Frankenberger, 1988; Green and Cliver, 1975; Venhuizen, 1995), the cumulative impact of large numbers of OWTS in a locality can create a threat to both the local environment and public health (Carroll and Goonetilleke, 2005). The spatial density of OWTS influences surface and ground water environmental quality due to 1) increased probability of system malfunction and 2) cumulative nutrient and pathogen load that exceeds the capacity of local soils to assimilate (US EPA, 1977; US EPA, 2002). Literature provides the following environmental impact density classes; low (1-4 unit km^{-2}), medium (5-15 unit km^{-2}), high (16-100 unit km^{-2}), and extremely high (> 100 unit km^{-2}) (US EPA, 1977; Yates, 1985). Carroll and Goonetilleke (2005), using principal component analysis (PCA) on temporally and spatially monitored well-water systems in Queensland, Australia, confirmed that a system density of 290 units km^{-2} significantly increased both nutrient and pathogen concentrations in shallow groundwater systems.

Conventional OWTS have been used in the Alabama Black Belt area for decades in spite of the fact that many soils in this region, although rich agriculturally, have a high smectitic clay content with severe hydraulic limitations. Soil hydraulic limitations to wastewater absorption include low permeability, high seasonal water table, presence of restrictive layers, and likelihood of flooding. According to the Geological Survey of Alabama (1993), Alabama Black Belt soils are underlain at a general depth of approximately 6 m, with shallower formations found at 12 cm to 2 m, by a relatively impermeable layer of fossiliferous clayey chalk and chalky marl to a depth of approximately 122 m. Below that are the Eutaw and Tuscaloosa aquifers, the only significant groundwater sources in the Alabama Black Belt region. When top soil layers become saturated, the low permeability of the underlying chalk limits deep percolation to underground aquifers. Thus, surface ponding and runoff from conventional OWTS drain fields is the more common environmental and health concern from malfunctioning OWTS in the Alabama Black Belt.

Since the 1950s, over 10 million acres of row crops have been abandoned in Alabama (McNider et al., 2005), much of it in the Black Belt region. As a result, rural economies in this region have suffered and many existing conventional OWTS are in disrepair. The weakened economy in the Alabama Black Belt makes the retrofit of individual OWTS cost-prohibitive for many households (McCoy et al., 2004). In 2002, in Lowndes County, an Alabama court ruled against 37 families who discharged raw wastewater into their backyards and nearby ditches, drawing national attention to onsite system deficiencies in the Alabama Black Belt (McCoy et al., 2004). According to the US EPA (2002), approximately 44% of the houses in Alabama are served by OWTSs with an average system malfunction rate of 20%. The relatively high OWTS usage rate in Alabama along with the geographical, economical, and political uniqueness of the Black

Belt area indicate the need for an assessment of general soil conditions for wastewater dispersal in the region.

The present study uses spatial analysis to assess environmental and health risk of conventional OWTS in the Alabama Black Belt. The first objective was to indicate suitability of soils in the Alabama Black Belt for conventional OWTS.. A GIS based Soil Suitability Rating System (OWTS-SSRS) was developed as an interpretation of the current Alabama Onsite Sewage Disposal Rules (ADPH, 2006) over the soils within the Alabama Black Belt area using Soil Survey Geographic (SSURGO) digital soils data (NRCS, 2005). The existing Natural Resources Conservation Service soil limitation rating system (NRCS-SLRS) for septic tank absorption fields was presented as an availble national assessment of the study area. The second objective was to rate the Black Belt study area with respect to conventional OWTS environmental and health risk using spatial results from the new OWTS-SSRS and derived demographics such as OWTS age, size, and density. Based on results, two strategies to mitigate OWTS related environmental and health threats in the Alabama Black Belt are proposed.

3.3 Materials and Methods

3.3.1 Study Area Delineation

The Black Belt study area boundary was defined within a GIS (ArcMap 9.2, ESRI, CA) using two Common Resource Area (CRA) maps, spatial resolution 400 km^2, version 1.1 (NRCS, 2005), and the Alabama State Soil Geographic Database (STATSGO), spatial resolution 6.25 km^2, version 1.0 (NRCS, 2005). The two CRAs used were 135A.1 - Blackland Prairie and 135A.2 - Flatwoods/Blackland Prairie Margins. The majority of the study area lies within Blackland Prairie major land resource areas made up of the following Alabama STATSGO mapping units; AL119 - Vaiden-Sumter-Oktibbeha-Marietta, AL121 - Savannah-Ora-Faceville, AL143 - Vaiden-Sumter-Oktibbeha, AL147 - Vaiden-Minter-Kipling-Angie, AL148 - Vaiden-Sumter-Sucarnoochee-Kipling-Demopolis, AL166 - Sumter-Searcy-Oktibbeha-Demopolis-Congaree-Brantley, AL168 - Sumter-Oktibbeha-Luverne-Conecuh, AL173 - Vaiden-Sumter-Leeper, AL238 - Sumter-Rock outcrop-Oktibbeha-Kipling-Demopolis-Binnsville, and AL239 - Vaiden-Okolona-Kipling.

In an effort to delineate a continuous study area across central Alabama, non-continuous CRAs were redefined to include narrow areas of CRA 133A.7 - Coastal Plain Floodplains and Low Terraces which bisect the study area. CRAs 135A.1 and 135A.2 were first overlaid and unioned with STATSGO delineations, then a 1500-meter buffer was applied to the union. Next, CRA 133A.1 - Southern Hilly Gulf Coastal Plain and CRA 133A.7 were overlaid and unioned with the generated 1500-meter buffer. The polygon from this delineation became the study area boundary (Figure 3.1). The resulting study area extends across central Alabama, including parts of Sumter, Greene, Pickens, Hale, Marengo, Perry, Dallas, Lowndes, Butler, Wilcox, Montgomery, Macon, Bullock and Russell counties, with a total area of 13981 km^2 and an estimated population of 394,000 (US Census, 2000).

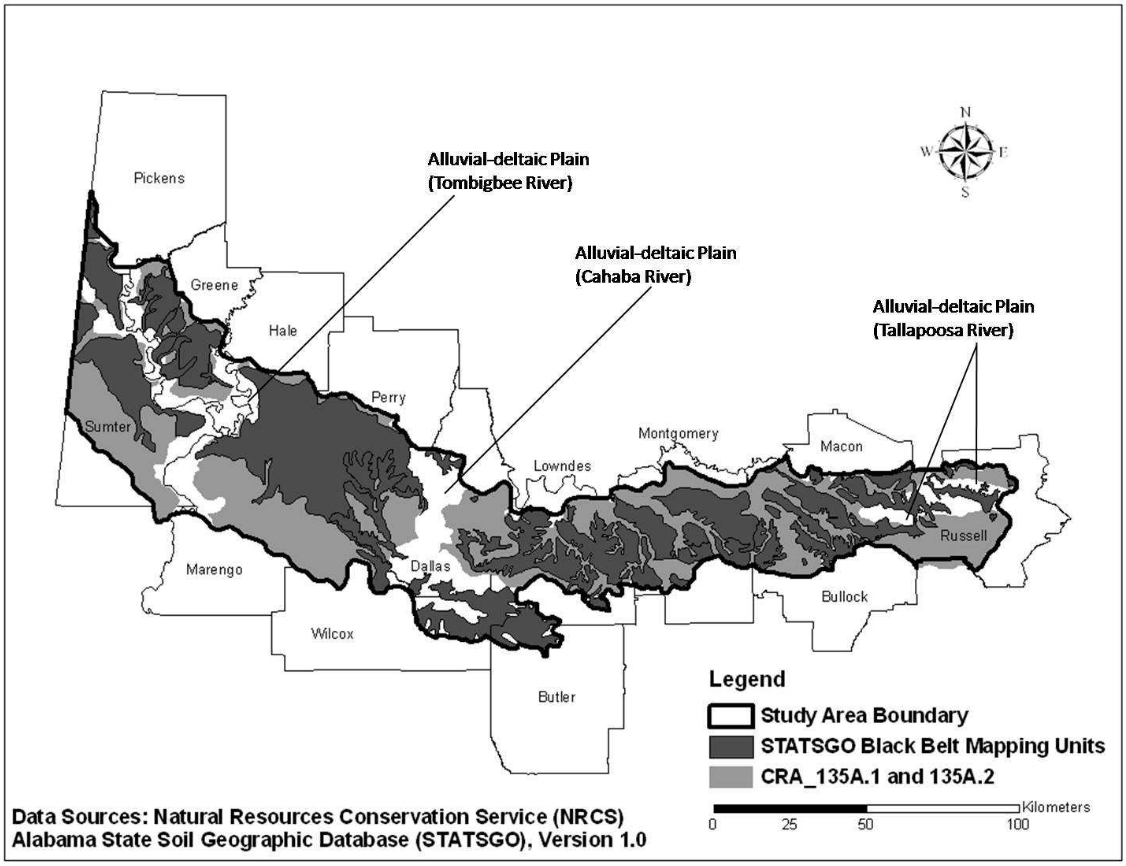

Figure 3.1 The Black Belt study area boundary showing STATSGO Black Belt mapping units overlaid with CRA 135A.1 (Blackland Prairie) and 135A.2 (Flatwoods/Blackland Prairie Margins) land resource area data.

3.3.2 NRCS SOIL LIMITATION RATING SYSTEM (NRCS-SLRS) FOR SEPTIC TANK ABSORPTION FIELDS.

Based on available county soil survey information, NRCS developed a national septic tank absorption field limitation rating system for guiding both heavily and sparsely populated areas in site selection for safe disposal of household effluent (NRCS, 1993 2007; USDH, 1969). The NRCS Limitation Ratings indicate that the soil has properties that may limit the functionality for the intended use and do not indicate whether the soil is unsuited for that use (NRCS, 1993). Thirteen soil and site condition criteria (Table 3.1) from SSURGO (NRCS, 2007) are considered in the NRCS soil limitation rating system

(NRCS-SLRS). Soil mapping units are rated as Limiting, Somewhat Limiting, or Not Limiting for OWTS siting based on the dominant soil series for each soil mapping unit. Dominant soil series is defined by NRCS as the associated soil series with the highest percentage within the mapping unit. Soil mapping units covered by water or otherwise inaccessible or undevelopable such as military areas are listed by NRCS as Not Rated.

In this study, the NRCS-SLRS result was extracted directly from digital SSURGO soil mapping units (spatial resolution 0.02 km^2) using Soil Data Viewer (Version 5.1, USDA) for spatial display within the study area. SSURGO data for Dallas and Lowndes Counties were not available at the time of analysis; therefore, soil information and rating system comparison for these two counties is not included.

Table 3.1 NRCS Soil Limitation Rating System (NRCS-SLRS) rating criteria for septic tank absorption fields.

Rating criteria / NRCS Ranking	Maximum K_{sat} in 60-150 cm (um s^{-1})	Flooding occurrence	Content of large stones	Minimum K_{sat} in 60 cm-restrictive layer (um s^{-1})	Depth to permafrost (cm)	Ponding duration	Depth to bedrock (cm)	Depth to cemented pan (cm)	Slope	Subsidence (cm)	Depth to saturation zone (cm)	Seepage (min cm^{-1})	Unstable fill
Not Limiting	≥41.6	"none"	<25%	<41.6	>100	others	≥182	≥182	<8%	<60	≥180	>12	Others
Somewhat Limiting	≥41.6	"very rare" or "rare"	25-50%	9.80-41.6	50-100	others	100-182	100-182	8-15%	<60	120-180	>12	Others
Limiting	<41.6	"occasional" or "very frequent" or "frequent	>50%	>9.80	<50	“very brief” or “brief” “long” or “very long”	≤100	≤100	>15%	≥60	<120	≤12	Unstable fill

Source: Derived from SSURGO database (NRCS, 2007).

3.3.3 NEW OWTS SOIL SUITABILITY RATING SYSTEM (OWTS-SSRS)

To develop a new soil rating system for subsurface gravity dispersal of septic tank effluent, each soil mapping unit from the SSURGO database (NRCS, 2007) in the study area was ranked for suitability according to current Site Evaluation Criteria from the Alabama Onsite Sewage Disposal Rules (ADPH, 2006). A rating of Suitable, Marginally Suitable, or Unsuitable (Table 3.2) was assigned to each mapping unit based on properties of the dominant soil series. These three ratings indicate whether or not an OWTS system can be saftely located on a particular site. In the SSURGO database, soil mapping units within military areas, water bodies, or other unavailable areas carry no soil information except for land area and thus were not rated. The following numeric code; (1) for Suitable, (2) for Marginally Suitable, (3) for Unsuitable, and (4) for Not Rated was applied to each soil mapping unit for subsequent spatial analysis and environmental and health risk ranking.

The OWTS-SSRS result was extracted directly from digital SSURGO soil mapping units in the study area for spatial display and further analysis. As in NRCS-SLRS, digital soil information for Dallas and Lowndes Counties is not reported because county soil survey information was not completed at the time of this study.

Table 3.2 New OWTS Suitability Soil Rating System (OWTS-SSRS) rating criteria.

Rating criteria / OWTS Ranking	Percolation rate from 60 cm depth to restrictive layer (min cm^{-1})	Depth to restrictive layer (cm)	Depth to seasonal water table (cm)	Slope (%)	Flooding
Suitable	<35	>91	>91	0-25	None, rare
Marginally Suitable	35-47	>91	>91	26-40	Occasionally
Unsuitable	>47	<91	<91	>40	Frequent

Based on: Site Evaluation Criteria, Alabama Onsite Sewage Disposal Rules (ADPH, 2006).

3.3.4 Onsite Wastewater Treatment System Status in the Alabama Black Belt Area

The most recent US Census does not provide individual household sewer disposal information. Due to legislative reasons, the last year such information was provided was 1990 (Department of Housing and Urban Development, US Census Bureau, personal communication, May 11, 2009). Consequently, OWTS demographics used for this study were derived from available 2000 US Census block group data, the second smallest mapping unit for which US Census data is available. The spatial resolution of the US Census data used for the study area was determined to be 1.70 km^2.

Census block groups in this study were categorized as rural if over 70% of the block group population was rated rural by the US Census. Otherwise, the block group was rated as urban. Households classed as urban were assumed to be connected to a public sewer system. The assumption was made that if a household fell within a rural block area, the household was served by an OWTS. The method used to extract US Census information did not distinguish between single- and multi-family units or commercial and industrial units, so may not accurately represent individual OWTS status in each Census block group.

The lifetime of an OWTS depends on its design, installation, and maintenance. If all other things were equal, a higher OWTS age suggests a higher probability of malfunction (US EPA, 2002). In the present study, the average house age in each rural block was used to represent average OWTS age. For example, an average house age of 20 years in a block group indicated an average OWTS age of 20 years. The ratio of total population to the total number of household units in a rural block group was used to represent the average OWTS size (person unit^{-1}) in a block group. The ratio of the number of household units to the area of the corresponding rural block group was used to represent average OWTS density (unit km^{-2}). A higher OWTS size and density suggests a higher septic effluent load. It has been demonstrated that the higher the average OWTS size or density in an area, the higher the probability of malfunctioning OWTS (US EPA, 1977).

3.3.5 Prioritizing Environmental and Health Risk Areas

An environmental and health risk map was developed to prioritize the potential risk from OWTS in the 14-county Black Belt study area. OWTS-SSRS ratings and corresponding metrics of OWTS age, size, and density from US Census data were normalized, weighted, and summed to indicate the spatial distribution of environmental and health risk in the study area. Data files containing OWTS-SSRS ratings, OWTS age, size, and density were each converted into a 141 m × 141 m raster dataset, then into numeric ASCII format. A total of 1053436 raster cells were used in the study area and subsequent analysis. The 141 m resolution was used rather than the 0.02 km^2 resolution of the SSURGO dataset because it represents the smallest mapping unit of the spatial datasets used in the study.

Since there are no direct field measurements of potential environmental and health risk of OWTS in the study area, the relative weighting of the four variables with respect to the potential of environmental and health risk from OWTS were assumed equal.

Thus, the potential environmental and health risk from a OWTS within each individual raster cell of the study area was calculated as shown in Eq. 3.1. The resulting values from Eq. 3.l over the entire study area were then normalized from 0 to 100 with higher values indicating a higher potential environmental and health risk.

Environmental and health risk rating = [OWTS density] + [OWTS-SSRS] + [OWTS size] + [OWTS age]……………………….Eq. 3.1

3.4 RESULTS AND DISCUSSION

3.4.1 *SOIL CONDITIONS OF THE ALABAMA BLACK BELT AREA.*

Based on analysis of the SSURGO database, 173 soil series occur within the Black Belt study area. Excluding missing Dallas and Lowndes counties, the fifteen most widely occurring soil series are listed in Table 3.3. Each soil represents at least 2% of the study area, and in combination 51% of the total study area. It is apparent that clayey soils dominate, with Conecuh, Kipling, Luverne, Sucarnoochee, Sumter, Vaiden, and Wilcox being the most prominent clay soil series. These widespread clay soils are generally unsuitable for conventional OWTS.

Mapped results from the newly developed OWTS-SSRS (Figure 3.2) indicate that approximately 52% of the study area (Dallas and Lowndes Counties excluded) is rated Unsuitable for conventional OWTS, 31% is rated as Marginally Suitable, and 15% is rated as Suitable. The remaining 2% land or water area is Not Rated. The corresponding NRCS-SLRS rating map (Figure 3.3) indicates that approximately 89% of the study area

Table 3.3 Major soil series in the Alabama Black Belt study area based on SSURGO 2007 database.

Soil series name	Area (km^2)	Percentage of total study area	NRCS description
Luverne	2786	13	Mixed, semiactive, thermic Typic Hapludults
Smithdale	1600	7	Fine-loamy, siliceous, subactive, thermic Typic Hapludults
Oktibbeha	702	3	Very-fine, smectitic, thermic Chromic Dystruderts
Conecuh	702	3	Fine, smectitic, thermic Vertic Hapludults
Sumter	642	3	Fine-silty, carbonatic, thermic Rendollic Eutrudepts
Sucarnoochee	579	3	Fine, smectitic, thermic Chromic Epiaquerts
Mantachie	515	2	Fine-loamy, siliceous, active, acid, thermic Fluventic Endoaquepts
Kinston	509	2	Fine-loamy, siliceous, semiactive, acid, thermic Fluvaquentic Endoaquepts
Kipling	483	2	Fine, smectitic, thermic Vertic Paleudalfs
Vaiden	460	2	Very-fine, smectitic, thermic Aquic Dystruderts
Mooreville	420	2	Fine-loamy, siliceous, active, thermic Fluvaquentic Drystrudepts
Demopolis	417	2	Loamy, carbonatic, thermic, shallow Typic Udorthents
Savannah	406	2	Fine-loamy, siliceous, semiactive, thermic Typic Fragiudults
Wilcox	396	2	Very-fine, smectitic, thermic Chromic Dystruderts
Halso	323	2	Fine, smectitic, thermic Vertic Hapludults
Subtotal	10940	51	
Remaining area*	10565	49	
Total	21505	100	

Note: 1. Dallas and Lowndes counties excluded due to unavailability of SSURGO data.
2. Includes remaining 1580soil series, water body, and military areas.

is Limiting for conventional OWTS, 8% is Somewhat Limiting, 1% is Not Limiting, and the remaining 2% is Not Rated. Approximately 5% of the area that is rated as Limiting by NRCS-SLRS is rated as Suitable for conventional OWTS by the new OWTS-SSRS, while 43% of the area that is rated Suitable by the OWTS-SSRS is rated Not Limiting by NRCS-SLRS.

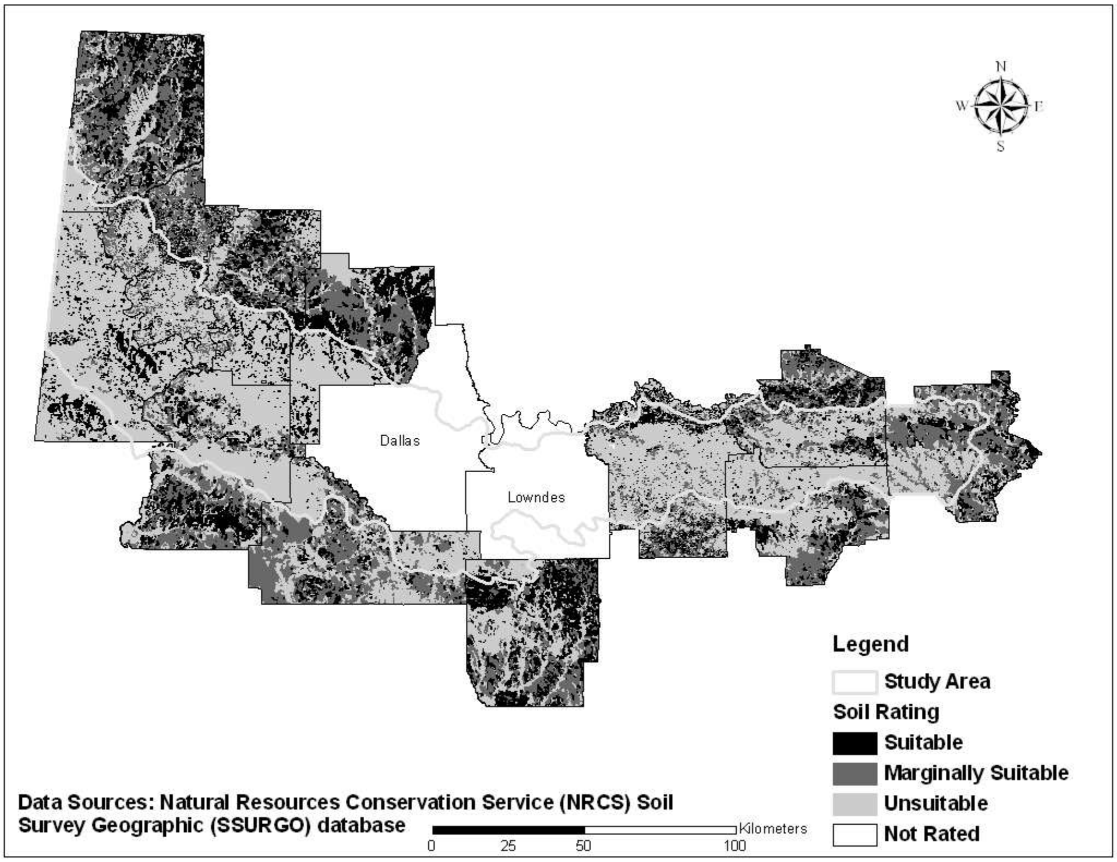

Figure 3.2 Soil rating results of the new OWTS Soil Suitability Rating System (OWTS-SSRS), Alabama Black Belt. (Dallas and Lowndes Counties excluded due to unavailability of SSURGO data)

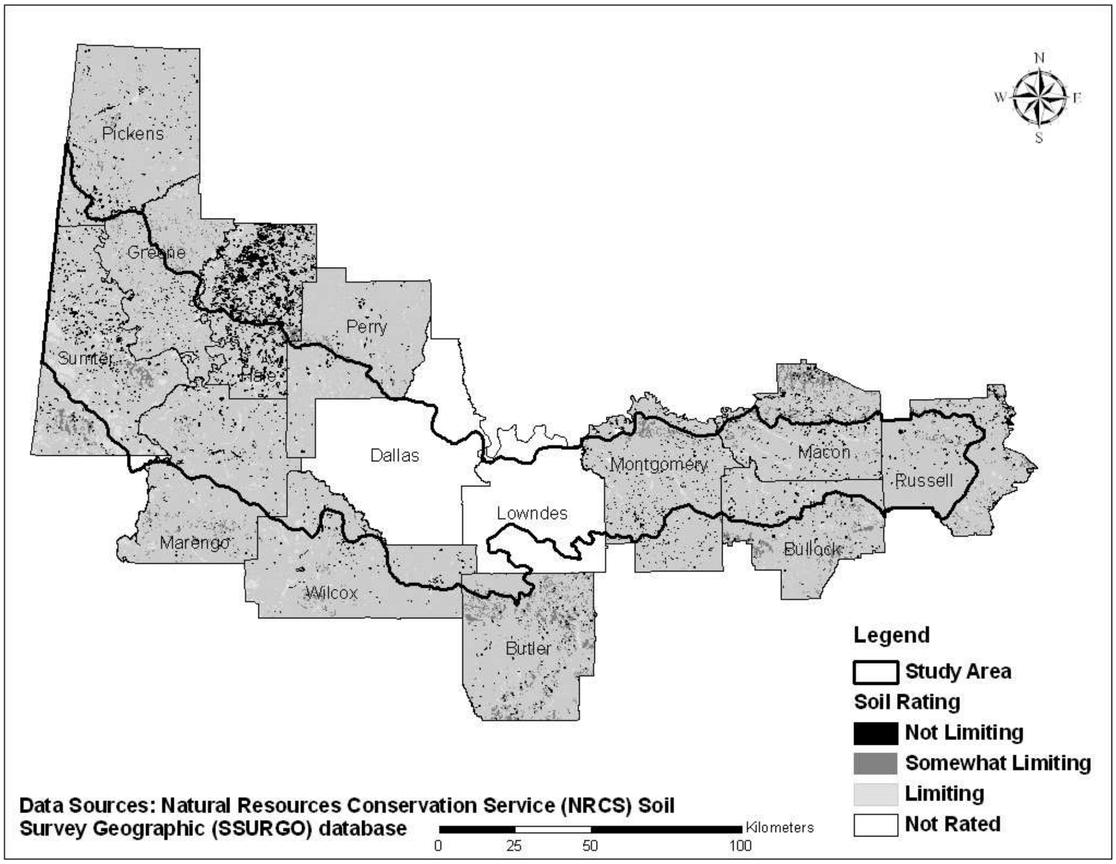

Figure 3.3 Soil rating results of the NRCS Soil Limitation Rating System (NRCS-SLRS) for septic tank absorption field, Alabama Black Belt. (Dallas and Lowndes Counties excluded due to unavailability of SSURGO data)

Although the two soil rating systems share some similar site rating criteria, such as depth to restrictive layer, depth to seasonal water table, slope, and flooding frequency, the major difference is that they use two different measurements to characterize water movement in the soils. The NRCS-SLRS uses K_{sat} while the OWTS-SSRS uses percolation rate. Even though statistic relationships were developed between K_{sat} and percoaltion rate (Fritton et al., 1986), it should be noted that K_{sat} is different from percolation rate and not directly comparable. K_{sat} is a measurement of how fast water can pass through a one-dimensional saturated soil medium under one unit hydraulic gradient, while percolation rate is a three-dimentional infiltration measurement for a generally unsaturated soil under a variable hydraulic head (SSSA, 2002b). Therefore, the two soil rating systems can not be quantitatively compared.

Furthermore, it is important to note when attempting to compare the two rating systems that the term “Limited” does not equate to the rating “Unsuitable.” Although both soil rating systems are based on long term field experience (NRCS, 1993; USDH, 1969; ADPH, 2006), the NRCS-SLRS was developed for national level guidance on sites being evaluated for a wide range of potential land uses, while the newer OWTS-SSRS was based on a specific environmental rule (ADPH, 2006) that regulates the design and permitting of conventional OWTS within the state of Alabama. The NRCS Limitation Ratings indicate soil properties that may limit the functionality for the intended use, but do not indicate whether the soil is unsuited for that use. Conversely, the “Unsuitable” rating indicates that the soil limitations are so severe as to make the soil unsuitable for the intended use. Soils identified as “Limited” in the NRCS system could fall in either the ”Suitable”, “Marginally suitable” or “Unsuitable” categories in the new Suitability Rating system.

Despite the differences between the two soil rating systems, the general findings from both systems is that a large percentage of land within the Alabama Black Belt is not recommended for conventional OWTS. This situation calls for alternative systems that can function properly on clayey soils. To make the developed OWTS-SSRS more beneficial for the Alabama Black Belt, the new OWTS-SSRS could be expanded to include ratings for alternative engineered systems such as aeration treatment units, packed-bed media filters, mounds, or subsurface drip irrigation. Engineered systems such as these can exert less burden on native soils for contaminant dispersal by providing secondary in-line wastewater treatment, additional treatment media, or improved dosing strategies. Although beyond the scope of the present work, an expanded soil rating system that includes advanced engineered systems would benefit regional decision makers in their evaluation of decentralized versus centralized systems in the Black Land Prairie soils area.

3.4.2 Current Status of Conventional Onsite Wastewater Treatment Systems

Spatial analysis of US Census block groups in the study area indicates that rural (non-urban) areas represent approximately 95% of the Alabama Black Belt area. The average size and density of OWTS in rural block groups is illustrated in Figure 3.4. Approximately 12% of rural block groups have an estimated OWTS density higher than 15 unit km^{-2}, posing a potentially high risk to the environment (US EPA, 1977). These 12% at-risk rural block groups are found generally clustered around city fringes such as Montgomery, Selma, and Uniontown, with the highest single block group OWTS density (212 unit km^{-2}) found at Uniontown. In 2004, the city of Uniontown in Perry County was awarded $350,000 from the Alabama state government to subsidize a public sewer service package to eliminate malfunctioning septic tanks on the south side of town and extend sewer service to 74 households. This project benefitted more than 200 households (Alabama Department of Economic and Community Affairs, 2004). Uniontown currently has a lagoon system serving approximately 2000 households that treats between 0.5 and

1.0 MGD of public wastewater. The municipality is currently seeking resources to expand its public sewer service to benefit more residences (Uniontown Public Works Department, personal communication, May 27, 2009).

OWTS size in the study area ranges from 1 to 18 persons system^{-1}, with approximately 73% of rural block groups having an OWTS size of 2 persons system^{-1} or less. Approximately 99% of rural block groups have an estimated OWTS size lower than 3 persons system^{-1}. The remaining 1% of rural block groups that have an estimated OWTS size greater than 3 persons system^{-1} are found around the city fringe of Montgomery, a major urban center in the Black Belt area and the capital of the state (Figure 3.4). The finding that rural block groups in close proximity to urban areas have high OWTS density and size is expected given the higher population concentrations in these urban and surrounding fringe areas.

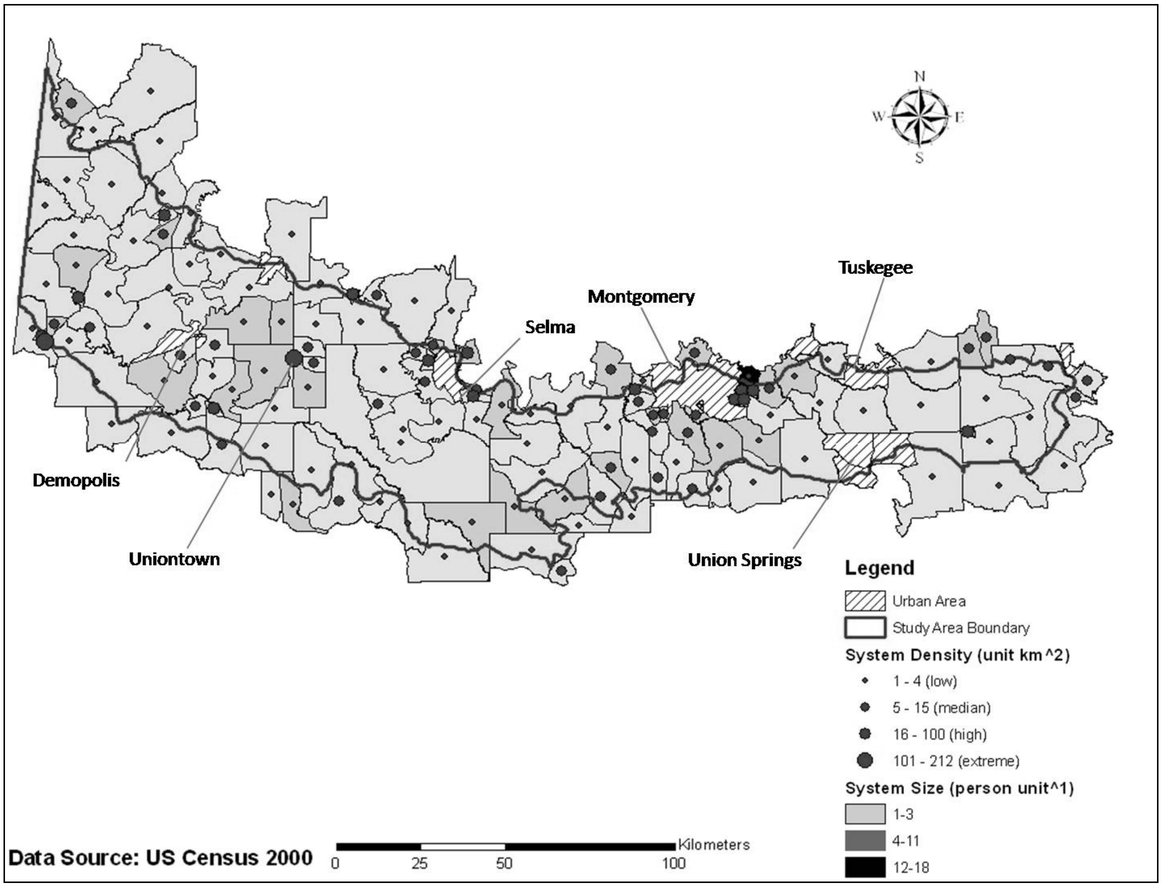

Figure 3.4 Average OWTS density and size in rural block groups, Alabama Black Belt.

The estimated home age in rural census block groups of the study area (data not shown) indicates that approximately 97% of rural block groups in 2000 had an average house age over 20 years; and approximately 93% among them had an average house age between 20 to 30 years. In 1990, Alabama was ranked fifth in states that rely on OWTS for household wastewater treatment and dispersal (44% households relying on OWTS compared to the highest, Vermont, with 55%) (US EPA, 2002). Maintenance of individual septic systems has already been recognized as a financial difficulty for many households in the Alabama Black Belt (McCoy et al. 2004), a burden that is increased due to widespread aging of conventional OWTS.

OWTS status in the study area is summarized as follows: 1) OWTS are widely used in the Alabama Black Belt; 2) a significant number of existing OWTS in the study area have been operating for more than 20 years; 3) system size is generally maintained below 2 persons unit^{-1} except around major urban centers; and 4) 12% of census block groups have an estimated OWTS density higher than the EPA regulated threshold for negative environmental impact (15 unit km^{-2}) and are generally found clustered around city fringes.

3.4.3 PRIORITIZING ENVIRONMENTAL AND HEALTH RISK AREAS

The spatial distribution of potential environmental and health risk using the new soil suitability ratings, OWTS age, size, and density in each 141 m×141 m raster cell of the study area was mapped and is illustrated in Figure 3.5. Lowndes and Dallas counties were excluded due to unavailability of SSURGO data. Relatively higher potential environmental and health risks are observed around city fringe areas such as Montgomery and around smaller towns such as Selma and Uniontown where OWTS densities are relatively high (>100 units km^{-2}). This finding is not unexpected since city fringe areas with high OWTS densities have in fact been the focus of public health concern for decades as public sewer systems have continuously expanded to keep up with urban and suburban sprawl (Boyle and Otis, 1981; DeWalle and Schaff, 1980). This study confirms through both PCA and spatial analysis what has been reported by previous researchers that OWTS density is the factor that most influences local environmental and sanitary conditions (Lipp et al., 2001; Carroll and Goonetilleke, 2005).

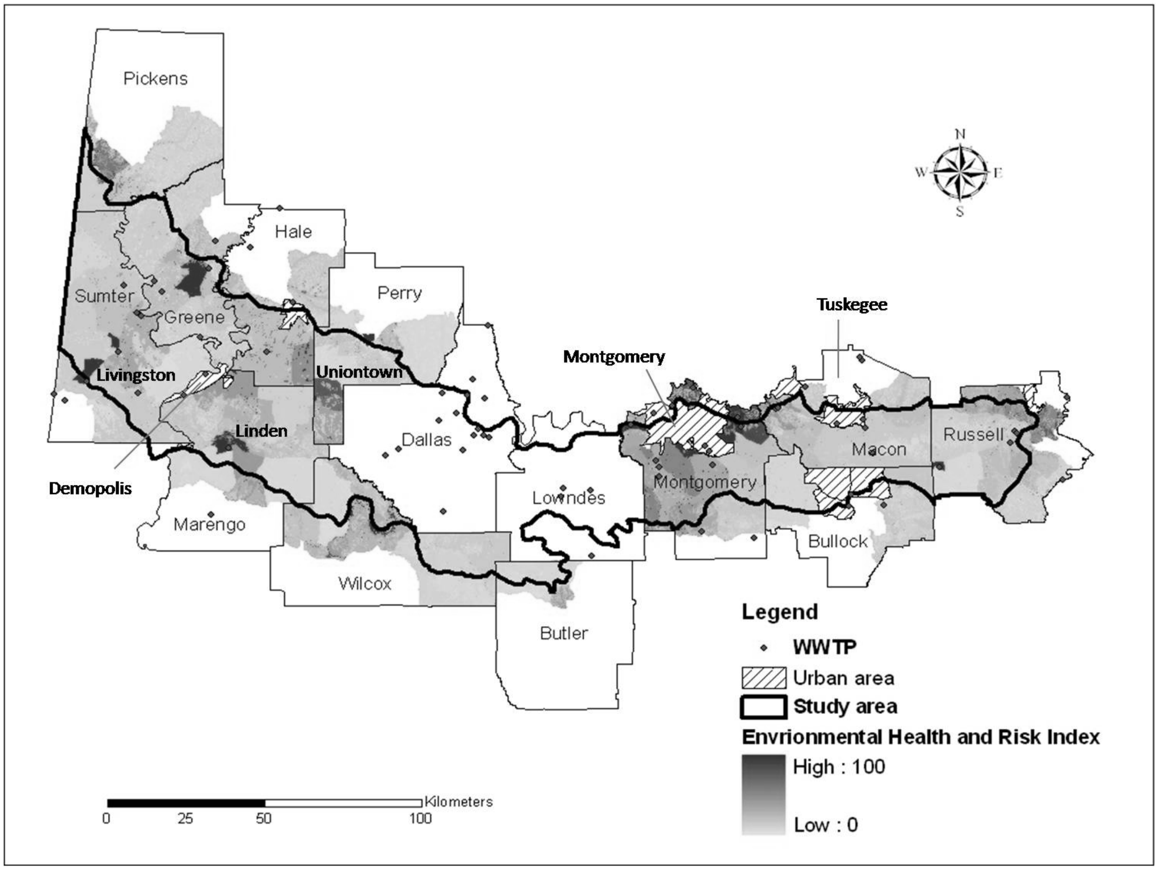

Figure 3.5 Environmental risk analysis results in the Black Belt study area. Dallas and Lowndes Counties excluded due to unavailability of SSURGO data. Urban areas excluded due to general availability of public sewer service.

3.4.4 Application of Risk Assessment Mapping

To illustrate the application of the risk analysis map, rural environmental and health risk ratings were overlaid with a 30 m false color Orthoimage (USGS, 2003) of Montgomery, Alabama. On the enlarged environmental and health risk map (Figure 3.6), the city boundary (thin black line) is shown along with current public WWTP service extents (green with black centerline).

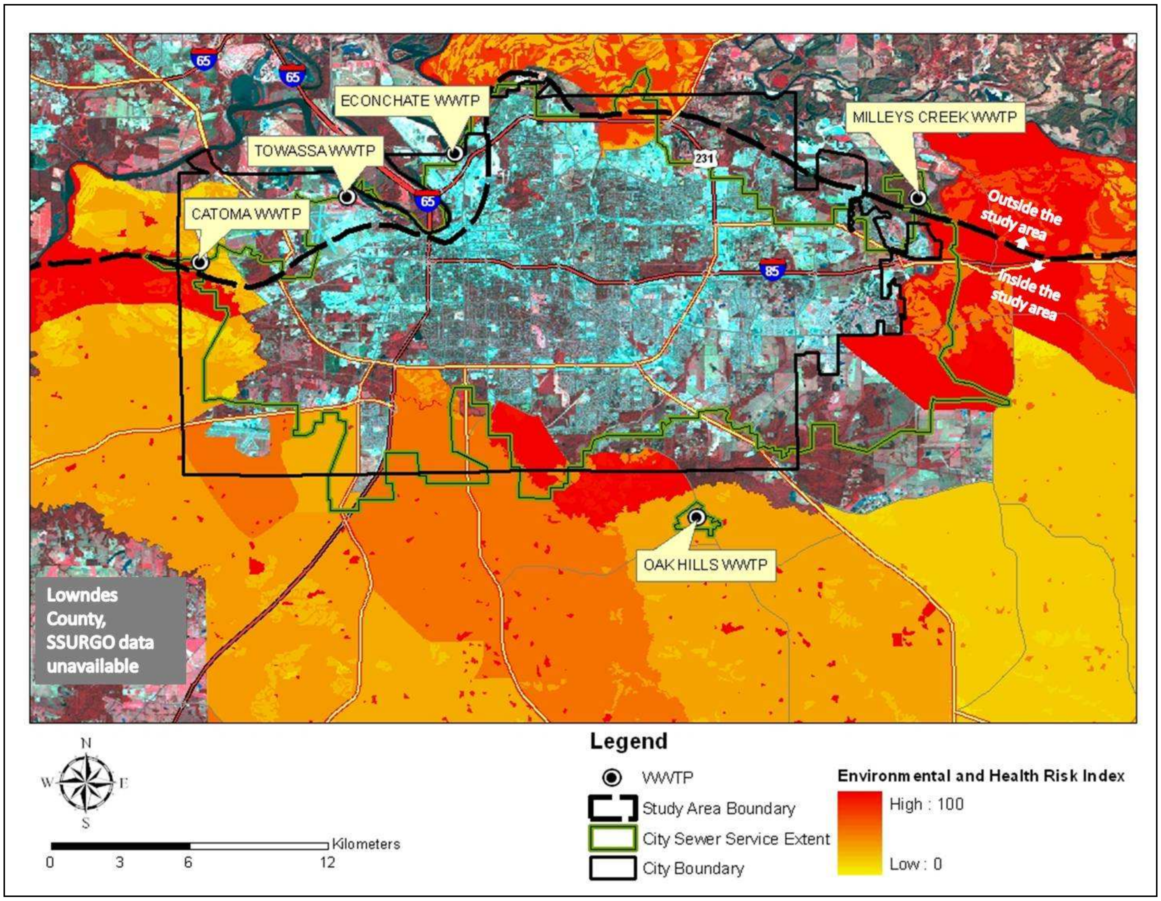

Figure 3.6 Environmental risk analysis of the greater Montgomery area, Alabama. Areas not rated are either classed as urban areas, or census block group extents are completely outside the study area.

Figure 3.6 indicates that public WWTPs are currently serving the general Montgomery metropolitan area, including numerous areas indicated as high environmental and health risk around the city fringe. However, several potential high risk areas within the city boundary to the west and south remain unsewered. The awareness that clayey soils and high OWTS densities are prevalent within the greater Montgomery municipal area has prompted the Montgomery Water Works & Sanitary Sewer Board to extend service to certain clayey soil areas in advance of OWTS malfunction (Montgomery Water Works & Sanitary Sewer Board, personal communication, February

12, 2009). For example, the Oak Hills WWTP located south of the city boundary currently serves an area indicated as having relative low risk, but provides potential future sewer service to surrounding high risk, unsewered areas to the north and west. Likewise, the Milleys Creek and Catoma WWTPs on the east and west sides of the city may be able to expand their respective service areas to adjacent high risk areas as necessary. In general, in the case of Montgomery, OWTS related environmental and health risks have been recognized and government efforts have been made to limit that risk. It is hoped that city fringe and other areas within Alabama can benefit from spatial assessment of environment and health risk from OWTS and respond with similar proactive planning strategies.

Although city fringes are found to have generally higher environmental and health risk from OWTS, especially in clayey soils, the result of this study does not suggest that the risk in rural areas in the Alabama Black Belt are insignificant. Prevailing system age greater than 20 years throughout this predominantly rural region strongly suggests the need to recondition aging OWTS. Since over half of rural sites in the Alabama Black Belt area are not suitable for conventional OWTS, alternative engineered systems such as mound systems or secondary treated drip irrigation systems approved by the Alabama Department of Public Health (ADPH, 2006) become the practical option for more than 62000 individual households in the region. Subsidized septic system retrofits may be recommended in certain rural communities facing higher public health threat. Assessment mapping of the type provided in this study can be used to target limited public resources.

Across the Alabama Black Belt region, continued targeted governmental efforts may be needed to successfully manage OWTS related environmental and health threats.

3.5 CONCLUSION

This study developed a new OWTS soil suitability rating system (OWTS-SSRS) based on current Alabama state OWTS regulations (ADPH, 2006) and on site conditions derived from SSURGO digital soil information (NRCS, 2007). The new OWTS-SSRS was compared to the existing NRCS soil limitation rating system (NRCS-SLRS) for siting of septic tank adsorption fields. The older NRCS rating system, a more conservative rating system, was also based on county soil survey data. Both assessment tools indicate that soil properties within the Black Belt study area of Alabama are generally unsuitable for conventional OWTS due to the prevalence of low permeability clayey soils, shallow ground water table, underground restrictive layers, steep slope, and/or flooding frequency. The new OWTS-SSRS rated 52% of the Alabama Black Belt study area as Unsuitable for OWTS, while the NRCS-SLRS rated 89% of the study area as Limiting for septic tank absorption fields. The difference between the two soil rating systems derives mainly from threshold values used by each to classify similar site conditions. The new OWTS-SSRS uses less restrictive values than the nationally distributed NRCS-SLRS but follows current Alabama Department of Public Health regulations. The new OWTS-SSRS is considered more practical for the state of Alabama because all potential OWTS sites in Alabama are verified by field reconnaissance before approval for installation. In those cases where soil site conditions prohibit the use of conventional OWTS, alternative engineered systems are mandated. With generally unfavorable soil conditions for conventional OWTS in the Alabama Black Belt, the expansion of the new OWTS-SSRS to include alternative engineered systems is recommended to further benefit the Alabama Black Belt in terms of wastewater dispersal.

Mapped results presented indicate that areas around city fringes have a higher environmental and health threat as a consequence of older OWTS of larger size and higher density. Consequently, city fringe areas associated with system densities greater than 100 unit km^{-2} should receive timely attention to mitigate risk from critical wastewater loading. However, because rural areas also need to assess and manage the potential risk of OWTS loading, two strategies to limit the potential environmental and health risk from OWTS malfunction in the Alabama Black Belt area are suggested. For city fringe communities, the proactive response is to extend municipal sewer service to high risk clay soil areas in advance of widespread OWTS malfunction. For isolated rural households outside the practical range of municipal sewer service or decentralized community systems, subsidized retrofitting, repair, or replacement of aged OWTS with alternative engineered systems is recommended.

Finally, this study demonstrates how spatial technologies and planning strategies can be used to target potentially serious regional non-point source pollution threats from aging and malfunctioning OWTS. OWTS risk assessment tools such as regional mapping products can be used to educate stakeholders about the direct link between soil and water stewardship, local environment, and regional public health. The GIS and demographic methods presented in this paper can be replicated to generate soil rating maps for the remaining counties of the Alabama Black Belt area or the entire state once digital

SSURGO soil data is made available. An expanded soil rating system that includes an evaluation of alternative engineered systems is recommended to facilitate individual, community, and government response to targeted critical risk areas.

CHAPTER FOUR

SOIL MOISTURE CONTROLLED WASTEWATER DISPERSAL – YEAR ONE: LABORATORY AND FIELD TESTING

4.1 ABSTRACT

Conventional onsite wastewater treatment systems (OWTS) in the Alabama Black Belt often rely on clayey, smectitic (shrink-swell) Vertisols for effluent dispersal. This study explores an alternative wastewater dispersal method for soils of this region. A field moisture controlled subsurface drip irrigation (SDI) system was designed, assembled, and tested under laboratory and field conditions. The objectives of this study were to: 1) design an automated control system that incorporates soil moisture monitoring into a conventional wastewater SDI system; 2) complete laboratory testing of the control system using soils of dissimilar permeability; 3) test automated system response to seasonal field conditions for a one year period using preprogrammed field moisture control set points; and 4) test system water balance with a seasonal cropping system in the drain field. A soil moisture monitor/control data logger (Delta-T, UK) was wired in series with a commercial wastewater SDI system (Geoflow, CA) to achieve soil moisture controlled SDI dosing. The experimental system was designed to disperse wastewater only when field moisture within the upper 45 cm was between 0.40 m^3 m^{-3} and 0.45 m^3

m^{-3}. Field testing took place from August 2006 to June 2007 at the Alabama Black Belt Research and Extension Center in Marion Junction, AL. A 500 m^2 experimental drain field was sized for a family of three producing approximately 1 m^3 of wastewater per day (ADPH, 2006). Soils are classified as very-fine, smectitic, thermic Oxyaquic Hapluderts. A warm season sorghum-sudangrass (*Sorghum bicolor*) and a cool season mix of winter wheat (*Triticum aestivum*) and rye (*Secale cereale*) was planted in rotation in the drain field. SDI wastewater tubing was installed at 20-25 cm depth, with two capacitance type soil moisture sensors buried at 20 cm and 45 cm depths near the center of the irrigated field. One year operation indicated that although the system was internally robust, unexpected power and water outages curtailed operation on occasion. The system effectively withheld water during wet soil conditions as designed and provided seasonally varying water dosing rates over periods of dryer soil conditions. Zero water dosing during wet soil conditions and during field crop harvesting and planting seasons indicates that at least two months of wastewater storage is required for this experimental system. Although percolation is a necessary component of effluent treatment in an OWTS, a monthly water balance of the experimental site estimates that over 30% of water applied during the drought period from March 2007 to June 2007 was lost to percolation below 45 cm, presumably as a result of dry weather clay soil cracking. Improved system control over percolation is recommended by placing multiple soil moisture sensors vertically and horizontally or by reducing the emitter and drip line spacing.

4.2 INTRODUCTION

Conventional onsite septic tank systems include an underground septic tank with a gravity drain field for the effluent. These systems, also called conventional onsite wastewater treatment systems (OWTS) (The Onsite Consortium, 2007), are the most common decentralized wastewater disposal method in the Alabama Black Belt area because of the relatively low cost of installation, operation, and maintenance (ADPH, 2006). Collected residential sewage goes through primary settling and biological reaction in the septic tank. Upon reaching a preset overflow, supernatant is dispersed (where effluent is getting treated) by gravity to a drain field where percolation through an unsaturated soil zone provides aerobic treatment of the effluent (ADPH, 2006). The environmental challenge for conventional OWTS comes from the almost complete reliance on soil properties for proper waste treatment (Oron, 1996). Soils having too high or too low a percolation rate are generally not suitable for conventional onsite septic systems (US EPA, 2002). In the shrink-swell clay soils that dominate the Black Belt region of Alabama, conventional OWTS can pose a genuine environmental and health threat if not designed and operated properly (McCoy et al., 2004).

Drain field failure and subsequent nutrient overload is a recognized risk from OWTS. Typical nitrogen concentrations in septic tank effluent range from 40-80 mg L^{-1} (Walker et al., 1973), of which approximately 75% is ammonium nitrogen and 25% is organic nitrogen (Otis et al., 1974). Reported total phosphorus concentration in septic tank effluent ranges from 3 to 20 mg L^{-1} (Robertson et al., 1998; Whelan, 1988), with about 85% as orthophosphate (Reneau and Pettry, 1976). Charles et al. (2005) analyzed several intensive septic tank effluent field surveys from 1976 to 1999 in Australia and the US and concluded that nutrient overloading may be occurring in a significant number of OWTS designed under current Australian and US regulations. They recommended that the 80th percentile of effluent survey values, 250 mg TN L^{-1} and 36 mg TP L^{-1}, should be used in OWTS design to minimize the nutrient overloading associated with onsite drain field failure.

Incidences of poor treatment performance from onsite treatment systems, particularly onsite septic systems, are common in the US and worldwide (US EPA, 2002; Carroll and Goonetilleke, 2005), and are a significant source of water pollution (Beggs et al., 2004). Lipp et al. (2001) demonstrated the adverse pathogen impact from onsite sewage systems to a coastal community in Sarasota Bay, FL. Carroll and Goonetilleke (2005) confirmed that a high system density (290 systems km^{-2}) significantly impacted shallow groundwater systems due to the cumulative nutrient and pathogen load that exceeds the capacity of local soils to assimilate (US EPA, 1977; US EPA, 2002).

A series of GIS analyses conducted by He et al. (2007) evaluated environmental and health risk to ground and surface water from conventional onsite septic systems in the Alabama Black Belt soil area. In 2000, more than 97% of the rural census block groups in this region had onsite systems with an average age of over 20 years. This data confirms the widespread use and aging of conventional onsite septic systems in the area. Subsequent risk analysis and ranking revealed that in absence of centralized municipal wastewater collection, ground and surface water resources immediately surrounding city fringes are at higher risk of being impaired by high OWTS densities over 15 units km^{-2}.

Subsurface drip irrigation (SDI) has the potential to address some of the issues with onsite septic systems because the application of water is below the soil surface through emitters. Additionally, discharge can be rated by standard methods of uniformity and efficiency (Camp, 1998). Wastewater disposal through SDI can provide improved application efficiency and more uniform distribution of effluent throughout the reuse area, reducing the risk of ground and surface water contamination (Jnad et al., 2001).

Commercially available wastewater SDI systems designed for onsite septic tank effluent are usually coupled with a small, advanced or secondary treatment system with sufficient treatment and filtering capacity to prevent clogging of SDI emitters. Most SDI control panels utilize a preset time interval for hydraulic dosing of drain fields but do not incorporate any automated control besides high and low water cut-off and alarm (Jnad et al., 2001). Soil moisture controlled SDI systems have been shown to improve water use efficiency by reducing evapotranspiration (Dukes et al., 2005). For domestic wastewater dosing in the Alabama Black Belt region, automated dispersal systems have potential to reduce the risk of drain field hydraulic overloading by not exceeding the field capacity in native clay and other heavy soils.

In this study, a pilot scale SDI wastewater dispersal system controlled by volumetric soil moisture content was built and evaluated in the laboratory for subsequent field installation and testing. This study completes the retrofit of a commercially available SDI wastewater dispersal system with independent soil moisture feedback to prevent drain field hydraulic overload. The concept is to allow wastewater dispersal only when field moisture below field capacity so as to limit ponding and deep percolation while enhancing crop water uptake and aerobic soil treatment. Although this experimental wastewater SDI dispersal system may not be cost effective for the majority of rural home owners in the Black Belt, water balance observations in this study provide information regarding system feasibility.

The goal of the study was to evaluate the hydraulic management of an experimental wastewater SDI system in a clay soil site. The objectives were to: 1) design an automated control system that incorporates soil moisture monitoring into a conventional wastewater SDI system; 2) complete laboratory testing of the control system using soils of dissimilar permeability; 3) test the automated system response to seasonal field conditions for a one year period using preprogrammed field moisture control set points; and 4) evaluate system water balance with a seasonal cropping system in the drain field.

4.3 Materials and Methods

The experimental system was assembled in the laboratory to test operation of electrical and hydraulic components. The system was then installed and evaluated with clean well water on a 500 m^2 natural Houston clay site from September 06, 2006 to June 14, 2007.

4.3.1 Laboratory Assembly and Testing Methods

Controller Interface

A wastewater SDI controller (GEO1, Geoflow, CA) was wired to a data logger/controller (GP1, Delta-T, UK) to provide both real-time soil moisture control and data logging capabilities for the experimental system. The GEO1 is an elapsed time meter (ETM) controller that uses only low reservoir water level to interrupt a programmable on/off wastewater dosing sequence. The GP1 data logger/controller is manufactured for research with the capability to collect and archive data from the following test devices; 2 capacitance type soil moisture sensors (ML2 ThetaProbe, Delta-T, UK) with typical errors of ± 0.01 m^3 m^{-3} after being validated with intact soil cores, 1 soil temperature sensor, 1 flow sensor, and 1 tipping bucket rain gauge. Logged data from the GP1 can be programmed, viewed, retrieved, and archived for selected time intervals using accompanying software (DeltaLINK-PC, Delta-T, UK). The GP1 data logger/controller was connected to the two soil moisture sensors. The programmable GP1 controlled the opening and closing of an external 12V circuit based on preset thresholds.

The electrical schematic in Figure 4.1 depicts the interface of the GEO1 control panel with the GP1 data logger/controller. GP1 12V output controls a 12V/115V intermediate relay. The 115V side of the intermediate relay was installed in series with the existing low water float switch (#2) circuit. A 12V light-emitting diode (LED) was installed between terminals 14-13 of the intermediate relay as a visual indicator of the status of relay A.

Two water float switches (#2 and #4) out of four typically provided with the GEO1 controller (10 amps, 120/230V AC) were used in this study. Float 4 is a high water level alarm, used to indicate excessive effluent in the reservoir. Float 2 signals low water level, in which case the ETM is interrupted until a safe liquid pumping level is restored. Terminals 2 and 4 in the GEO1 control panel correspond to the positive side of the two float switches, with two #5 terminals provided as ground terminals. Terminal 12 in the intermediate relay was wired into terminal 2 of the GEO1 control panel while terminal 8 was piggybacked onto the positive side of float 2. The negative side of float #2 was grounded to terminal 5. With this electrical design, the dosing sequence is activated only when both: 1) the intermediate relay is closed, indicating that soil moisture readings are within the designated range, and 2) the circuit for float #2 is closed, indicating an adequate water level for pumping.

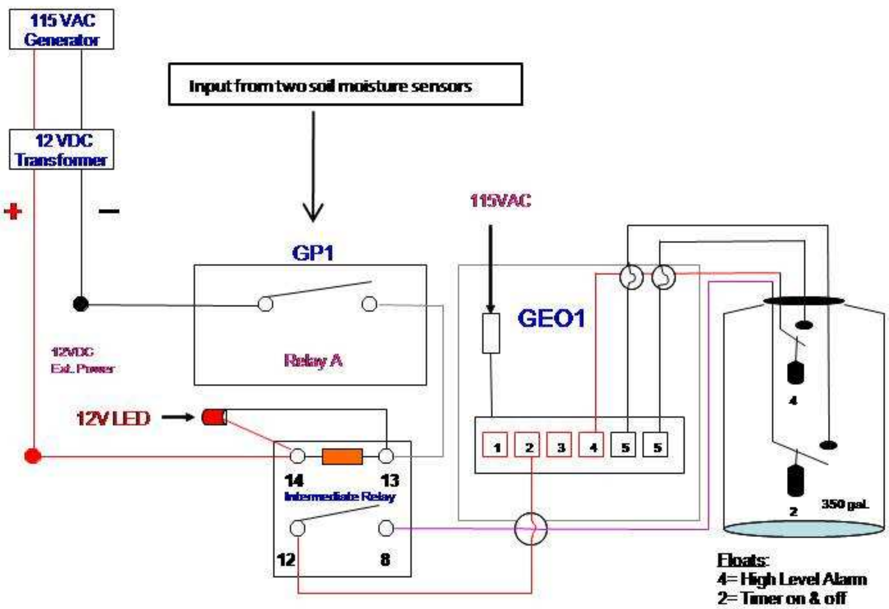

Figure 4.1 Connection diagram illustrating interface of soil moisture sensors and control to existing float-based irrigation dosing system. (Soil moisture sensors not shown)

LABORATORY DESIGN AND TESTING

The integrated control and data logging system provided by the GP1 and GEO1 was laboratory tested using an SDI system consisting of two lengths of wastewater drip tubing (Figure 4.2). A 0.37 kW submersible effluent pump (Myers, OH) was wired into the GEO1 control panel and placed into a plastic 1250 liter water tank. Pumped water flowed through a headworks box equipped with a vortex filter and a 240 kPa (35 psi) pressure regulator before flowing into two parallel, 6.1 m long SDI drip tubes. The drip tube was WFPC16–2–24, 16 mm diameter (Geoflow, CA) with 0.61 m emitters spacing. The maximum allowable particle size that can pass through emitters is 100 µm. The downstream end of each drip tube was attached to a 2.54 cm return manifold, equipped with an air vacuum breaker. The 7.62 m return line returned water through the headworks box and into the water tank.

The GEO1 operating sequence, based on the SDI manufacturer's recommendation (Geoflow, 2003), was set to a 5-minute dosing period followed by a 25-minute resting period, providing for approximately two dosing cycles each hour. A forward field flush operation that flushes drip tubes to clean potentially clogged emitters was programmed to occur every 10th dosing cycle. The 5-minute field flush valve was manually set to provide a system pressure (241 kPa) sufficient to maintain a flush velocity of 0.61 m s^{-1}. Each dosing and field flush was followed by a manufacturer programmed 15-second filter flush, 5-second pump delay, and 1-minute drip drain period.

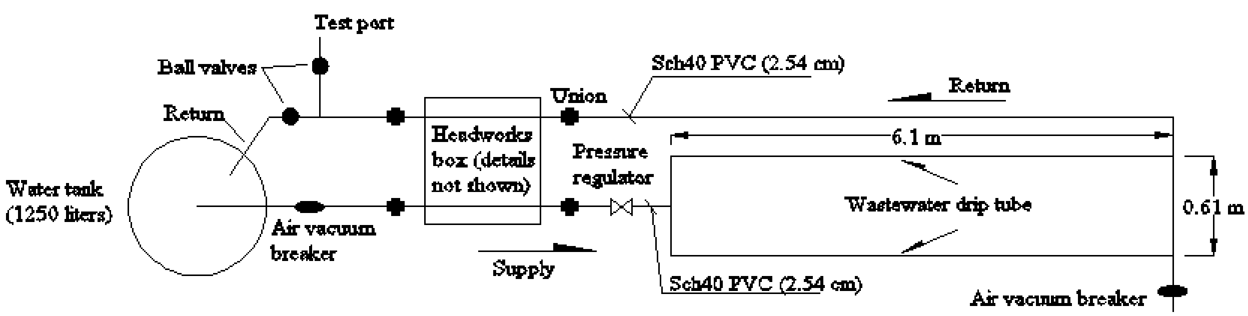

Figure 4.2 Schematic of the laboratory SDI system layout.

Measured drip tube emitter flows in the laboratory layout (n = 60) ranged from 1.80 to 2.30 lph, with an average of 2.02 lph. Observed coefficient of variance (C_v) was 0.067, indicating average emitter performance. (ASAE EP 405.1, 2003)

Real time soil moisture feedback-control was tested using two different soil textures. Bagged playground sand (American Countryside, AL) and surface horizon Houston clay at the field site were used to evaluate system response to very low and very high percolation soils. Particle size distribution, porosity, saturated hydraulic conductivity (K_{sat}), and field capacity of the two media were measured using standard methods (Table 4.1).

Table 4.1 Properties of Houston clay and sand media used in laboratory testing.

	Particle Size Distribution[1]			Porosity[2]	K_{sat}[3]	Field Capacity[4]
	Sand	Silt	Clay			
	%			%	$\mu m\ s^{-1}$	%, vol
Houston clay	7	40	53	63	4.2	41
Fine sand	100	0	0	37	165	10

1. Using the pipet analysis method (Soil Survey Investigation Staff, 2004)
2. Calculated from measured bulk density (Gravimetric method (SSSA, 2002b) using intact soil cores) and particle density (Liquid displacement method (SSSA, 2002b)).
3. Houston clay was measured with permeameter (Ksat Inc., CA) at 45 cm depth, playground sand was laboratory measured with constant head method (SSSA, 2002b).
4. Measured with laboratory pressure plate method (SSSA, 2002b).

Based on NRCS Web Soil Survey information (NRCS, 2008), the field capacity (1/3 bar) of the Houston clay soil site (site information will be provided later) is approximately 0.42 $m^3\ m^{-3}$. Consequently, the soil moisture thresholds used for control testing were set at 0.40 $m^3\ m^{-3}$ (on) and 0.45 $m^3\ m^{-3}$ (off). The intent with these thresholds is to avoid hydraulic overloading at the experimental site, while providing sufficient moisture for plant uptake and aerobic soil treatment of effluent. The sand media was tested using the same thresholds as the clay in order to: 1) observe the more frequent response of the GP1 controller/data logger over changing soil moisture conditions within a limited time frame; and 2) evaluate the corresponding effectiveness of the control system in a highly permeable medium.

The GP1 logical criteria used in the laboratory study are illustrated in Figure 4.3. Sensors #1 and #2 represent hypothetical soil moisture readings at 20 cm and 45 cm depth, respectively. If there is sufficient water above the pump intake, initial soil moisture readings (Condition A) allow normal dosing pump operation. Dosing increases soil moisture readings until either one of the two soil moisture sensors reads above 0.45 $m^3 m^{-3}$ (Condition B). The dosing system will remain idle until free drainage or evapotranspiration lowers the volumetric moisture reading of one of the two soil moisture sensors below 0.40 $m^3 m^{-3}$ and the other one to below 0.45 $m^3 m^{-3}$ (Condition D). The GEO1 dosing sequence will initiate, increasing soil moisture level back to a system cut off level (Condition F). System operation subsequently cycles between conditions B and F. The dosing system remains idle at any time there is insufficient water in the reservoir (#2 float circuit is opened) or if the electric supply is interrupted.

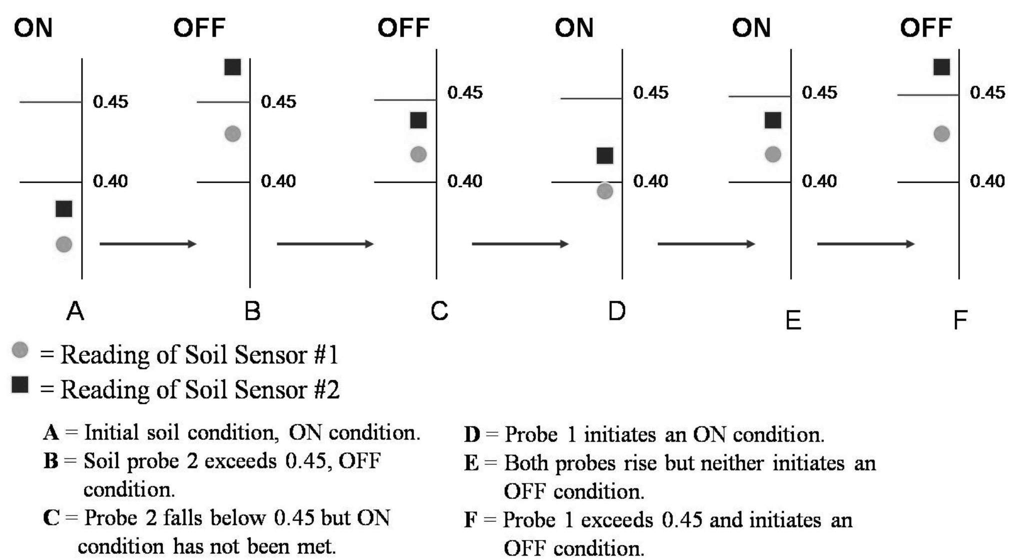

Figure 4.3 GP1 logical dosing sequence based on volumetric soil moisture, $m^3 m^{-3}$.

For each test, selected media was placed in two 2-liter free-draining containers. Each container had one buried moisture sensor and was placed under one operating drip tube emitter along each drip tube. The tipping bucket rain gauge was positioned under a third emitter to simulate rain gauge data logging. Laboratory evaluation included readings for the two soil moisture sensors, one soil temperature sensor, a flow meter, rain gage, as well as system operating voltage.

4.3.2 LABORATORY TESTING RESULTS

SAND MEDIA

In the container filled with sand water from the emitter was drained so quickly that the moisture content of the media never rose above 0.45 $m^3\ m^{-3}$ (Figure 4.4). Consequently, dosing was never interrupted, providing continuous dosing as expected.

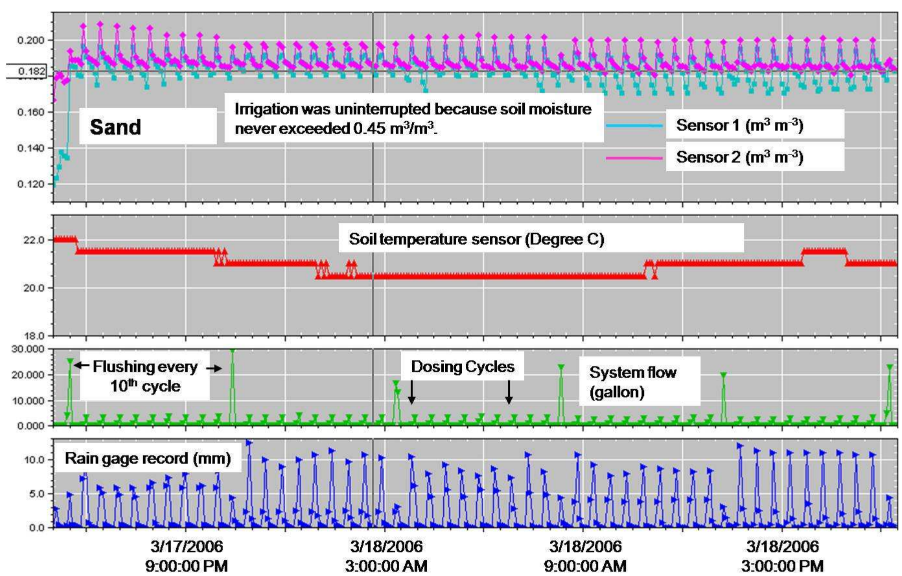

Figure 4.4 Sand media with float #2 in a closed (ON) position and suffcient water in reservoir.

To verify that GEO1 float level interruption would curtail dosing even when moisture content allowed, the low water float switch (#2) was lifted out of the water tank to open the float circuit interrupting the GEO1 operating sequence (Figure 4.5). As designed, the simulated low water level stopped pump operation (no flow recorded), even though moisture sensors control continued to call for irrigation. As a result, no water was emitted by the drip tube (Figure 4.5). The readings of the two soil moisture sensors thereafter decreased as the water drained from the containers.

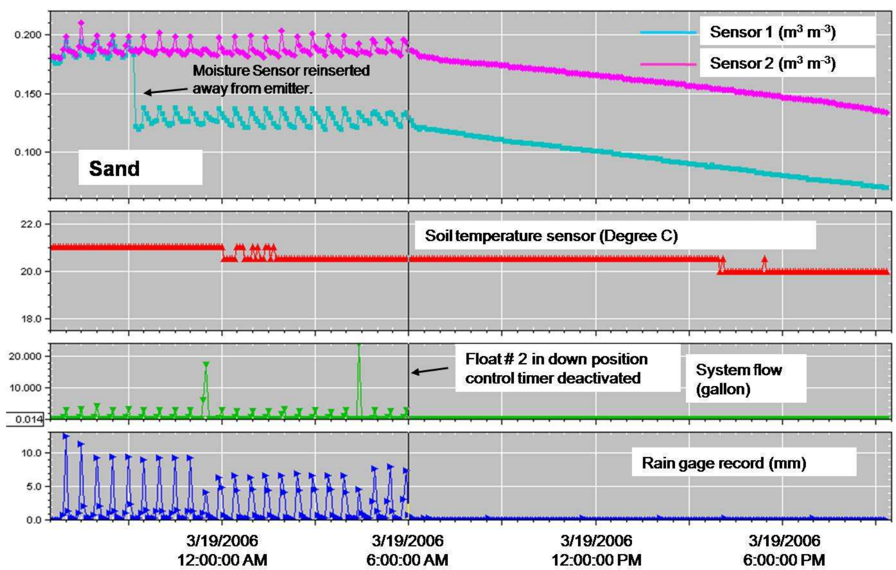

Figure 4.5 Sand media with float #2 in open (OFF) position to test low water level response.

CLAY SOIL

A comparable test of the Houston clay soil indicated that the system responded as designed with a significantly lower dosing frequency (no dosing for two days after initial dosing) compared to the sand media (average 2 dosings per hour). As indicated in Figure 4.6, when there was sufficient water in the reservoir the soil moisture level rose to above 0.45 $m^3\ m^{-3}$ after only two dosing cycles, effectively shutting down the pump. After approximately 14 hours, the pump was activated when 0.40 m^3 m^{-3} was measured by soil moisture sensor #2. Since the dosing cycle occurred at the same time that a forward field flush cycle was preprogrammed, a higher system flow was recorded as noted in Figure 4.6. The soil moisture level was brought to and maintained above 0.40 m^3 m^{-3} by this dosing cycle. This observation was expected since the water holding capacity (as porosity in Table 4.1) for the Houston clay soil is almost twice as much as the sand media, while its saturated hydraulic conductivity is about 40 times lower.

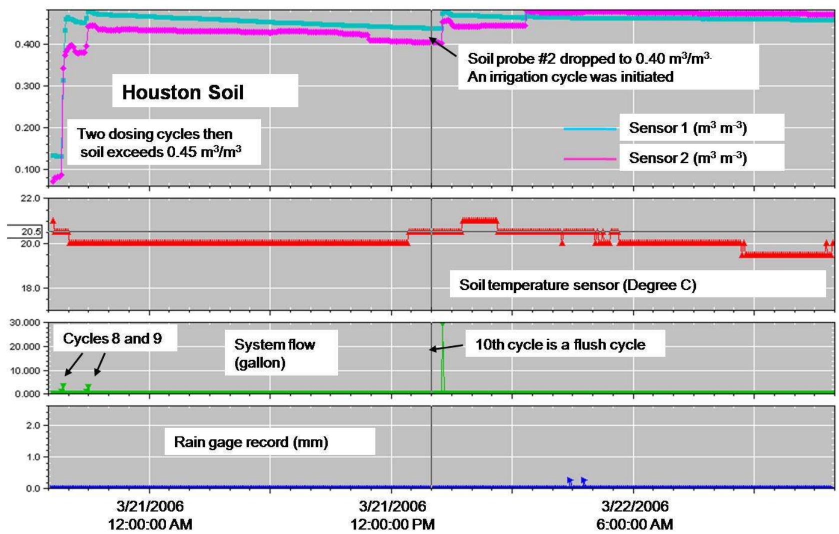

Figure 4.6 Houston clay soil test with float #2 in a closed position and enough water in reservoir.

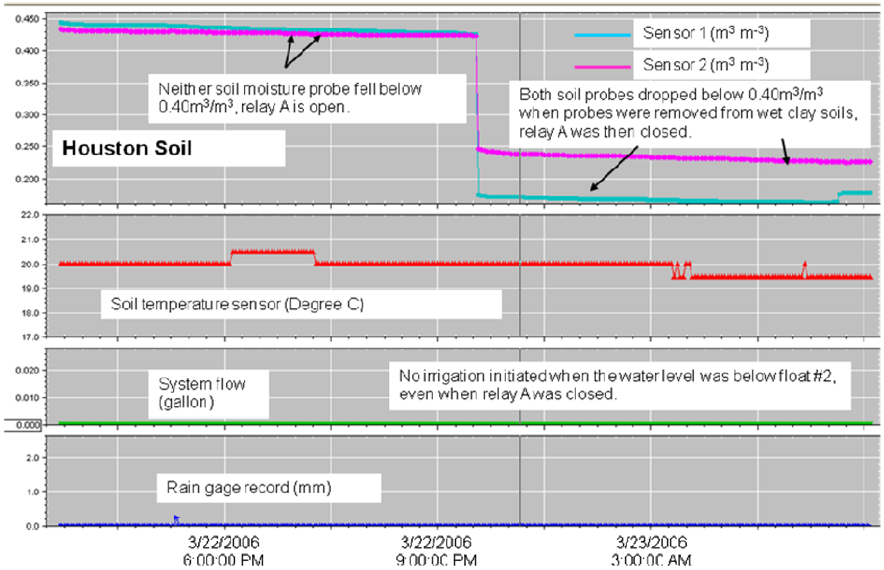

Figure 4.7 Houston clay soil test with float #2 in an open position to simulate low water level in reservoir.

The two soil moisture sensors were taken out of the two containers filled with clay soil to intentionally lower moisture readings below 0.40 $m^3 m^{-3}$ and allow activation of relay A (Figure 4.7). At the same time, to verify if GEO1 interruption would stop dosing even when soil moisture allowed, the #2 float (low water switch) in the water tank was manually opened. As designed, the open #2 float interrupted water pumping, and no dosing occurred in the SDI wastewater system, even when soil moisture dropped below 0.40 $m^3 m^{-3}$ (Figure 4.7).

Consequently, laboratory results indicated that the soil moisture control system operated as designed under the soil moisture thresholds evaluated. In order to prevent hydraulic overloading in sandy fields, lower thresholds would need to be applied to account for much lower field capacity and water holding capacity. The experimental system was subsequently upscaled to a Houston soil field site to evaluate hydraulic performance in a clayey soil.

4.3.3 FIELD INSTALLATION AND TESTING

SITE SELECTION AND CHARACTERIZATION

The site selected for the field study is in Marion Junction, Alabama, at the Alabama Black Belt Research and Extension Center (ABBREC), approximately 10 miles west of Selma, Alabama. A Houston clay soil site with 1% slope was selected because it provided the fewest impediments to year-round SDI wastewater dosing, while providing low permeability and high shrink-swell features common to the region. An electric service (220v/110v, max. 60A) and a water supply well (max. 9000 lph) provided necessary utility connections. A truck-mounted Giddings® probe and sleeve was used to retrieve core samples from the site for soil profile characterization for the Houston soil (Table 4.2). Five soil horizons (Ap1, Ap2, BA, Bkss1, Bkss2) were identified to 1.52 m. Dark clay was prominent at the surface to approximately 42 cm depth, with redoximorphic features at 88 cm indicating significant periods of saturated or anaerobic conditions during most years. Particle size distribution indicates increasing clay content with depth, up to 71% at 152 cm. According to the Geological Survey of Alabama (1993), the experimental site is underlain by a relatively impermeable layer of fossiliferous clayey chalk of greater than 100 m thick at a general depth of 6 m, with shallower formations found at 12 cm to 2 m as well.

Table 4.2 Soil description and texture data to 1.52 m depth at the experimental site, Black Belt Research and Extension Center, Marion Junction, Alabama.

Horizon Depth (cm)	Color	Description*	Particle Size Distribution (%)		
			Sand	Silt	Clay
Ap1 (0-23)	very dark gray (2.5Y 3/1)	clay	7.09	39.63	53.28
Ap2 (23-42)	dark grayish brown (2.5Y 4/2)	clay	8.27	38.04	53.7
BA (42-63)	olive brown (2.5Y 4/3)	clay; few fine black manganese oxide concretions; common medium and coarse calcium carbonate concretions and soft accumulations	10.17	33.38	56.45
Bkss1 (63-88)	olive brown (2.5Y 4/3)	clay; large wedge-shaped aggregates that are bordered by intersecting slickensides; very plastic, sticky; few fine black manganese oxide concretions; common medium and coarse calcium carbonate concretions and soft accumulations; calcareous	3.50	35.93	60.57
Bkss2 (88-152+)	light olive brown (2.5Y 5/6); common light brownish gray (10YR 6/2) redox depletions; common dark yellowish brown (10YR 4/6) redox concentrations	clay; large wedge-shaped aggregates that are bordered by intersecting slickensides; very plastic, sticky; few fine black manganese oxide concretions; common medium and coarse calcium carbonate concretions and soft accumulations; calcareous	3.10	25.80	71.10

*Soil described as per National Cooperative Soil Survey Standard.

FIELD EXPERIMENT DESIGN

Based on Alabama Department of Health (2006) regulations for onsite sewage disposal, the allowable hydraulic loading (dosing) rates for a Houston clay soil is 2.04 lpd m^{-2}. The design flow rate of the experimental system was set at 1022 liters per day, equivalent to the daily wastewater flow of a 3-person home in a decentralized subdivision system (Alabama Department of Public Health, June 14, 2005, personal communication). Consequently, a total drainage or disposal soil area of 500 m^2 was required for the design. Based on standard practice of 0.61 m spacing between drip lines and 0.61 m spacing between emitters (Geoflow, 2003), a total of 823 m of drip line was required.

Figure 4.8 presents the field layout consisting of 30 drip laterals 27.43 m in length. The SDI system was hydraulically divided into two subplots of 15 drip tubes each, designed to accommodate two matching irrigation treatments. In this current study, both treatment sub plots were supplied with clean well water and identical cultural practices, and were subsequently were subsequently analyzed as one field.

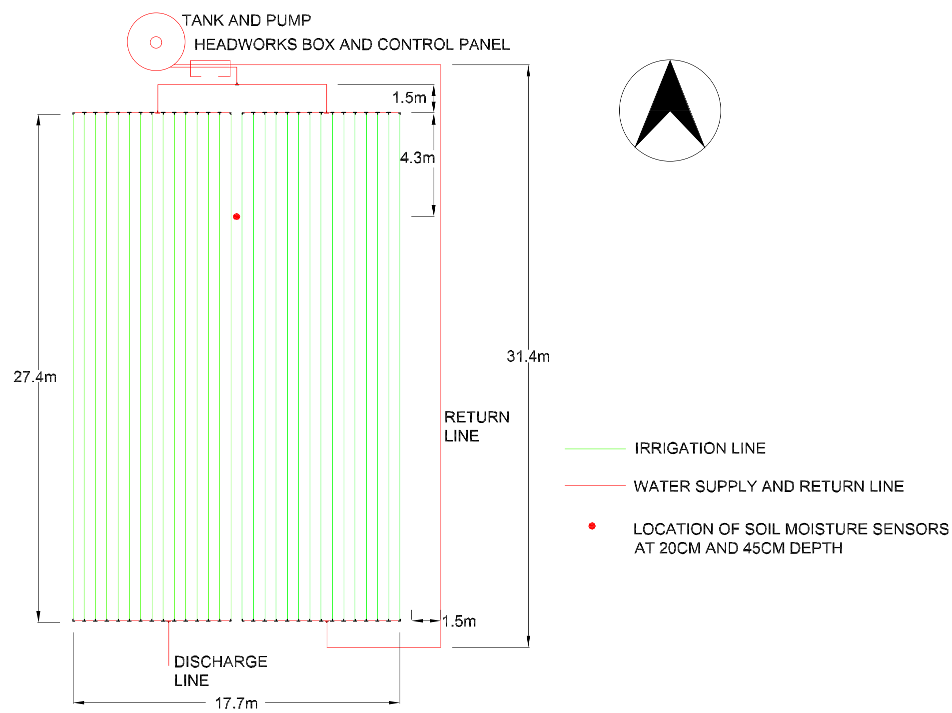

Figure 4.8 Design layout of the experimental wastewater SDI system, including clean water reservoir tank and pump, headwords box, control panel, supply and return manifolds, and 30 drip laterals at 0.61m spacing. The SDI system was divided into two equal subplots.

FIELD INSTALLATION AND SETUP

On June 26, 2006, three weeks prior to SDI system installation, field tillage was conducted to loosen extremely dry soil at the site. On July 19, 2006, installation began and was completed in two days. Headworks box, SDI control panel, soil moisture data logger/controller, and SDI tubing were the same as in the laboratory test. A 7571 L (2000 gallon) plastic septic tank (FRALO, NY) was used as the water reservoir. A 0.37 kW submersible pump was installed inside the plastic septic tank and wired to the GEO1, with power supplied from an existing meter near a water supply well 60 m away from the tank. The GP1 and GEO1 were mounted on a single panel next to the pump and headworks box (Figure 4.8). The tank was automatically filled by the well and a pressure tank activated by a mechanical float valve in the tank. All SDI main supply and return lines were Sch 40 PVC. Drip tubes were installed at a depth of 20-25 cm. Necessary electric components were field grounded, exposed wires were waterproofed, and control panel openings were sealed to prevent damage from animals and insects. The two capacitance soil moisture sensors were buried in one location at 20 and 45-cm depth to provide soil moisture monitoring during system operation (Figure 4.8).

Field crops were planted two weeks after SDI installation was completed. Crops grown during the one year study included sorghum-sudangrass (*Sorghum bicolor* (L.) Moench) from August 3, 2006 to November 1, 2006 and a mixture of winter wheat (*Triticum aestivum)* and rye (*Secale cereale)* from November 1, 2006 to June 08, 2007. Sorghum-sudangrass was planted with a grain drill at 33.6 kg seed per hectare on 18 cm row spacing. Winter wheat was planted with a grain drill on 18 cm row spacing at 67.2 kg per hectare; and ryegrass was broadcast at 22.4 kg per hectare.

The soil moisture operating thresholds were 0.40 to 0.45 $m^3\ m^{-3}$, identical to the laboratory test. The SDI wastewater controller was set to a 5-minute dosing period followed by a 55-minute resting period, providing approximately one dosing cycle per hour. Since only clean well water was used in this study, a forward field flush was programmed to occur only every 1000th cycle. Dosing and field flushing were followed by a 15-second filter flush and a 5-second pump delay, per manufacturer's recommendations. The GP1 data logger/controller was set to record rainfall (mm), soil moisture (v/v), flow volume (gallon), voltage (v), and temperature (degree C) every 15 minutes.

SYSTEM HYDRAULIC PERFORMANCE EVALUATION

A monthly water balance was developed for the drain field from September 2006 to June 2007 to evaluate the impact of automatic system control on soil water. Components of the water balance included depth of disposed water (D), precipitation (P), evapotranspiration (ET), percolation below 45 cm depth (θ), and water content change in the upper 45 cm ($\Delta\theta$).

Drain field surface runoff and soil lateral flow was neglected because the experimental site was relatively level. Percolation below 45 cm depth was estimated by mass balance difference between water balance components, including water inputs, estimated monthly drain field soil moisture change, and calculated field ET (Eq. 4.1). If calculated net monthly percolation indicated a positive moisture gain to soil above 45 cm depth (ie. From surrounding soil), then this value was identified as an error term to properly balance water input and outputs.

$$\theta = D + P - ET - \Delta\theta \quad \text{Eq. 4.1}$$

The change in monthly soil moisture content within the upper 45 cm of the soil ($\Delta\theta$) was estimated as the difference between weighted field water content (Eq. 4.2) at the beginning and ending dates of each month based on in situ soil moisture readings. Since soil moisture content was measured at 20 and 45 cm depth, the assumption was made that 1) soil moisture content varied linearly between 20 and 45 cm depth, and 2) soil moisture content in the upper 20 cm depth was represented by the reading at 20 cm.

$$\theta_{upper\ 45\ cm} = (\theta_{20cm} \times 20\ cm) + [\ (\theta_{20cm} + \theta_{45cm}) \times 25\ cm/2\ ] \quad \text{Eq. 4.2}$$

Field evapotranspiration (ET) was estimated using the Penman-Monteith method (FAO, 2006). Data for ET calculation was obtained from the Alabama Agricultural

Weather Information Service (AWIS). Since all necessary weather data for the Penman-Monteith method was not available at the Black Belt station, selected weather data from Thorsby weather station approximately 77 km from the experimental site was used. Weather data at 1.52 m height above the surface included daily maximum, minimum, and average air temperature, daily maximum and minimum relative humidity, daily solar radiation, and daily maximum wind speed.

4.4 FIELD TESTING RESULTS

4.4.1 SYSTEM OPERATION RESULTS

Daily rainfall, soil moisture, daily system hydraulic disposal rate, soil temperature, and calculated daily ET are illustrated in Figure 4.9 from September 6, 2006 to June 14, 2007. After the system was placed into operation, an automatically controlled hydraulic dosing rate of approximately 1.4 cm d^{-1} was maintained until soil moisture stabilized at higher levels around September 17, 2006 when the hydraulic dosing rate dropped to below 1.0 cm d^{-1}. On September 12, a 12 mm precipitation event increased field moisture, and soil moisture readings thereafter stabilized at approximately 0.30 m^3 m^{-3} at 20 cm depth and 0.45 m^3 m^{-3} at 45 cm depth. The hydraulic dosing rate was stabilized at 0.7-1.0 cm d^{-1}. The water supply to the tank was abruptly cut off on the last day of September due to a reservoir float malfunction. As designed, the water dosing pump automatically stopped although soil moisture levels called for irrigation. After the water supply problem was corrected the next day, the system resumed soil moisture

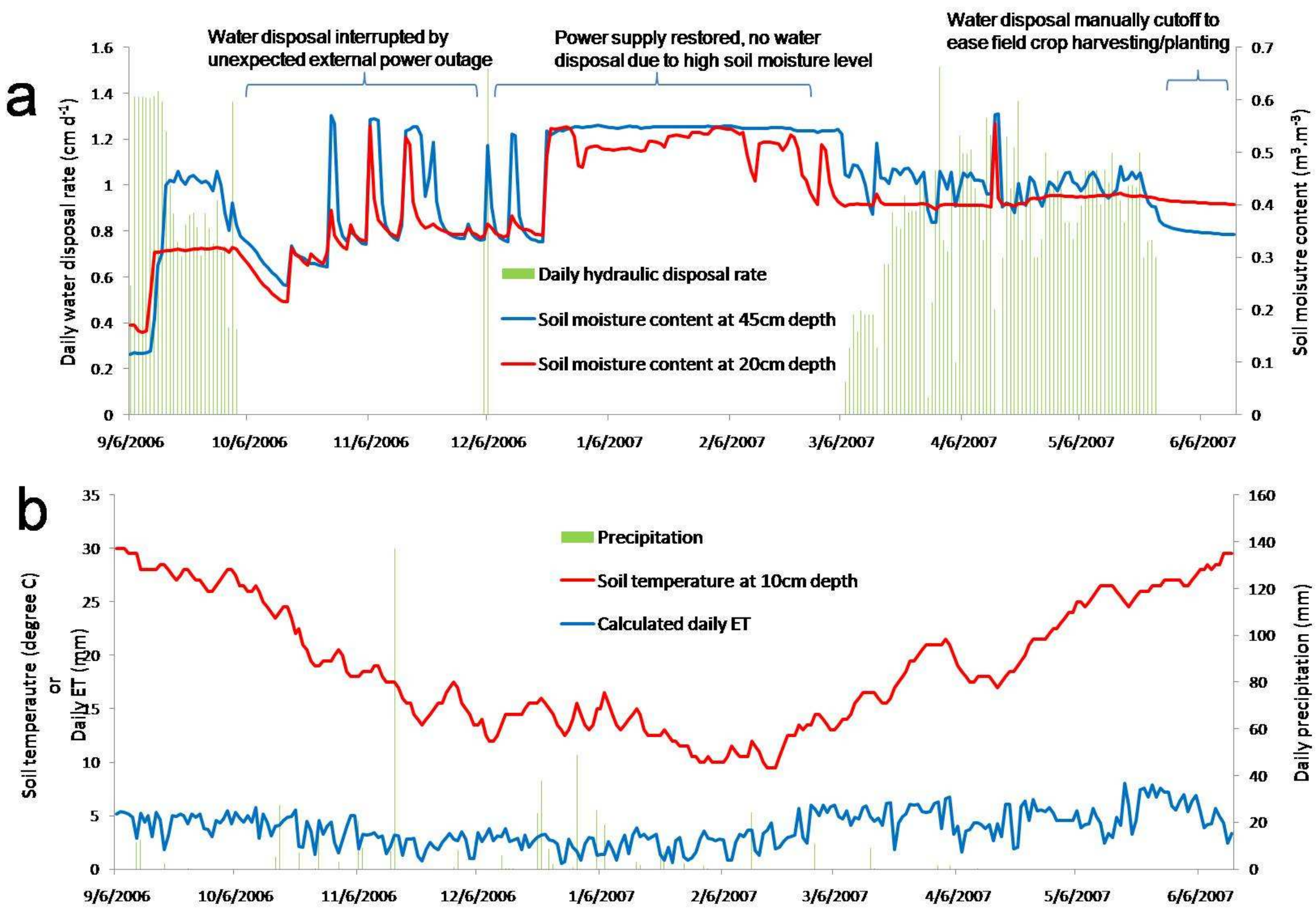

Figure 4.9 Field data recorded from September 2006 to June 2007. (a. field monitored soil moisture contents and system daily hydraulic disposal rates; b. field monitored daily precipitation, soil temperature at 10 cm depth. (Calculated daily ET was added to graph for reference)

controlled water dosing. This water supply interruption afforded the opportunity to verify a successful low water level control in tandem with soil moisture control in the system.

The experimental system was left unattended for two months during the winter of 2006/07. The data logger indicated that an external power supply interruption occurred on October 3, 2006 until December 5, 2006 (Figure 4.9), curtailing pump operation for more than two months. After the power outage was corrected on December 5, 2006, the system resumed normal operation (Figure 4.9). During the power outage, data logging backed up by an on-board battery provided a time series of soil moisture under natural rainfall at the site.

As indicated from Figure 4.9, there were likely several opportunities for water dosing during November and December 2007 if the external power supply had not been interrupted. Nevertheless, soil moisture content increased rapidly and maintained conditions >0.50 m^3 m^{-3} after a 61 mm precipitation event on December 20-21, 2006, followed by several other large events and additional precipitation until the end of February 2007. Field records indicate that the system cutoff threshold of 0.45 m^3 m^{-3} was exceeded until mid-February 2007 due to low temperatures (average 12 ^oC), low evapotranspiration, and continued rainfall that kept the site from draining. Rural households adopting this type of OWTS would require an alternative dispersal method or additional 2-month storage during periods of power interruption or lengthy wet field conditions.

It was also observed that the experimental system never initiated water dosing when either of the two soil moisture sensors gave a reading higher than 0.45 m^3 m^{-3}. The only surface ponding that was likely was during wet winter months when the soil moisture sensors indicated higher than 0.45 $m^3 m^{-3}$. Consequently, surface ponding is not considered a major concern of the experimental system.

During the first week of March 2007, recorded soil moisture levels at 20 cm depth began to drop, followed by a drop in 45 cm depth readings due to increased soil warming and ET from the growing winter wheat/rye crop (Figure 4.9). On March 7, 2007, the GEO1 wastewater dosing control system was successfully activated by the GP1 data logger/controller when the recorded 20 cm soil moisture readings decreased below 0.40 $m^3 m^{-3}$, even though 45 cm soil moisture was still above 0.40 $m^3 m^{-3}$. Operation of the wastewater SDI system continued thereafter amid increasing soil temperatures, storm events, and crop growth until May 25, 2007 when the winter wheat/rye mix was harvested (Figure 4.9). Since crop harvesting and planting relies on heavy machinery that cannot withstand wet field conditions, water dosing was manually disabled from May 25, 2007 to June 14, 2007 to ease harvesting of winter wheat and rye and planting of sorghum-sudangrass. After that, the experimental system was placed back on automatic control and prepared for the second year of the study (not included in this paper). The nearly one month cut off period required for agronomic crop management would be difficult to justify in a single household application, but may find application in a community sized decentralized system with larger available dispersal area. Considering the likelihood of a 2-month zero dosing period during wet winter conditions, at least a 2-month septic tank storage requirement is anticipated.

4.4.3 System Hydraulic Performance

An estimated monthly water balance is presented in Figure 4.10. Water was dosed during spring through fall, with a peak value of approximately 23.32 cm month^{-1} in April of 2007 during drought conditions. This peak dosing rate is almost four times higher than Alabama recommendations for hydraulic loading (6.00 cm month^{-1}) of clayey soils similar to the test site (ADPH, 2006). Since the experimental system was never limited by water supply except during malfunction from October 2006 to January 2007, recorded hydraulic dosing rates represent the maximum system hydraulic dosing rates can be achieved by the system under the pre-set soil moisture control thresholds. Advantages demonstrated by this experimental system include: 1) avoidance of wet soil conditions by withholding wastewater dosing until field moisture content drops to a pre-determined "operational" window; 2) temporary increase in wastewater hydraulic dosing with real-time soil moisture sensing under favorable field conditions.

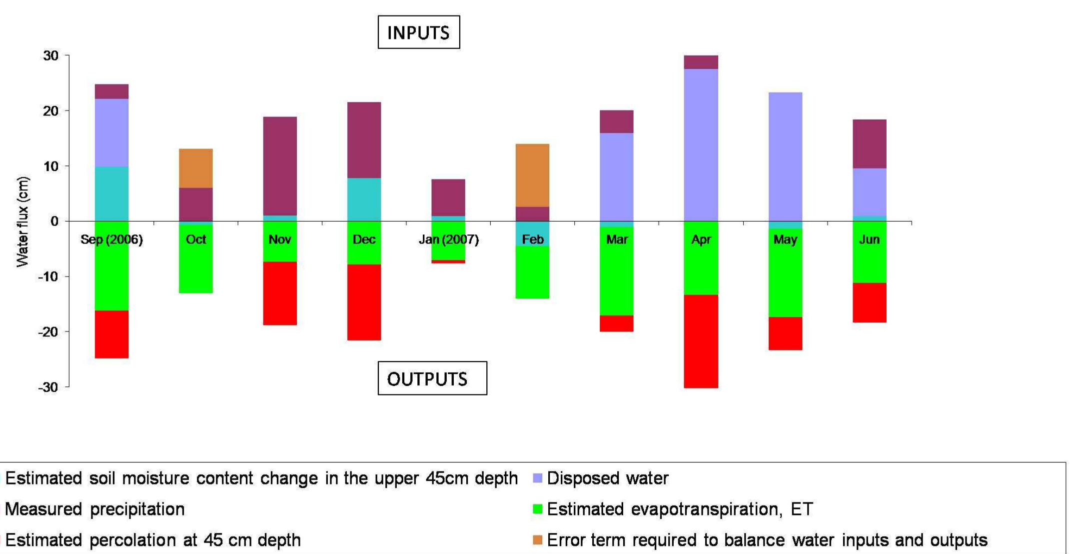

Figure 4.10 Estimated monthly drain field water balance from September 2006 to June 2007. (Drain field surface ponding is not considered. Positive Y axis represents water inputs; negative Y axis represents water outputs)

According to the monthly water balance, more than 30% of applied water percolated below 45 cm depth during the September 2006, November 2006 to December 2006, and March 2007 to June 2007. The estimated percolation below 45 cm depth during November 2006 to December of 2006 when the experimental system did not dose any water into the field indicates that the drain field is not suitable for wastewater dosing during normal winter months. On the other hand, the experimental system did not aggravate the already saturated soil conditions in the drain field during winter wet periods when most conventional septic systems would be experiencing drain field surface ponding.

The period from March 2007 to June 2007 coincided with a historic drought March through June precipitation was 248 mm versus 492 mm in an average year. During system startup in September 2006, the test site was also dry in the upper 45 cm (< 0.20 m^3 m^{-3}, Figure 4.9). It is recognized that shrinking and swelling of clay-rich smectitic soils create dynamic crack formations that change soil physical and hydraulic properties (Bouma et al., 1981). Preferential channels can form which alter the landscape hydrology and facilitate rapid transport of water into the soil (Bouma et al., 1981; Youngs, 1995; Kishne et al., 2009). Although the cracking extent of the clay soil at the test site was not quantified during this study, surface cracking was consistently observed. Cracking development to a depth of around 50 cm is normal for Vertisols (Amidu et al., 2007) and more than 100 cm depth crack development has been reported in Houston clays (Kishne et al., 2009).

Since the test site is a low permeable Houston clay soil, a possible explanation for the estimated percolation loss during the dry period of 2007 is that dosed water did not adequately moisten the upper soils so as to curtail soil crack development. Presumably, when the wetting front reached the soil cracks between the soil moisture sensors and the drip emitters, much of the water moved by preferential flow away from the soil moisture sensors, draining the soil profile at a higher rate than would have occurred in a more homogenized soil structure.

In order for soils to provide effective effluent treatment, water has to percolate through soil horizons above underground restrictive layers (AWOCT, 2005). For the experimental site, evidence of a seasonal water table exists at around 88 cm depth (Table 4.2), indicating that there is an additional 43 cm of soil below the 45 cm depth to mitigate environmental pollution at the site. Water percolation below 45 cm would be of more serious concern if a local water table existed.

The estimated magnitude of percolation at the experimental site suggests that the experimental soil moisture controlled hydraulic dosing system as designed was not effective in preventing clay soil shrinking during dry soil conditions. One reason for the experimental system's ineffectiveness in controlling soil cracking could be the 0.61 m spacing between emitters and drip lines. It is likely that the wetting diameter of the emitters did not fully moisten the field area leaving dry areas susceptible to cracking during dry soil conditions. Consequently, reduced emitter and drip line spacing may enhance water distribution, and limit soil cracking. Another possible reason that the system did not effectively prevent soil cracking is that there were limited field moisture monitoring locations used in this study. If the wetting influence from percolated water were monitored by soil moisture sensors at more frequent spacing and at deeper depths, preferential flow may be more effectively controlled. Recommendations for system improvements include the use of multiple soil moisture monitors vertically and horizontally to more adequately reflect drain field moisture conditions and site heterogeneity. Emitter and drip line spacing could also be reduced to increase soil moisture uniformity and limit crack development in the field. Above suggestions require additional field testing to evaluate their impact on system hydraulic performance, including water loss to preferential flow.

4.5 CONCLUSION

Assembly of an experimental soil-moisture based wastewater dosing system was completed by integrating two commercially available systems, a research grade soil moisture data logger/controller and a wastewater SDI control system. Wastewater SDI dosing was laboratory tested using two soil moisture sensors in a clay and sand media. During the laboratory test, the system responded to real-time readings of soil moisture as designed by 1) withholding water dosing during wet soil conditions of ≥ 0.45 m^3 m^{-3} or low reservoir condition, and 2) initiating and continuing water dosing under allowable soil conditions between 0.40-0.45 m^3 m^{-3}.

Once installed in the field, the experimental system responded to seasonally changing field conditions, effectively adjusting water disposal according to real-time soil moisture. Although the experimental system was internally robust, unexpected power outages shut down the system periodically, emphasizing the need to conduct regular system checks for successful operation. The experimental system effectively stopped water dosing during wet soil conditions and prevented surface ponding from system water dosing. The observed zero water dosing period during wet winter conditions indicates the need for a wastewater storage capacity of at least two months. This constraint likely creates an insurmountable challenge for application of this system for individual rural homeowners.

Observed water management of the experimental system indicated that more than 30% of applied water was lost to percolation below 45 cm during dry soil conditions, presumably a result of soil cracking. Water percolation loss indicates that the experimental system, including lateral and emitter spacing configurations and soil moisture monitoring and feedback control, is not able to effectively limit water percolation during dry soil conditions. This finding suggests that clay shrinkage left the system unable to control water percolation. In spite of the successful operation of the soil moisture based dosing control, the current experimental system did not overcome the severe hydraulic limitations inherent in the Houston clay soil. Recommendations for system improvement include the use of multiple soil moisture sensorsvertically and horizontally to more adequately reflect drain field moisture conditions and site heterogeneity. Emitter and drip line spacing can also be reduced in an attempt to limit soil crack development in the field. Above recommendations require further field testing to evaluate their impact on system hydraulic performance.

CHAPTER FIVE

SOIL MOISTURE CONTROLLED WASTEWATER DISPERSAL-YEAR TWO: SYSTEM HYDRAULIC AND NUTRIENT EVALUATION

5.1 Abstract

Rural areas represent approximately 95% of the 14000 km^2 Alabama Black Belt, an area of widespread Vertisols, dominated by clayey, smectitic, shrink-swell soils. The area is characterized by widespread use of onsite wastewater treatment systems (OWTS) that rely on soil for wastewater dispersal. An experimental field moisture controlled subsurface drip irrigation (SDI) system designed for integrated use with a seasonal cropping system was installed and evaluated for two years on a 500 m^2 Houston clay site in west central Alabama. From September 2006 to June 2007 (year one) clean well water was applied; from June 2007 to September 2008 (year two) a synthetic wastewater was used. The objectives of the study were to: 1) evaluate two-year system hydraulic management in terms of seasonal water disposal, water percolation control, soil moisture profile, and annual water budget; 2) evaluate system nutrient management in year-two in terms of monthly soil water nutrient level, seasonal and annual field crop nutrient uptake, and annual soil nutrient profile. System feasibility is addressed based on results of the two-year field experiment.

Hydraulic dosing rates during the two year study fluctuated as expected with higher dosing rates during warm season and lower dosing rates during cold season. Drain field surface ponding was not observed during dry warm seasons and was not aggravated by hydraulic dosing during the cold season. Estimated water percolation loss below 45 cm occurred in warm season during which time approximately 30% of dosed water was lost to percolation. Average hydraulic dosing rate during the warm season of year two was 0.17 cm d^{-1}, more than half the 0.40 cm d^{-1} rate observed during the same period of year one. Differences are likely due to the higher, more normal precipitation in year two. The estimated annual water balance based on two years of experimental data indicates the need for a minimum 2-month wastewater storage requirement, verifying that the system is not suitable as a stand-alone wastewater disposal and treatment method. Soil moisture profiles monitored during year two from May 2008 to September 2008 suggest that significant percolation did not occur below 100 cm.

Annual crop nutrient uptake represented approximately 32% of applied nitrogen and 31% of applied phosphorus during year two. Results suggest there was potential for nutrient (N, P) leaching during the experiment as a result of moisture based dosing. However, soil cores sampled near the end of the experiment provided no direct evidence of drain field N or P accumulation or percolation below 100 cm depth. Additional, replicated field sampling is required before definite conclusions can be drawn regarding the nutrient impact of wastewater dosing at this site.

In spite of operation of the soil moisture based dosing system, the experimental SDI system is not suitable for direct application of wastewater in clay soils of the

Alabama Black Belt. System limitations which need further study include: 1) near zero or zero low hydraulic dosing rates during wet soil conditions resulting in an impractical wastewater storage requirement; 2) lack of system response to seasonal soil shrink-swell resulting in potentially large dry weather percolation losses below 45 cm depth; 3) nutrient loading imbalance with respect to crop uptake due to soil moisture based dosing; and 4) field system and crop operation and maintenance requirements that are impractical for land owners with limited resources. Based on field observations, the recommended application of the experimental system is to supplement other wastewater treatment systems that can function during wet periods when this system provides little or no hydraulic dosing. Recommendations are made to improve experimental system control over water percolation below 45 cm.

5.2 INTRODUCTION

Onsite wastewater treatment systems (OWTS) in the US currently treat 15 million metric tons of wastewater per day, serving approximately 30% of U.S. households (Spicer, 2002; AOWTC, 2005). This widespread rural treatment and dispersal method functions as an important supplement to centralized public wastewater treatment plants (WWTP). Despite wide adoption, conventional OWTS consisting of a septic tank and a gravity fed effluent disposal field can pose a significant threat to the environment by polluting surface and groundwater with nutrients and pathogens (US EPA, 2002).

During the mid-1940s, infant mortality was linked to NO_3^--N concentrations in drinking water, especially in rural areas of the United States (Hergert, 1986; Fare, 1993).

Public hygiene issues related to onsite septic systems in urban fringe areas has received attention since 1980 (Boyle and Otis, 1981; DeWalle and Schaff, 1980). Typical nitrogen concentration in septic tank effluent ranges from 40-80 mg L^{-1} (Walker et al., 1973), of which approximately 75% is ammonium nitrogen and 25% is organic nitrogen (Otis et al., 1974). Reported total phosphorus concentration in septic tank effluent ranges from 3 to 20 mg L^{-1} (Robertson et al. 1998, Whelan 1988), with approximately 85% as orthophosphate (Reneau and Pettry 1976). Charles et al. (2005) analyzed the results of several intensive septic tank effluent field surveys from 1976 to 1999 in Australia and the US, including their own field survey of 200 septic tanks in Australia. To minimize nutrient overloading associated with most conventional OWTS failure, Charles et al. (2005) suggested a design value of 250 mg L^{-1} for total nitrogen and 36 mg L^{-1} for total phosphorus to estimate drain field nutrient load. They concluded that widespread nutrient overloading is likely occurring in a significant number of conventional OWTS designed under current Australian and the US regulations (Charles et al., 2005).

The failure of conventional OWTS in the Alabama Black Belt area is a widespread and recognized problem (McCoy et al., 2004; He et al., 2007). The most common cause of conventional OWTS failure in this region is low permeable smectitic soils characterized by shrink-swell (ADPH, 2006). Based on spatial analysis of SSURGO (NRCS, 2007) soils data with Alabama Onsite Sewage Disposal Rules (ADPH, 2006), over 77% of the Alabama Black Belt area is unsuitable for conventional OWTS mainly due to low soil permeability (ADPH, 2006; He et al., 2007). Furthermore, the severe shrinking observed in smectitic soils during extended dry periods can stimulate preferential flows in a field (Hoogmoed and Bouma, 1980; Beven, 1981). Preferential flow has been identified as a potential conduit for water and nutrient deep percolation that can threaten underground water systems (Weaver et al., 2005; Larsson et al., 2007; Muukkonen et al., 2009). Alternative engineered systems such as mound systems and drip irrigation systems receiving secondary treated effluent are currently mandated on new sites that are hydraulically limited to mitigate public health safety issues in the Alabama Black Belt (ADPH, 2006). Nevertheless, conventional OWTS are widely used in the Alabama Black Belt region partly because economic conditions in the region do not support retrofits or widespread replacement with more advanced systems (McCoy et al., 2004).

This study evaluates a commercially available subsurface drip irrigation (SDI) wastewater disposal system retrofitted with soil moisture feedback control. SDI was used to more uniformly distribute wastewater and to supply nutrients favorable for crop uptake (Phene and Ruskin, 1995). Drain field soil moisture control for wastewater disposal was incorporated as a proven technology for water percolation control in agricultural irrigation (Dukes and Scholberg, 2005). A managed cropping system for enhanced water, nutrients, and contaminants removal was incorporated into the drain field design to provide enhanced water and nutrient uptake (Askegaard and Eriksen, 2008; Ferraro et al., 2003; Giupponi, 1998; Wang et al., 2008).

In this study, an experimental soil moisture controlled SDI wastewater dosing system was field tested on a 500 m^2 Houston clay soil using clean well water from September 06, 2006 to June 13, 2007 (year one), and with a synthetic wastewater from

June 14, 2007 to September 8, 2008 (year two). The objectives of the study were to: 1) evaluate two-year system hydraulic management in terms of seasonal water disposal, water percolation control, soil moisture profile, and annual water budget; 2) evaluate system nutrient management in year-two in terms of monthly soil water nutrient level, seasonal and annual field crop nutrient uptake, and annual soil nutrient profile. System feasibility is addressed based on results of the two-year field experiment.

5.3 MATERIALS AND METHODS

5.3.1 EXPERIMENTAL DESIGN AND OPERATION

SITE SOIL PHYSICAL AND CHEMICAL PROPERTIES

The experimental site is located at the Alabama Black Belt Research and Extension Center, 11 miles west of Selma in west central Alabama. Soil at the site is a Houston clay (very-fine, smectitic, thermic Oxyaquic Hapludert). Measured soil physical and chemistry properties are summarized in Table 5.1. The soil profile is dominated by clay with CEC ranging from 23-30 meq 100 g^{-1}. Porosity ranges from 52-64%. The soil profile has a phosphorus adsorption capacity of approximately 17000 mg P kg^{-1} soil.

Table 5.1 Soil physical and chemical properties at the experimental site.

Soil horizon	Depth (cm)[1]	Particle Size Distribution[1], %			Ave. CEC[2] cmol kg^{-1} ±1SD	Bulk density from NRCS web soil survey[3] g cm^{-1}	Ave. Particle density[4] g cm^{-3}±1SD	Ave. Porosity[6] %	Maximum soil phosphorus adsorption capacity[6] mg P kg^{-1} soil
		Sand	Silt	Clay					
Ap1	0-20	7.09	39.63	53.28	29.11±1.82	1.10-1.45	2.62 ±0.01	63±1	17000
Ap2	20-40	8.27	38.04	53.70	25.48 ±0.13	1.10-1.45	2.59 ±0.01	58 ±4	17000
BA	40-60	10.17	33.38	56.45	23.81 ±0.61	1.10-1.45	2.59 ±0.01	55±2	17000
Bkss1	60-80	3.50	35.93	60.57	24.70 ±1.41	1.10-1.45	2.62 ±0.02	59±1	17000
Bkss2	80-100	3.10	25.80	71.10	25.67 ±0.19	1.10-1.45	2.60 ±0.03	54±2	17000

Notes:

1. Determined by the pipet method (Soil Survey Investigation Staff, 2004).
2. Determined by the ammonium acetate (pH 7) method (SSSA, 2002a) using composite soil subsamples from the soil cores sampled from all four field treatments taken on June 24, 2008. (1 cmolc kg^{-1} equals to 1 meq 100 g^{-1}).
3. NRCS Web Soil Survey version 2.1, survey area data version 4, September 16, 2008.
4. Determined with liquid displacement method (SSSA, 2002b) using composite soil subsamples from the soil cores of all four field treatments taken on June 24, 2008.
5. Calculated from bulk density from NRCS and lab measured particle density.
6. Determined by the method of Self-Davis et al. (2000) using composite soil subsamples from the soil cores of all four field treatments sampled on June 24, 2008.

SDI DESIGN AND FIELD TREATMENTS LAYOUT AND OPERATION

Based on Alabama Department of Health (2006) regulations for onsite sewage disposal, the allowable hydraulic loading (dosing) rates for the Houston clay soil is 2.04 lpd m^{-2}. The design flow rate for the experimental field layout was 1022 liters per day, equivalent to the daily wastewater flow of a 3-person home in a decentralized subdivision system (Alabama Department of Public Health, June 14, 2005, personal communication). Consequently, a total drainage or disposal soil area of 500 m^2 was required for this design. Based on standard practice of a 0.61 m spacing between drip lines and 0.61 m emitter spacing (Geoflow, 2003), a total of 823 m of drip line was required.

The SDI system consists of 30 drip tubes, WFPC16–2–24, 16 mm diameter (Geoflow, CA), 27 m long at 0.61 m lateral spacing installed approximately 20-25 cm deep (Figure 5.1). The maximum particle size that can pass through emitters without obstruction is 100 µm. The SDI system was supplied by well water stored in a 7600 L above-ground plastic septic tank (Fralo, NY). The SDI system was hydraulically divided into two subplots (I and II, Figure 5.1), each with 15 drip tubes. The design flow rate of the entire SDI system was 76 lpm. The experimental site was disked to a depth of 20-25 cm before SDI installation to reduce site heterogeneity in the top soil and to provide more friable conditions for tube installation.

The soil moisture operating thresholds were 0.40 to 0.45 m^3 m^{-3}, identical to the laboratory test. The SDI wastewater controller was set to a 5-minute dosing period followed by a 55-minute resting period, providing approximately one dosing cycle per hour. Since only clean well water was used in this study, a forward field flush was programmed to occur only every 1000th cycle. Dosing and field flushing were followed by a 15-second filter flush and a 5-second pump delay, per manufacturer's recommendations. The GP1 data logger/controller was set to record rainfall (mm), soil moisture (v/v), flow volume (gallon), voltage (v), and temperature (degree C) every 15 minutes.

Based on the NRCS Web Soil Survey (NRCS, 2008), the field capacity (1/3 bar) at the 45 cm of the experimental site is 0.42 $m^3 m^{-3}$. Consequently, the soil moisture ($m^3 m^{-3}$) thresholds used for SDI control were set at 0.40 (on) and 0.45 (off) with the intent to avoid hydraulic overloading the experimental site while maintaining adequate soil moisture for managed crop uptake and aerobic soil treatment for wastewater. System hydraulic disposal occurred when either of the two soil moisture sensors read < 0.40 $m^3 m^{-3}$. System hydraulic dosing was not enabled when either of the two soil moisture sensors read above 0.45 $m^3 m^{-3}$.

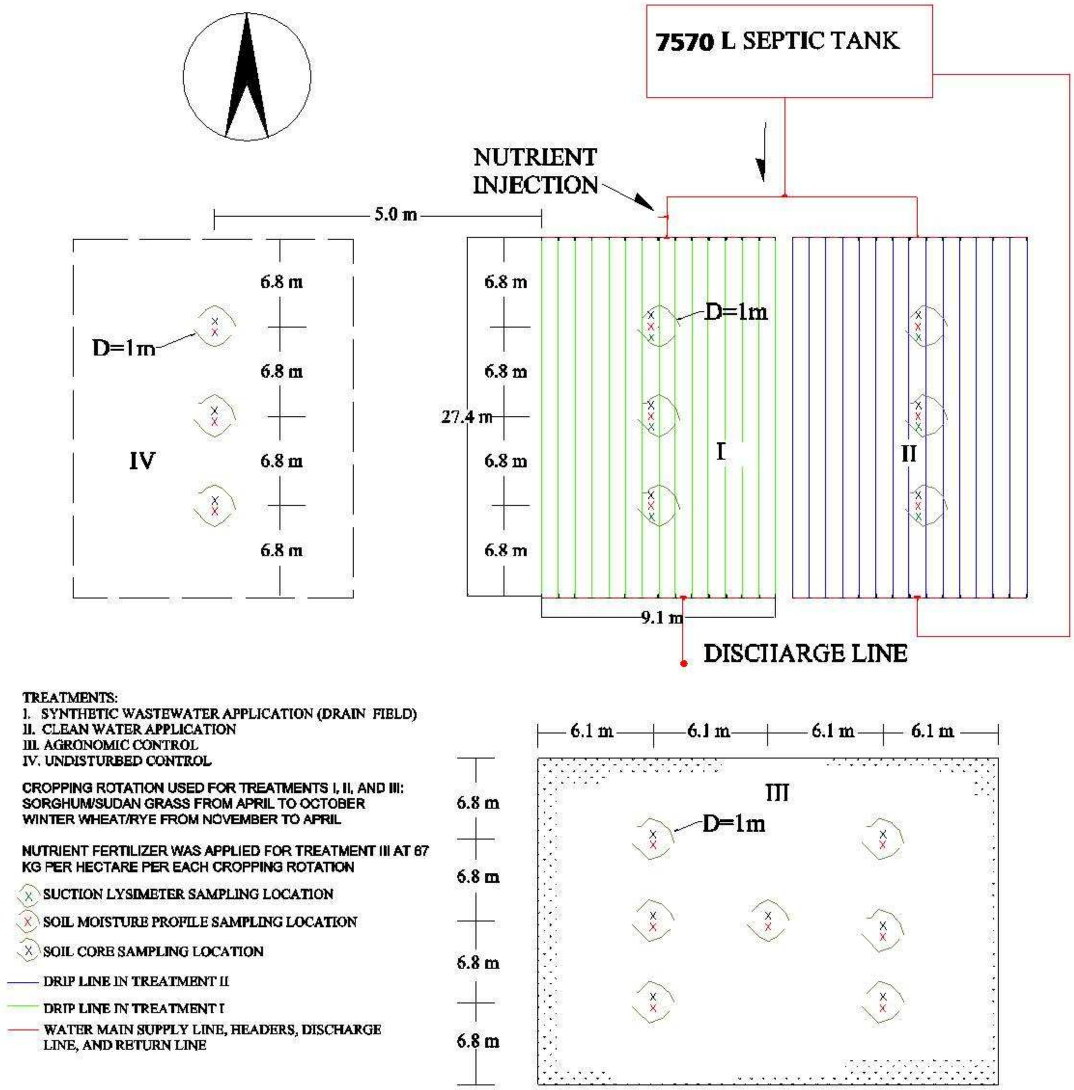

Figure 5.1 Experimental layout, treatments, and sampling locations.

The experimental SDI system (Treatments I and II) was operated with clean well water from September 6, 2006 to June 13, 2007. Starting from June 14, 2007 until June 24, 2008, Treatment I received synthetic and Treatment II continued to receive clean well water. A 100 µm mesh filter was installed on the main water supply to the SDI system to screen well water of particles before entering drip lines. Synthetic wastewater was prepared by dissolving a commercial 30-10-10 fertilizer in a 1140 L nurse tank with well water at a ratio of 113 kg fertilizer to a full tank. Synthetic wastewater was then screened with a second 100 µm mesh filter before injection into the treatment I supply line. The chemical injection pump (Neptune, Japan) for treatment I was operated at a flow rate of 14.2 lph whenever main SDI dosing pump came on. Synthetic wastewater entered the drip lines at a ratio of approximately 1:160 (synthetic wastewater: clean well water). The resulting nutrient content was approximately 80 mg TN L^{-1}, 10 mg P L^{-1}, and 100 mg TOC L^{-1} throughout year two. Table 5.2 presents chemical analyses of clean well water and prepared fertilizer solutions from five dates throughout the study.

Treatment I received all crop water and nutrient supply from rainfall and synthetic wastewater. Treatment II received rainfall and clean well water to indicate potential soil water nutrient increase in treatment I due to wastewater application. Treatment III, 27 m × 18 m plot was used as an agronomic control to indicate differences in treatment nutrient uptake efficiency and soil nutrient profiles. Treatment III was not irrigated but received surface applied fertilizer at the beginning of each crop growing season (2 seasons $year^{-1}$, 67 kg N ha^{-1} $season^{-1}$). An identical crop rotation, sorghum-sudangrass (*Sorghum bicolor*), winter wheat (*Triticum aestivum*) and rye (*Secale cereale*) mix, was rotated in treatments I, II, and III (Table 5.3). Treatment IV, an undisturbed area of approximately 242 m^2 west of treatments I and II (Figure 5.1), was used to indicate the background soil conditions.

Table 5.2 Chemical analyses of clean well water and injected fertilizer solution (in average, mg L^{-1}) during year two synthetic wastewater experiment, June 14, 2007 to June 24, 2008.

Measured ingredients	Solution	6/15/2007[1]	8/03/2007[1]	9/28/2007	3/20/2008	6/16/2008[1]
NH_4^+-N	Well water	0.27	0.28	0.31	0.29	1.67
	Nutrient solution	855	1280	16.2	810	1550
NO_3^--N	Well water	N.D.[2]	N.D.	N.D.	N.D.	0.44
	Nutrient solution	27.7	38.0	N.D.	N.D.	24.3
TKN	Well water	N.D.	N.D.	N.D.	N.D.	0
	Nutrient solution	15700	N.A.[3]	N.A	N.A	N.A
Total phosphorus	Well water	N.D.	N.D.	N.D.	N.D.	0.12
	Nutrient solution	1920	2420	2800	1280	1890
TOC	Well water	<1	<1	<1	<1	<1
	Nutrient solution	22600	18400	21300	22600	23000
pH	Well water	5.32	6.13	6.59	6.51	6.40
	Nutrient solution	8.19	8.20	8.50	7.63	8.10

1. Fresh nutrient solutions prepared on these dates.
2. None Detectable.
3. Not Available. TKN was not measured on these dates and assumed approximately close to that of 6/15/2007.

Table 5.3 Crop rotation of treatments 1-IV during the second year synthetic wastewater experiment, June 14, 2007 to June 24, 2008.

Field treatments	Warm season[1] rotation	Cool season[2] rotation	Fertilizer application	Irrigation
Treatment I (Synthetic wastewater application)	Two seasons of Sorghum-sudan grass growth (Jun 07- Aug 07, and Aug 07- Nov 07)	One season of wheat/rye mix (Nov 07-Jun 08)	Proportional to value of applied wastewater	SDI synthetic wastewater
Treatment II (Clean well water application)	Two seasons of Sorghum-sudan grass growth (Jun 07- Aug 07, and Aug 07- Nov 07)	One season of wheat/rye mix (Nov 07-Jun 08)	None	SDI clean well water disposal
Treatment III (Agronomic control)	Two seasons of Sorghum-sudan grass growth (Jun 07- Aug 07, and Aug 07- Nov 07)	One season of wheat/rye mix (Nov 07-Jun 08)	67 kg N ha^{-1} at the beginning of each crop growing season	None
Treatment IV (Undisturbed control)	No crop	No crop	None	None

1. Sorghum-sudan grass was planted at 33.6 kg seeds ha^{-1} at 18 cm spacing.
2. Winter wheat was planted at 67.2 kg seeds ha^{-1} and rye was planted at 22.4 kg seeds ha^{-1} at 18 cm spacing.

5.3.2 SYSTEM WATER MANAGEMENT CASE STUDY - YEARS ONE AND TWO, SEPTEMBER 6, 2006 TO JUNE 24, 2008.

MONTHLY WATER BALANCE DEVELOPMENT- YEARS ONE AND TWO

A monthly water balance was developed for treatment I from September 6, 2006 to June 24, 2008 to evaluate the impact of seasonal soil moisture on automatic system control. Components of the calculated water balance include depth of disposed water, precipitation, evapotranspiration, percolation below 45 cm depth, and water content change in the upper 45 cm, as described in Chapter 4, without considering of drain field surface ponding.

SITE K_{SAT} AND FIELD CAPACITY VERIFICATION - YEAR TWO

After the end of year two, two site uniformity of treatments I and II (Figure 5.1) was quantified in terms of field capacity and soil saturated hydraulic conductivity (K_{sat}) to determine how closely field conditions conformed to original system design parameters. Field capacity was measured on June 24, 2008 by taking intact soil cores at 20 cm depth from nine uniformly distributed locations. Volumetric soil moisture at field capacity (1/3 bar) was measured using the laboratory pressure plate method (SSSA, 2002b). K_{sat} was also measured onsite during June-July, 2008 using a permeameter (Ksat Inc., CA) at six uniformly distributed locations at 45 cm depth. Resulting site maps of field capacity and K_{sat} distribution were generated using inverse distance weight (IDW) method within a GIS (ArcMap9.2, ESRI, CA). The Christiansen uniformity coefficient (C_u) (Soil Conservation Service, 1970) was calculated for both field capacity and K_{sat} to quantify site uniformity for these two important hydraulic parameters.

SOIL MOISTURE PROFILES - YEAR TWO

A pre-calibrated capacitance soil moisture profiler (PR2, Delta-T, UK) was used to record soil moisture profiles at three locations on eight spate dates in each of the four treatments (Figure 5.1) at depths of 10, 20, 30, 40, 60, and 100 cm. Soil moisture profiles were obtained from May 2008 to September 2008 at approximately bi-weekly intervals. The soil moisture profiler provided reliable readings only when soil moisture content was below saturation. Profile measurements of volumetric soil moisture were used to identify if water moved through the profile as a consequence of hydraulic dosing.

WASTEWATER STORAGE ESTIMATION - YEAR ONE AND YEAR TWO

A wastewater storage requirement was estimated for the experimental system based on observed monthly hydraulic disposal rates in year one and year two. Historic precipitation from January 1978 to June 2008 was obtained from the Alabama Black Belt Research and Extension Center and compared to precipitation recorded onsite during the two-year study period. The allowable hydraulic dosing rate for a calendar month was estimated as the dosing rate observed for the month whose monthly precipitation fell closet to the 30-year average (50th percentile). Theoretical monthly residential wastewater supply was approximately 0.20 cm d^{-1} (6.00 cm $month^{-1}$) based on the allowable hydraulic dosing rate for Houston clay by Alabama Department of Public Health (ADPH, 2006). An annual wastewater storage requirement in terms of equivalent months of residential flow was determined as the cumulative difference between consistent monthly wastewater flows and allowable monthly wastewater dosing rates. A zero annual balance between monthly inflows and outflows does not gurantee a zero storage requirement. However, a zero balance indicates that the site is capable of absorbing all wastewater over the year in spite of prolonged periods of wet weather zero dosing.

5.3.3 SYSTEM NUTRIENT MANAGEMENT CASE STUDY – YEAR TWO (JUNE 14, 2007 TO JUNE 24, 2008)

N AND P LEVELS IN SOIL WATER - YEAR TWO

Suction lysimeters (Irrometer, CA) were installed at depths of 15, 30, and 45 cm at three locations in treatments I and II (Figure 5.1). Soil water samples were collected once per month from August 2007 to June 2008 during year two. A 50-60 kPa vacuum was applied to each lysimeter and allowed to sit 3-4 hours before the sample was collected in a clean 120 ml HDPE bottle. Samples were stored at 4 ^{o}C until analyzed. All lysimeter samples were filtered through 0.45 um membrane filters before their chemical analyses. Ammonium-N (NH_4^+-N) and nitrate-N (NO_3^--N) concentrations were determined with colorimetric analysis using a microplate reader (Sims et al., 1995). Total phosphorus (P) concentration was determined by inductively coupled argon plasma spectroscopy (ICAP 9000, Thermo Jarrel Ash, Franklin, MA).

CROP GROWTH AND NUTRIENT UPTAKE - YEAR TWO

Each season, harvested crops from treatments I and III were measured for total fresh and dry matter yield (105 °C) since they were the only two treatments that received supplemental nutrients (Table 5.3). To determine total N and P in plant tissue, dried plant samples were ground to pass a 1-mm mesh screen using a Wiley Mill (Thomas Scientific, PA). Samples were dry-ashed, digested with HCl (Hue and Evans, 1986) and analyzed via ICAP (ICAP 9000, Thermo Jarrel Ash, Franklin, MA). Above ground crop uptake of N and P was estimated by multiplying plant nutrient content by crop dry matter yield.

Soil core sampling at the end of year two

One meter long soil cores were taken from all four treatments on June 24, 2008 after winter wheat and rye harvest, approximately one-year from the June 14, 2007 start of synthetic wastewater application. Soil cores were collected using a tractor-mounted Giddings® hydraulic probe at three locations in each treatment (Figure 5.1). At the laboratory, each soil core was divided by depth into five subsamples: 0 to 20, 20 to 40, 40 to 60, 60-80, and 80-100 cm. Subsamples were dried in a ventilated oven (Heraeus, US) at 60 ^{o}C for four days, pulverized and screened to pass a 2-mm sieve. Total soil C and N was quantified by combustion using a LECO CHN-600 analyzer (LECO Corp., St. Joseph, MI). Total soil P was quantified using the perchloric acid procedure of Shelton and Harper (1941). Crop available P was determined using the Mississippi extract method (Lancaster, 1970) and analyzed by ICAP (ICAP 9000, Thermo Jarrel Ash, Franklin, MA). Soil pH was measured using 1:1 soil/water (m/m) slurries with a pH meter (Orion,US). Crop available N was determined by extracting soil subsamples with 1M KCl solution and analyzing the extract for NH_4^+ and NO_3^- (Sims et al., 1995). Water soluble P for each subsample was measured using the method of Self-Davis and Moore (2000). Subsamples of soil cores from all four treatments were composited by horizon to measure soil cation exchange capacity (CEC) using the ammonium acetate (pH=7) method (SSSA, 2002a). Soil maximum P adsorption capacity and soil P adsorption coefficient (K_d) were determined using the same composite samples and methods by Self-Davis et al. (2000).

5.4 RESULTS

5.4.1 SYSTEM WATER MANAGEMENT

FIELD OBSERVATION OF SYSTEM HYDRAULIC RESPONSE- YEARS ONE AND TWO (SEPTEMBER 6, 2006 TO JUNE 24, 2008)

Hydraulic disposal rates and soil moisture at two sampled depths from years one and two, September 6, 2006 to June 24, 2008, are presented in Figure 5.2a. Soil temperature at 10 cm, field ET, and daily precipitation for the same period are illustrated in Figure 5.2b. The experimental SDI dosing system did not function due to onsite power outage and water supply cutoff from October 2006 to January 2007 (Figure 5.2a). In addition, the experimental system was cut off manually for approximately one month in May and October to facilitate field crop harvesting and planting. For the remainder of the 2-year study period, SDI dosing was controlled by the soil moisture feedback system. Generally, the experimental system response to changing soil moisture was consistent throughout years one and two.

Throughout years one and two, relatively higher dosing rates and frequencies from were observed from late spring to late autumn as expected, with consistent near zero dosing periods during wet, winter months. System hydraulic dosing in year one was of higher magnitude and frequency than in year two. The highest hydraulic dosing rate, 1.18 cm d^{-1} occurred in April 2006. The average hydraulic dosing rate during the period from February 2007 to June 2007 was approximately 0.40 cm d^{-1}. There were almost 3 months

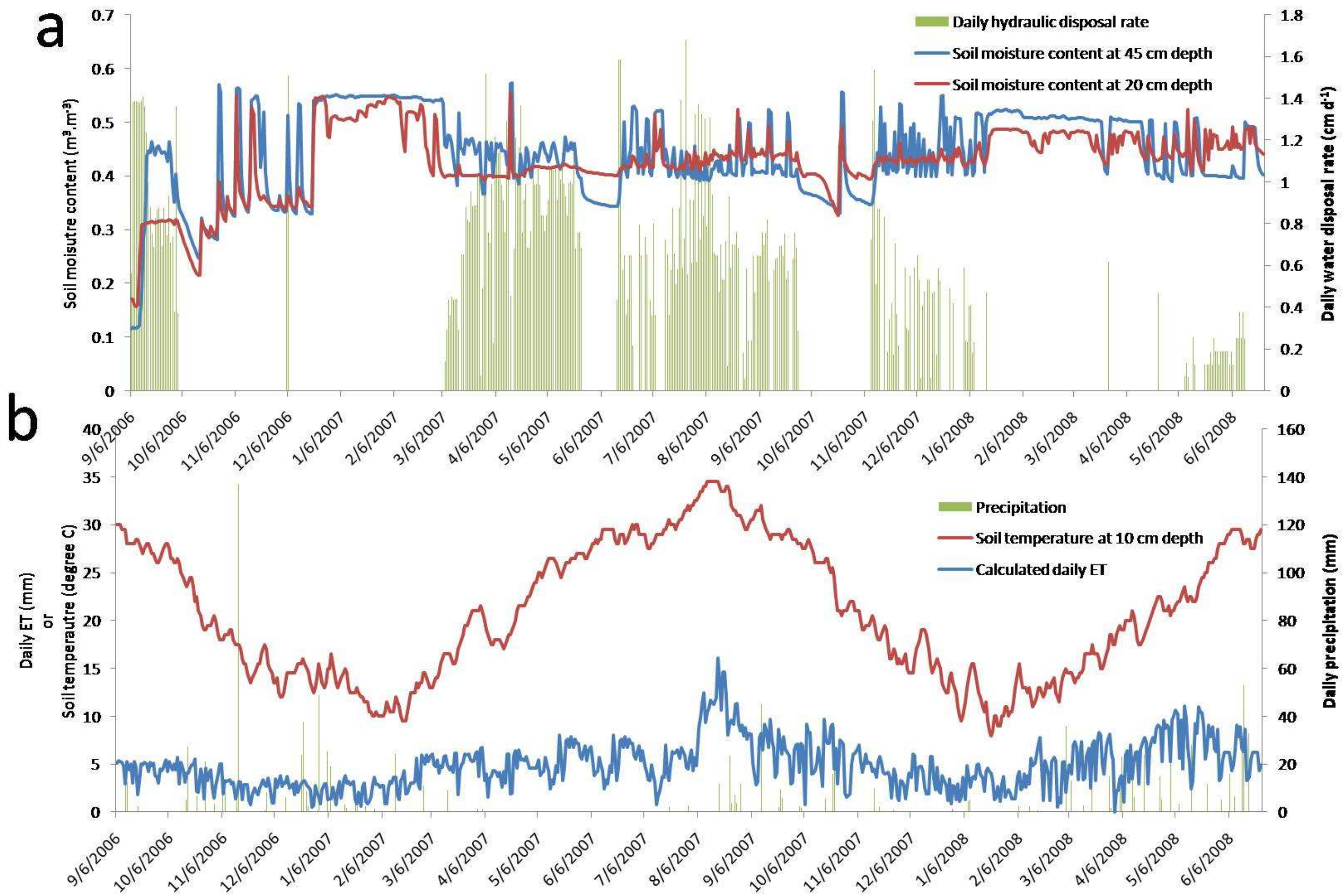

Figure 5.2 Recorded field data for years one and two, September 2006 to June 2008. (a. soil moisture content and daily hydraulic dosing rate; b. daily precipitation, soil temperature at 10 cm depth, and calculated daily ET. System operation was interrupted by unexpected power and water supply outages from October 2006 to March 2007. The system was manually shut off in May 2008 and October 2008 to facilitate crop harvesting and planting.)

during the same period in 2008 when there was almost no dosing. In fact, the average hydraulic dosing rate during this period was only 0.17 cm d^{-1}. The experimental system did not aggravate periods of drain field saturation during the wet winter of year two when the drain field was naturally saturated. The demonstrated advantages: 1) avoid wet soil conditions by withholding wastewater dosing until field moisture content drops to a pre-determined "operational" window; and 2) temporary increase wastewater hydraulic disposal with real-time soil moisture sensing based on seasonal soil conditions under favorable field conditions) of this the soil moisture based water management strategy in year one, are still applicable to the year two. However, the experimental system had a longer observed wastewater withholding period and a lower seasonal water dosing capability in year two.

As indicated from the monthly water balance table (Figure 5.3), water percolation loss during warm seasons below 45 cm was an unexpected consequence of operating the system. More than 30% of dosed water was estimated lost to percolation below 45 cm depth during year one (September 2006, winter of 2006/2007, March to June 2007). However, this large percolation loss was not observed during the same period of year two except for June 2008. Similar to the hypothesis proposed in Chapter 4 that it was soil cracking during the dry season that caused percolation loss, observed percolation during June 2008 is also believed to be caused by the same reason.

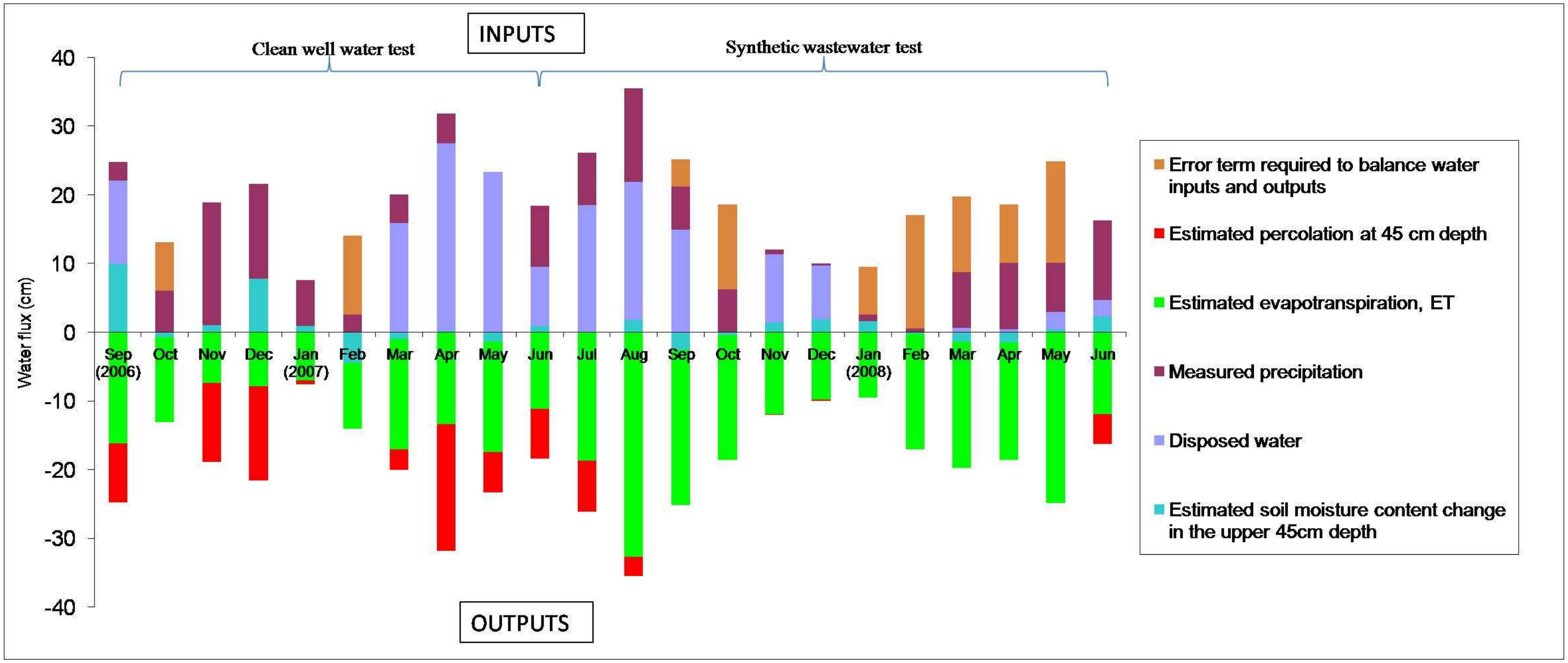

Figure 5.3 Estimated monthly drain field (treatment I) water balance from September 2006 to June 2008. (No consideration of drain field surface ponding. Positive Y axis represents water inputs; negative Y axis represents water outputs)

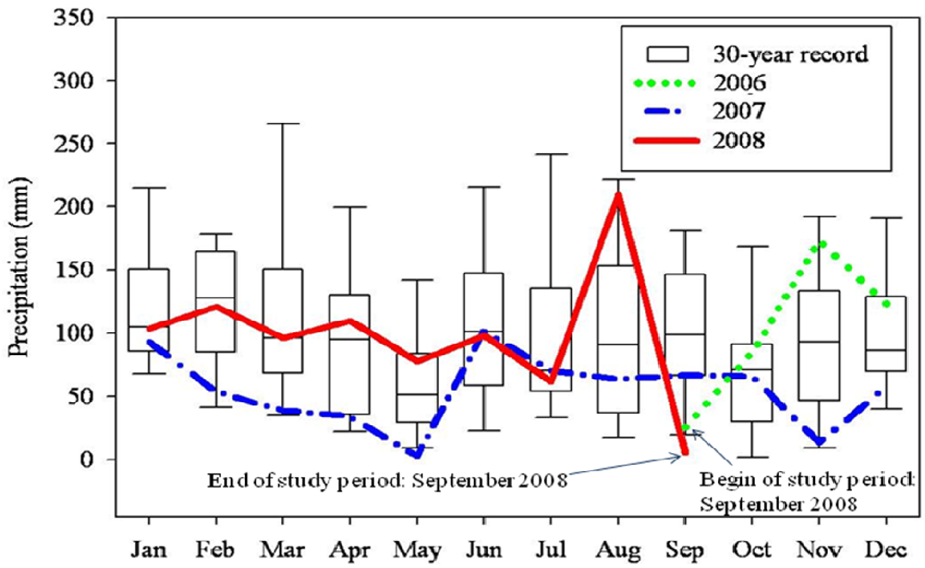

Figure 5.4 Observed monthly precipitation at experimental site versus 30-year precipitation record (1997-2008). (Source: Alabama Black Belt Research and Extension Station)

Different hydraulic dosing rate and water percolation loss between years one and year two requires an explanation. There was a significant difference in precipitation between years one and two. The period from March 2007 to June 2007 coincided with a historic drought; 248 mm precipitation versus 492 mm in an average year (Figure 5.4). The month of September 2006 was the system startup period during which time the test site was relatively dry in the upper 45 cm (below 0.2 m^3 m^{-3}, Figure 5.2). Observed data also indicate percolation below 45 cm depth in June 2008 likely due to increased rainfall and correspondingly reduced ET (Figure 5.3). However, more normal and higher rainfall returned during the same period in 2008 (Figure 5.4). Therefore, observed higher hydraulic dosing rates in the drought season of year one and the lower dosing rates during the normal rainfall warm season in year two can be explained by the significant weather change between years one and two.

Since multi-year rainfall variability is expected impact soil cracking on natural soils (Kishne et al., 2009), it is likely that soil cracking developed during the warm season in 2008 was not as severe as during the drought of 2007. As a consequence, percolation due to preferential flow would be lower in 2008 than in 2007, as indicated in Figure 5.3.

SITE SOIL HYDRAULIC PARAMETERS VERIFICATION - YEAR TWO

Measured field capacity in treatments I and II varied from 0.37 to 0.44 (m^3 m^{-3}) (Figure 5.5a), close to system operational thresholds (0.40 – 0.45 m^3 m^{-3}). The Christiansen uniformity coefficient for field capacity was 96.9%, indicating a high uniformity for that texture dependent soil parameter. Measured K_{sat} of the experimental site varied from 0.12 to 0.29 μm s^{-1} with relatively higher values in the upper slope section (Figure 5.5b). The Christiansen uniformity coefficient for K_{sat} was 76.2%, indicating a uniformly low permeability. In fact, the Houston clay is rated “Extreme” for conventional septic systems by the Alabama Department of Public Health, meaning that water percolation is extremely low in this type of soil and conventional septic systems are not suitable on this type of soil. Field soil testing indicated that permeability and field capacity corresponded well to system design and operational thresholds. This result provides further evidence that soil cracking likely stimulated preferential flows that caused substantial percolation loss during dry seasons.

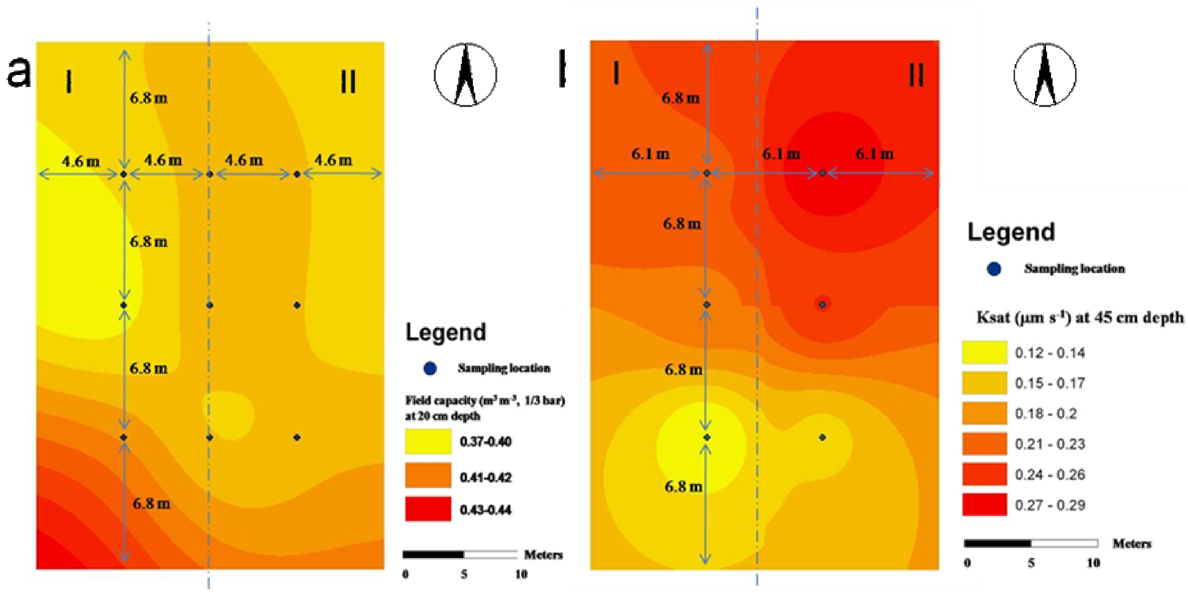

Figure 5.5 Spatial variation of (a) field capacity and (b) saturated hydraulic conductivity, K_{sat}, in the SDI drain field (treatments I and II). Soil hydraulic conductivity was measured onsite using a permeameter (Ksat Inc., CA) at 45 cm depth. Field capacity was measured using the laboratory pressure plate method on intact soil cores (SSSA, 2002b) at 18 cm depth.

Recommendations for system improvements, similar to those reported in Chapter 4, are to include the use of multiple soil moisture monitors vertically and horizontally to more adequately reflect drain field moisture conditions and site heterogeneity. Emitter and drip line spacing could also be reduced to increase soil moisture uniformity and limit crack development in the field. Above suggestions require additional field testing to evaluate their impact on system hydraulic performance, including water loss to preferential flow.

SOIL MOISTURE PROFILE – YEAR TWO

As demonstrated by the 30-year historic monthly precipitation record (Figure 5.4), during the summer time of year two (May to September 2008), normal to high rainfall returned (77.9 - 230 mm month^{-1}), followed by a low rainfall (43 mm) in September 2008.

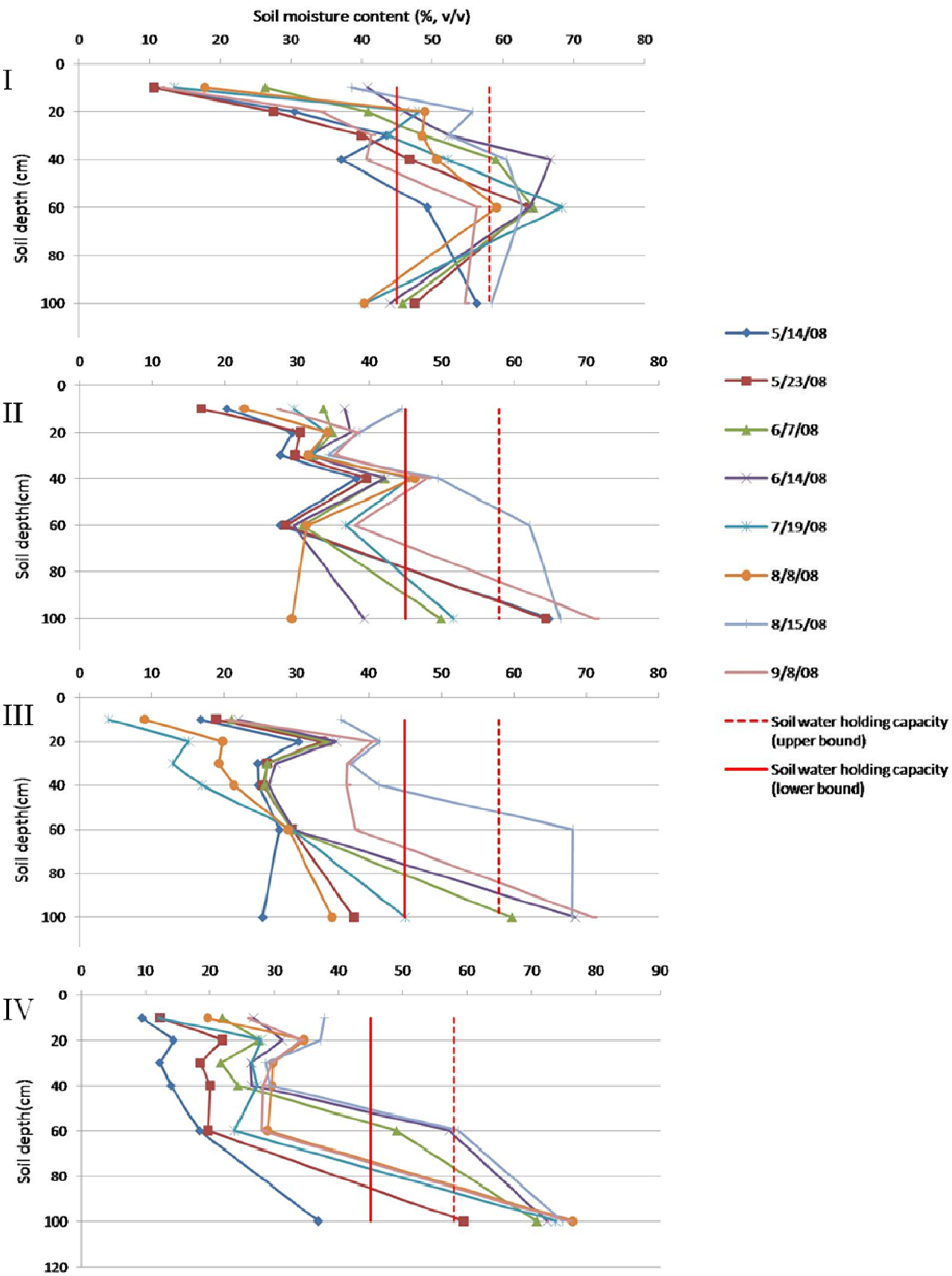

Figure 5.6 Year two soil moisture profiles, May 14-September 8, 2008. I. Synthetic wastewater application subplot; II. Clean well water application subplot (control); III. Agronomic practice subplot (control); IV. Undisturbed (blank) control. .

Soil moisture content in the upper 20-60 cm during this period was at least 0.10 m^3 m^{-3} greater in irrigated treatments I and II compared to treatments III and IV (Figure 5.6). Since water was dispersed in treatments I and II only when field moisture content fell below 0.40 m^3 m^{-3}, soil moisture observations document that soil moisture controlled dosing successfully managed soil moisture levels during the monitoring period. Interestingly, even though treatments I and II received the same volume of water, treatment I had a average higher soil moisture content (50 m^3 m^{-3} versus 35 m^3 m^{-3}) than treatment II within the 20-60 cm depth. This unexpected difference may have resulted from differences between treatments in their field capacity or K_{sat} (Figure 5.5).

Since soil moisture profilers provide false volumetric soil moisture readings above saturation, obtained profile readings are valid only if below the porosity of the monitored soil profile. NRCS web soil survey (NRCS, 2008) was used to estimate the porosities of the soil profile at the site, with 0.45-0.58 m^3 m^{-3} determined as the upper and lower bounds (Figure 5.6). Recorded soil moisture readings above soil porosity were observed in treatments I and II between 20-60 cm depth, possibly caused by water stored within soil cracks or at the profile tube soil interface. Numerous soil moisture readings at 100 cm depth were found higher than estimated soil porosity in treatments II, III, and IV, suggesting that free water may have accumulated at the sampled depths (personal correspondence, M. McClung, Nov 20, 2008). During the one meter coring, the presence of chalk was noted at or near 100 cm in all treatments except treatment I. According to the Geological Survey of Alabama (1993), the experimental site is underlain at a depth of approximately 6 m by a relatively impermeable layer of fossiliferous clayey chalk and chalky marl, with shallower formations found reverently at 12 cm to 2 m. Therefore, it is possible that a shallow restrictive layer (more restrictive than Houston clay in terms of water percolation) of clayey chalk near 100 cm depth in treatment II-IV. The water percolated from above layers may accumulate right on top of this restrictive layer and formed a water table. The soil moisture profile for treatment I indicates an increasing moisture trend to approximately 60 cm followed by a decreasing moisture trend to 100 cm depth. In the absence of preferential flow, this moisture profile would be expected to distribute dosed wastewater effectively throughout the soil profile.

WASTEWATER STORAGE REQUIREMENT- YEAR ONE AND YEAR TWO

Theoretical hydraulic dosing rates fluctuated throughout years one and two, while household wastewater flows were held constant. Consequently, where the monthly water balance indicated a negative balance it was necessary to provide wastewater storage requirement. Maximum allowable dosing rates for each month were assigned to those months with monthly precipitations closest to the 30-year historical normal (Table 5.4, Figure 5.7).

Determination of wastewater storage volume requirement in terms of a monthly effluent flow assumed that there was already a two-month wastewater storage requirement at the beginning of January from the previous winter. Required monthly wastewater storage increased or decreased as maximum allowable wastewater dosing rates fluctuated (Table 5.4). Wastewater storage requirement increased annually from January to June, indicating a maximum wastewater storage requirement of approximately seven months in August. After that, the wastewater storage requirement decreased, but ended the year with a three-month wastewater storage requirement that will be carried into the next calendar year. Table 5.4 indicates a major limitation of the Black Belt clay soils that over the two-year study period on average, only 4 months out of 12 had an observed positive water balance in favor of soil adsorption.

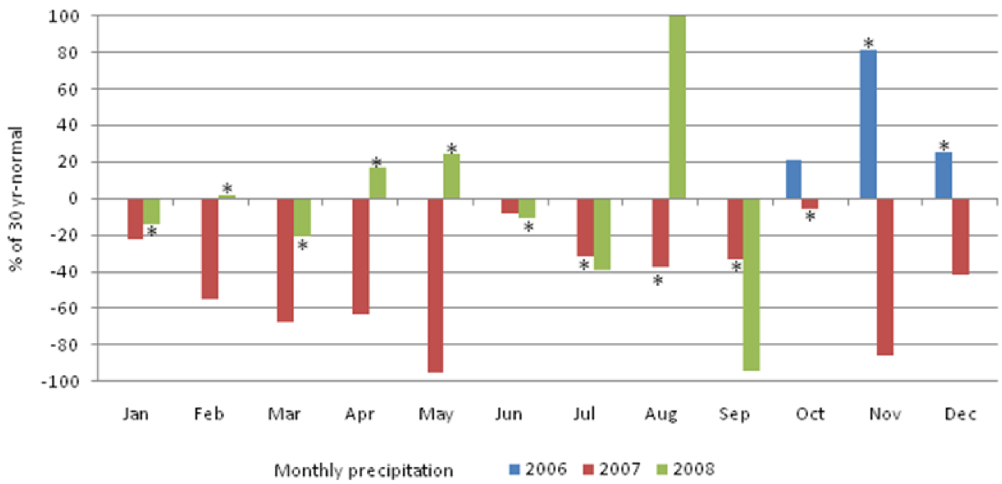

Figure 5.7 Year one and two monthly precipitation, October 2006 to September 2008, compared to 30-year normal. September 2006 is excluded since it is a system startup period. July 2008 to September 2008 is included as comparison to the records in the same period of 2007. Asterisks indicate the recorded month and year closest to 30-year normal, subsequently used to estimate average monthly dosing rate.

January and February zero dosing periods observed in year one and year two caused by naturally saturated soil conditions that curtailed wastewater disposal. The zero dosing period in October was a manual shut down required to facilitate warm season harvesting and cool season planting. If the wastewater dosing rate could be increased by expanding drain field size to offset the storage requirement, the wastewater storage requirement can be reduced and fixed. However, if a zero annual water balance could be achieved, a minimum two-month wastewater storage requirement still exists to accommodate during wet winters. Advantages of this experimental SDI system can be

Table 5.4 Monthly wastewater storage requirement based on observed monthly hydraulic disposal rates and Alabama Onsite Sewage Disposal Rules (ADPH, 2006).

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec	Annual summary
Maximum allowable monthly hydraulic wastewater dosing rates[1] in cm $month^{-1}$	0	0	0.62	0.47	2.47	2.44	14.9	20.1	14.9	0	9.9	0	65.9 cm
Monthly wastewater generation rate (cm $month^{-1}$)[2]	6.00	6.00	6.00	6.00	6.00	6.00	6.00	6.00	6.00	6.00	6.00	6.00	72.0 cm
Wastewater storage requirement from previous month (month)	2	3	4	4.9	5.9	6.5	7.1	5.6	3.2	1.7	2.7	2.1	
Wastewater storage requirement change from current month (month)[3]	+1	+1	+0.9	+0.9	+0.6	+0.6	-1.5	-2.4	-1.5	+1	-0.6	+1	
Accumulative wastewater storage requirement at the end of month (month)[4]	3	4	4.9	5.9	6.5	7.1	5.6	3.2	1.7	2.7	2.1	3	

Notes: 1. Field observed monthly hydraulic disposal rate for month when recorded precipitation falls closest to 30-year average.
2. Defined by the Alabama Onsite Sewage Disposal Rules (ADPH, 2006) for clay soils receiving effluent from conventional onsite septic systems. (Depends on soil hydraulic properties and receiving wastewater quality, not quantity).
3. Determined by dividing the difference between 'Expected monthly maximum hydraulic wastewater disposal rate' and 'Monthly wastewater generation rate' by 6.00 cm $month^{-1}$.
4. Determined by adding the number of 'Wastewater storage from previous month' and 'Wastewater storage generated in current month of each column'.

exploited if used in conjunction with other wastewater treatment systems that function during periods of low soil hydraulic dosing. For example, this system may be used as a supplement to existing municipal or decentralized community wastewater treatment facilities having adequate land, labor, and machinery.

Based on observed data, a seven-month wastewater storage requirement may be required (Table 5.4). A worst case analysis indicates that the wastewater storage requirement increases each year due to a negative water balance, which means that more effluent is produced than can be safely dispersed in a year. Thus, if a minimum 2-month wastewater storage is specified, every expectation is that this volume will provide sufficient storage in only the driest years.

5.4.2 SYSTEM NUTRIENT MANAGEMENT

FIELD CROP NUTRIENT UPTAKE - YEAR TWO (JUNE 14, 2007 TO JUNE 24, 2008)

Nutrient uptake performance and field crop yield in treatments I and III are listed in Table 5.5 for each growing season in year two. Dry mass yield of the 1st cutting of sorghum and sudangrass in 2007 was 3.7 times higher in treatment I than in treatment III. The 2nd cutting of sorghum and sudangrass and the subsequent winter wheat and rye had similar crop yields for both treatments. These results may indicate a reduced crop yield benefit from synthetic wastewater application into fall and winter as wastewater hydraulic dosing rate decreased. Annual crop nutrient uptake in treatment I represented approximately 32% of applied nitrogen and 31% of applied phosphorus in year two. Approximately 86% of applied nitrogen in treatment III was represented by crop nutrient uptake in the same period, suggesting nitrogen application in treatment I may not have been over supplied compared to the crop growth requirement.

Table 5.5 Crop yield, nutrient application, and uptake of treatments I and III during year two, June 14, 2007 to June 24, 2008.

	Growing crops	Dry mass yield, kg ha^{-1}	Total N applied, kg ha^{-1}	Total N uptake, kg ha^{-1} (% of total N applied)	Total P applied, kg ha^{-1}	Total P uptake, kg ha^{-1} (% of total P applied)
Treatment I (synthetic wastewater application subplot)	Sorghum-sudan grass 1st cut (Jun 07- Aug 07)	5910	267	103 (38%)	52	11 (21%)
	Sorghum-sudan grass 2nd cut (Aug 07- Nov 07)	820	347	14 (4%)	31	2 (6%)
	Winter wheat and rye (Nov 07-Jun 08)	13200	186	138 (75%)	11	17 (154%)[1]
	Annual (Jun 07-Jun 08)	19900	800	256 (32%)	94	29 (31%)
Treatment III (agronomic control subplot)	Sorghum-sudan grass 1st cut (Jun 07- Aug 07)	1620	67	28 (42%)	0	11
	Sorghum-sudan grass 2nd cut (Aug 07- Nov 07)	774	67	10 (15%)	0	5
	Winter wheat and rye (Nov 07-Jun 08)	11700	67	136 (203%)[1]	0	107
	Annual (Jun 07-Jun 08)	14000	202	174 (86%)	0	123

1. Crop dry mass yields were calculated as the average of three field measurements.
2. Total N and total P applied were calculated based synthetic wastewater composition (Table 5.2) and recorded system daily flow rate (Figure 5.1).
3. Crop uptake of nitrogen or phosphorus was greater than fertilized, meaning soil provided additional nitrogen or phosphorus to meet crop growth requirements.

Higher system hydraulic dosing rates in summer and fall brought higher nutrient load to treatment I compared to the spring and winter. Although the nutrient contribution from the synthetic wastewater resulted in over three times the crop yield in treatment I versus treatment III during the warm season, excess nitrogen and phosphorus in soil water was the apparent result (Figures 5.10-5.12). The lack of yield difference between treatments I and III after the warm season seems to indicate that unscavenged nitrogen could potentially be lost from a drain field due to the high mobility of nitrogen in soil (Sparks, 2003). These observations indicate that soil moisture controlled wastewater disposal may temporarily provide nutrient loads higher than crop uptake needs.

Soil Water Nutrient Concentration - Year Two (August 2007 to June 2008)

Field suction lysimeter data from treatments I and II during year two are presented for NH_4^+-N (Figure 5.8), NO_3^--N (Figure 5.9), and total phosphorus (TP) (Figure 5.10) to demonstrate the impact of applied wastewater on soil water nutrient levels under a normal rainfall year. With the measured data of treatment II as a control, soluble NH_4^+-N (all less than 10 mg L^{-1}) and TP levels (all less than 9 mg L^{-1}) in soil water samples from treatment I were lower than that of the applied synthetic wastewater (approximately 80 mg L^{-1} of NH_4^+-N and 10 mg L^{-1} of TP), suggesting that nutrients with synthetic wastewater underwent chemical, physical, or biological transformation and/or uptake that removed them from the water phase. Since the negligible water percolation below 45 cm depth during the first half of 2008 was only by estimation and there was an obvious water percolation below 45 cm depth in June 2008, the possibility exists that water did percolate below 45 cm during the lysimeter monitoring period. Measured nutrients were generally at relatively higher levels during the warm season compared to the cold season, suggesting that the increased nutrients supplied from higher hydraulic dosing rates in the warm season may have extended deeper than the 45 cm depth during year two. Also, if nutrients were applied during year one when a significant amount of water percolation was estimated, the chances of nutrient deep percolation would be expected even higher than in year two.

Suction lysimeter measurements provide only a snapshot of the nutrient levels in soil solution during year two. Since sampled depth reached only 45 cm depth, definite conclusions regarding nutrient leaching status cannot be made. The cumulative effect from the synthetic wastewater application (treatment I) was further evaluated by comparing soil nutrient profiles of the four treatments (Figures 5.11-5.16).

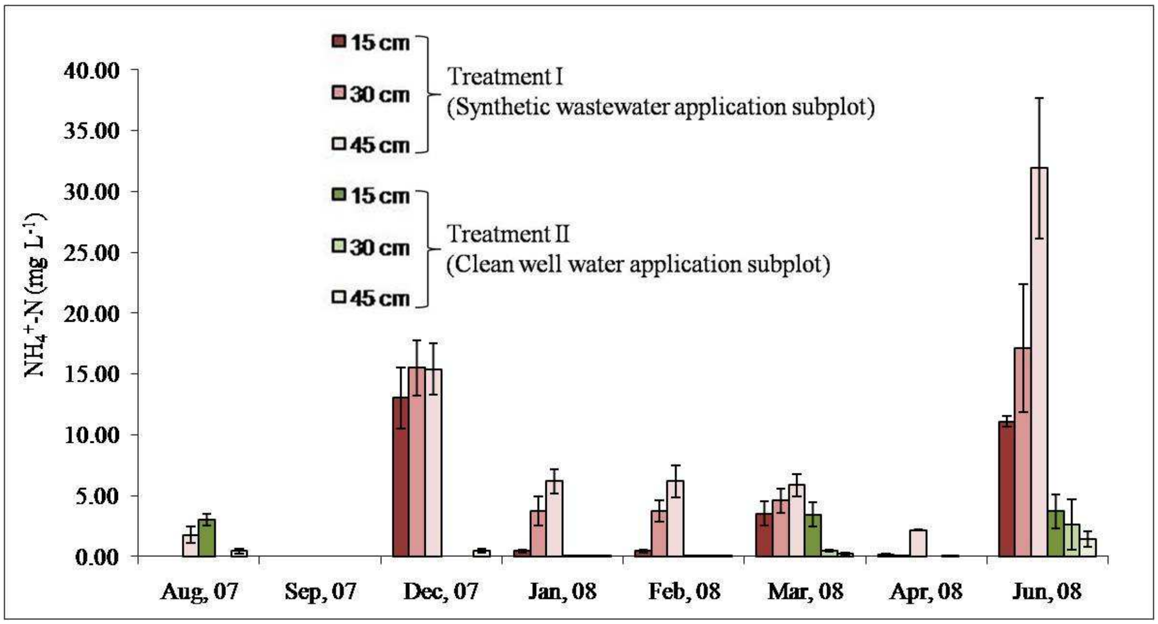

Figure 5.8 Soil water NH_4^+-N levels in treatments I and II during year two.

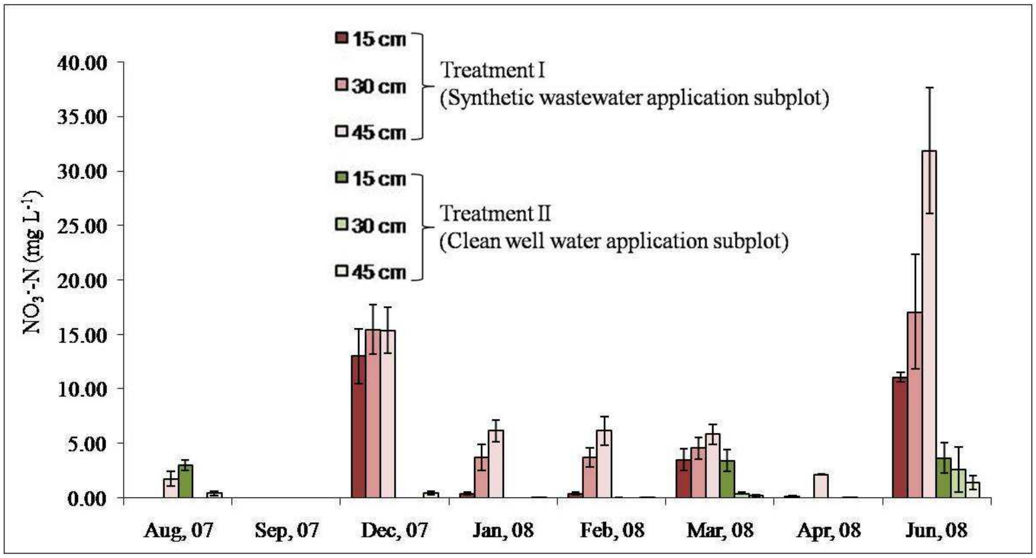

Figure 5.9 Soil water NO_3^--N levels in treatments I and II during year two.

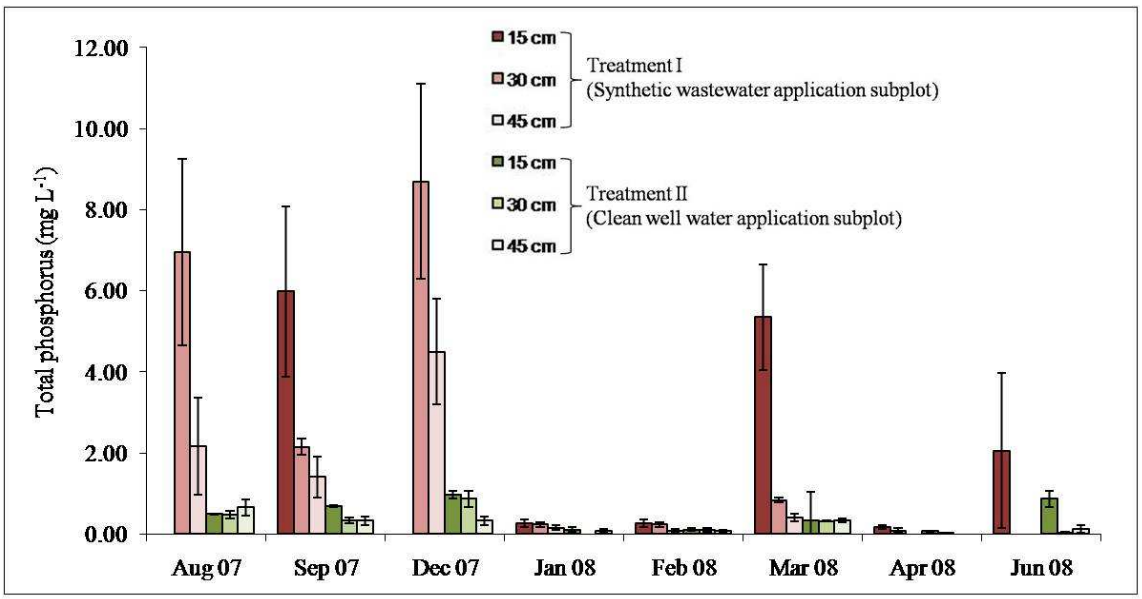

Figure 5.10 Soil water TP levels in treatments I and II during year two.

SOIL NUTRIENT PROFILE AT THE END OF SYNTHETIC WASTEWATER APPLICATION – YEAR TWO

Soil total nitrogen (TN) profiles (Figure 5.11) and soil crop available nitrogen profiles (Figure 5.12) for the four treatments after year two indicate gradually reduced nutrient concentrations with depth in all treatments over the sampled 100 cm soil depth, with no discernable differences between the treatments. Comparison of average soil cation exchange capacities (CEC) for the five soil horizons (Table 5.1) with crop available NH_4^+-N profiles (Figure 5.12) indicates that crop available NH_4^+-N in all treatments was approximately 1000 times lower than soil CEC. This observation, together with the measured soil pH = 8 of treatment I (Figure 5.13) indicating favorable conditions for NH_3 volatilization (Chang, 2006; Sigunga et al., 2002), suggests that soil solution chemistry reduced the potential for NH_4^+-N deep percolation.

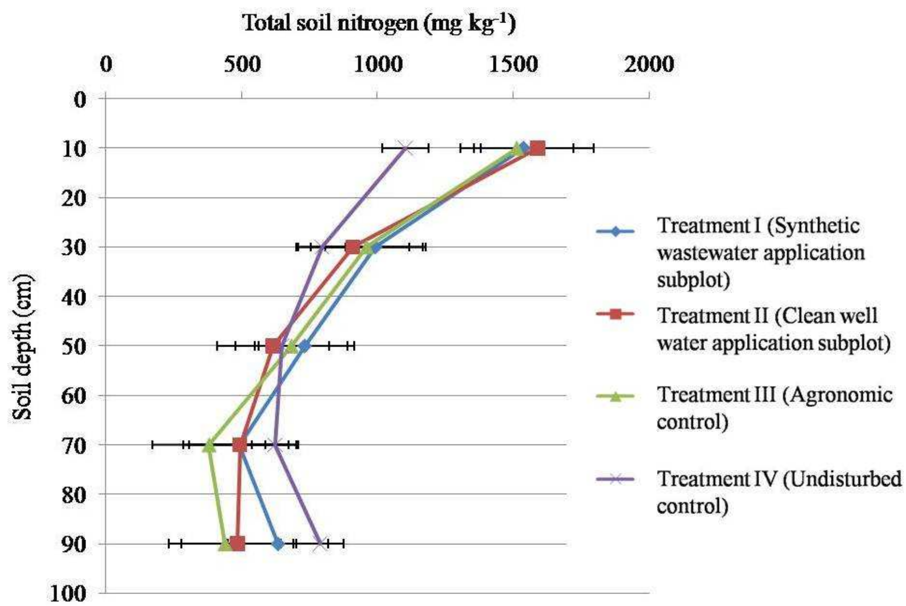

Figure 5.11 Year two total soil nitrogen profiles for field treatments I, II, III, and IV, June 24, 2008. (Data represent means of triplicate sub samples.)

The observed decreasing trend of crop available NO_3^--N with soil depth (Figure 5.12) indicates there was little likelihood of NO_3^--N leaching over the soil profile at the time the soil cores were sampled, as has been reported by other field studies. Pérez et al. (2003) reported that downward movement of a fertilizer NO_3^--N plume increased crop available NO_3^--N and water phase NO_3^--N levels along its pathway. Although the synthetic wastewater in the present study was applied consistently to treatment I when soil cores were taken and the undisturbed control plot was barren without any applied fertilizers, there is no apparent difference between the two treatments in terms of NO_3^--N leaching. In fact, it appears evident that NO_3^--N leaching did not occur in any treatment during the synthetic wastewater application period, or its influence on soil nitrogen profile in treatment I had already passed down to soils deeper thatn 100 cm as indicated by the slightly increased of soil total N and crop available NH_4-N. This is also an indication that soil profile sampling depth should be even deeper in future studies.

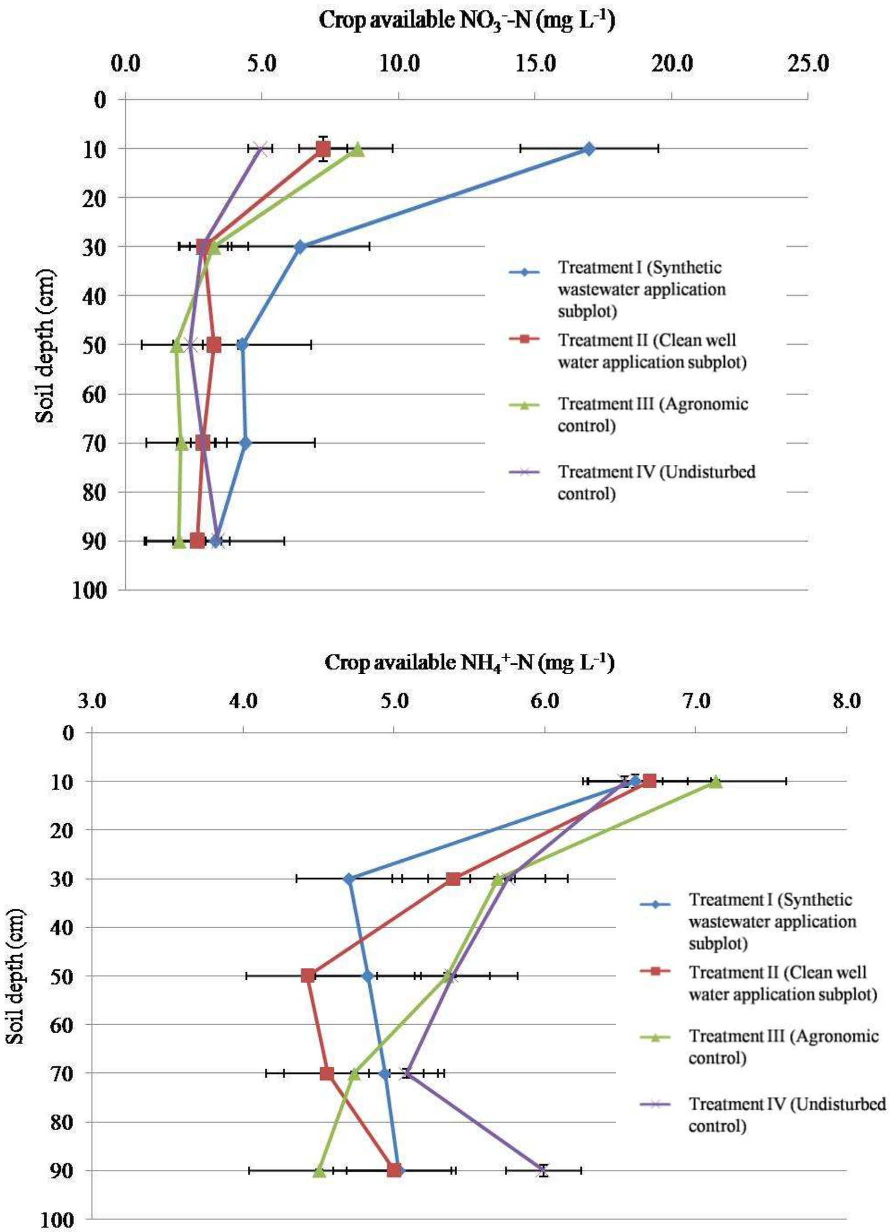

Figure 5.12 Year two soil crop available nitrogen (NH_3^+-N and NO_3^--N) profiles for field treatments I, II, III, and IV, June 24, 2008. (Data represent means of triplicate sub samples.)

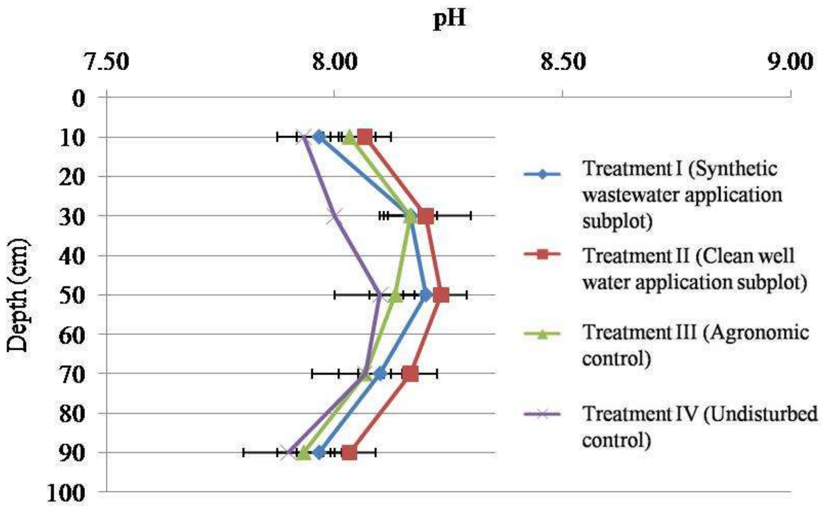

Figure 5.13 Year two soil pH profiles for field treatments I, II, III, and IV, June 24, 2008. (Data represent means of triplicate sub samples.)

Total nitrogen applied to treatment I during year two was 800 kg ha^{-1}, which represents approximately 12% of the total nitrogen contained in the soil (Table 5.6). This finding further indicates there is a low possibility of nitrogen leaching loss during the experimental period. However, as indicated previously, substantial water percolation losses during dry soil conditions may have occurred at the site due to preferential flows. Consequently, the possibility exists that preferential flows caused by soil cracking may have conducted applied synthetic wastewater into deeper (> 100 cm) soils without being detected by sampled soil cores. If a restrictive layer exists close to the 100 cm depth as suggested by soil moisture profiles (Figure 5.6), then nutrient accumulation might occurred at depth close to 100 cm, which however was not observed in this study.

Table 5.6 Year two partial field nutrient balance of the treatment I drain field for the synthetic wastewater application experiment from June 14, 2007 to June 24, 2008.

	Annual contribution from synthetic wastewater, kg ha^{-1}	Annual crop uptake, kg ha^{-1}	Total content in the upper 100 cm depth at the end of soil core sampling, kg ha^{-1}
Nitrogen	800	256	6783
Phosphorus	94	29	2000

1. Reported values are the averages of three field measurements.
2. Total N, P contents in the upper 100 cm depth at the end of soil core sampling were calculated with field measured bulk density and soil total N, P profiles.

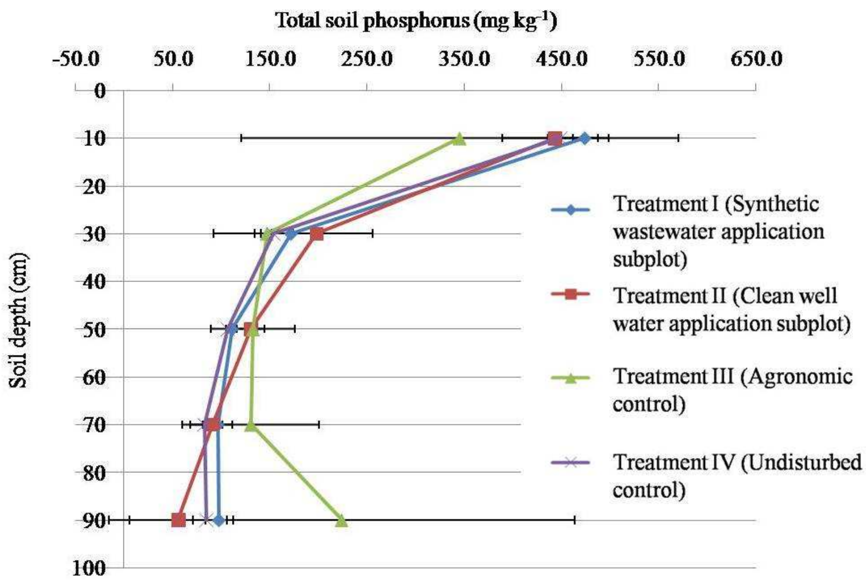

Figure 5.14 Year two total soil phosphorus profiles for field treatments I, II, III, and IV, June 24, 2008. (Data represent means of triplicate sub samples.)

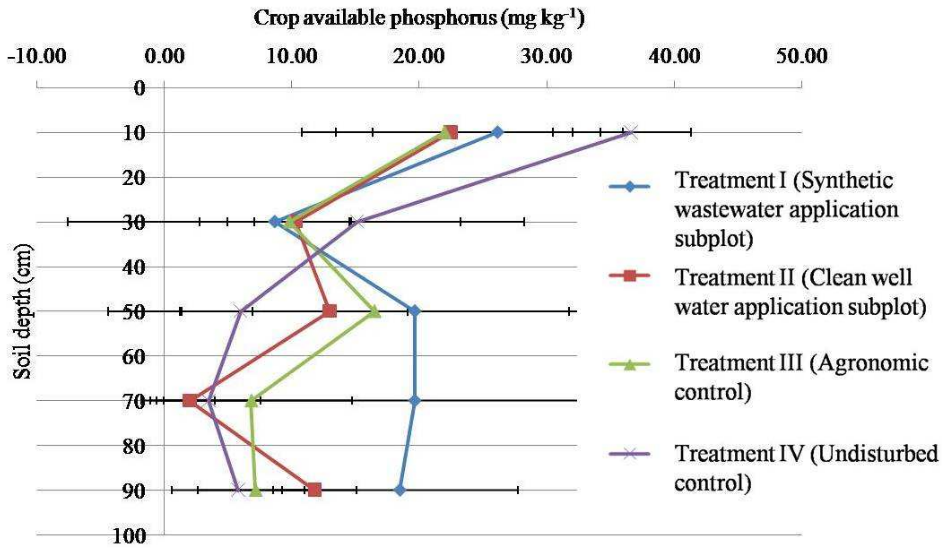

Figure 5.15 Year two soil crop available phosphorus profiles for field treatments I, II, III, and IV, June 24, 2008. (Data represent means of triplicate sub samples.)

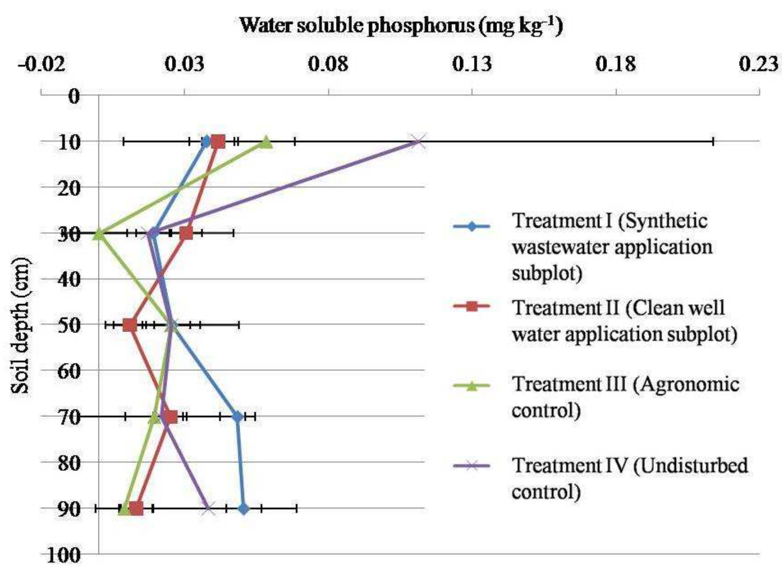

Figure 5.16 Year two soil water soluble phosphorus profiles for field treatments I, II, III, and IV, June 24, 2008. (Data represent means of triplicate sub samples.)

Year two soil TP (Figure 5.14), crop available P (Figure 5.15), and water soluble P (Figure 5.16) profiles of the four treatments indicate a decreasing trend of approximately 80% over the 100 cm soil profile depth, with no discernable difference between treatments. The measured soil phosphorus adsorption capacity of the experimental site was 16.67 g P kg^{-1} soil (Table 5.1), over 400 times higher than the measured crop available P of treatment I. Furthermore, the total amount of phosphorus applied to treatment I during the one-year wastewater application period was 94 kg ha^{-1}, approximately 4.7% of the total phosphorus contained in the upper 100 cm depth of soil (2000 kg P ha^{-1}) (Table 5.6). Thus, it appears that there is a low likelihood of field soil P accumulation after one year of synthetic wastewater application. However, it should be noted that preferential flows may have carried wastewater into soils deeper than 100 cm depth with less resulting influence on the top 100 cm soils. The year two study was carried out under a relatively normal rainfall year when water percolation below 45 cm was estimated as negligible. During year one a significant amount water percolation loss was estimated under a historic drought. Consequently, if nutrients had been applied in year one, the soil profile may have indicated signs of nutrient accumulation or deep percolation.

Previous researchers reported that P leaching is negligible in drain fields due to numerous P immobilization mechanisms such as chemical precipitation and chemical/physical adsorption (Sawhney and Hill, 1975; Venhuizen, 1995). Field results from this study appear to conform to literature regarding negligible concern for short-term P leaching in spite of the recognized potential for long term P accumulation due to annual P input. Findings are by no means conclusive since the nutrient application study was carried out for only one years. With considerable water percolation loss below 45 cm depth indicated, the possibility of higher P levels in deep soils in treatment I exists, especially if wastewater dosing were continued. Due to the limited number of soil core sampling locations (3 locations per treatment), it is possible that applied wastewater may have carried P into deeper soils through preferential flow.

5.4.3 System Application Benefits and Limitations

Based on observed data from years one and two of the field study, the experimental soil moisture controlled wastewater disposal system as designed and installed provides several advantages. The system first and foremost effectively withholds wastewater disposal during wet field conditions to mitigate health threats associated with drain field hydraulic overloading that leads to surface ponding. Second, wastewater may be stored or recycled for use during favorable field conditions, similar to agricultural or other liquid waste storage facilities. Third, during certain warm seasons, hydraulic disposal rates may be temporarily increased to levels higher than the hydraulic loading rate permitted in current OWTS drain field design (ADPH, 2006; AWOTC, 2005; EPA, 2003).

However, several operational deficiencies present solid road blocks for a successful application of the technology. First, there are extended zero and near zero hydraulic dosing rate periods (minimum 2-month) during wet soil conditions that result in an excessive (minimum 2-month) wastewater storage requirement. Over the two-year study period, on average only 4 months out of 12 had observed system hydraulic dosing rates higher than wastewater inflows. The large wastewater storage requirement prevents this experimental system from being acceptable as a stand-alone application. However, opportunities exist for use of this system supplement to other wastewater treatment technologies that can function when field conditions do not favor direct soil dispersal. Second, as indicated from the two-year monthly water balance (Figure 5.3), it is apparent that the experimental system cannot overcome water percolation losses (over 30% of dosed water) during severe drought conditions due to seasonal changes in shrink-swell clays. The system as designed cannot guarantee effective control over this type of water percolation loss in smectitic soils. Thirdly, because this experimental system is completely dependent upon soil moisture to control the dispersal of wastewater, wastewater applications during certain seasons may exceed crop uptake (approximately 70% of annual applied N and P were not accounted for by crop uptake), making the drain field at least temporarily a potential pollution source for groundwater. Fourth, but not the least, routine and frequent field and crop maintenance (three field crop harvesting and plantings per year under the seasonal rotation used in this study) is required of this system to ensure adequate operation. The combined requirements of land, equipment, and labor exclude this experimental system as a viable option for land owners with limited resources. Even if the above challenges were adequately resolved, more conclusive results would be required to document economic and environmental feasibility through adequately replicated field trials.

5.5 Conclusions

Over a two-year field study, an experimental wastewater SDI system that incorporates real-time soil moisture control and a seasonal cropping system was evaluated for hydraulic and nutrient management in the Alabama Black Belt clay soil. Soil-moisture controlled hydraulic disposal rates in the drain field varied between 1.18 cm day^{-1} in April 2007 to a nearly two-month zero disposal period (0.0 cm day^{-1}) during the preceding and succeeding winter seasons. Demonstrated advantages of the water management strategy of this experimental system include: 1) avoidance of soil moisture conditions above field capacity in the absence of consistent rainfall events; 2) seasonally increased wastewater disposal rates under favorable dry field conditions; 3) the experimental system dependence on soil moisture control for wastewater disposal may temporarily create a field nutrient imbalance with respect to crop uptake; and 4) routine system and crop maintenance requirements are not practical for land owners with limited resources.

There are several inherent limitations of the system. First, estimated water balance suggests that over 30% of dosed water percolated below 45 cm during the drought of 2007. Although water percolation is necessary for effluent treatment, this result indicates the inadequacy of the experimental system to limit preferential flow in smectitic soils during dry soil conditions. Second, a minimum 2-month wastewater storage requirement was found necessary, indicating that the experimental system is not suitable as a stand-alone wastewater dispersal and treatment method. Nevertheless, the experimental system as designed can be used in conjunction with other wastewater treatment methods that can function during wet periods when this experimental system provides zero or near zero hydraulic dosing.

Despite the potential environmental risks associated with deep percolation, field nutrient observations indicate that the experimental system did not result in discernable drain field nitrogen and phosphorus accumulation or percolation beyond 100 cm. These findings are by no means conclusive since preferential flows caused by clay soil shrinking and crack development may have carried nutrients beyond 100 cm without being detected in treatment I, where the possible presence of a restrictive layer at the 100 cm depth in treatments II-IV does not exist.

This study concludes that soil moisture controlled SDI dispersal of wastewater in native clay soils of the Alabama Black Belt is not suitable as a stand-alone disposal and treatment method. Major reasons cited include an excessive (2-month minimum) wastewater storage requirement and environmental questions related to groundwater that are left unresolved by this study. Nevertheless, the system as designed and installed has potential as a supplement to existing municipal or decentralized community wastewater treatment facilities with adequate land, machinery, and labor. In order to quantifying indicated system limitations, the system should be further studied in an attempt to limit soil cracking during dry field conditions and adequately control soil water balance. Possible improvements to this end include reduced emitter and drip line spacing and a more uniform distribution of soil moisture sensors within the field. In addition, the long-term fate and transport of nutrients (NH_4^+-N, NO_3^--N, and TP) in percolated water requires further field study utilizing a more intensive spatial and temporal field sampling. Replicated experiments on a variety of similar sites are recommended to provide robust results regarding the environmental and economic feasibility of soil moisture controlled SDI wastewater dispersal in the Alabama Black Belt.

CHAPTER SIX

CONCLUSIONS AND SUGGESTIONS FOR FUTURE RESEARCH

6.1 CURRENT OWTS STATUS IN THE ALABAMA BLACK BELT AREA

Demographic data from the US Census indicates that 1) OWTS are widely used in the Alabama Black Belt; 2) a significant number of existing OWTS in the study area have been operating for more than 20 years; 3) system size is generally maintained below 2 person unit^{-1} except around major urban centers; and 4) 12% of census block groups have an estimated OWTS density higher than the EPA regulated threshold for negative environmental impact (15 unit km^{-2}) and are generally found clustered around city fringes.

A new OWTS soil suitability rating system (OWTS-SSRS) developed based on current Alabama OWTS regulations was applied to the Alabama Black Belt area using soil information extracted from USDA-NRCS digital soil survey data (SSURGO). The new OWTS-SSRS was compared with an existing soil limitation rating system developed by NRCS (NRCS-SLRS). The new OWTS-SSRS rated approximately 52% of the Alabama Black Belt area as unsuitable for OWTS, while the more conservative NRCS-SLRS rated 89% of the area as limiting for OWTS.

Based on the results of this study, two strategies to limit the potential environmental and health risk from OWTS failure in the Alabama Black Belt area are suggested. For city fringe communities, a proactive response is to extend municipal sewer service to high risk clay soil areas in advance of widespread OWTS failure. For isolated rural households outside the range of municipal sewer service, retrofits or replacement of aged OWTS with more advanced dispersal systems is recommended, such as aeration treatment units, packed-bed media filters, mounds, or subsurface drip irrigation.. The need for alternative wastewater treatment and dispersal systems for clayey soil regions within the Alabama Black Belt area is clear. An appropriate alterntive for wastewater treatment and dispersal on clayey soils would benefit regional decision makers in targeted efforts to retrofit or replace failure systems in the Black Land Prairie soils area.

6.2 System Design

This research developed and tested a soil moisture controlled wastewater SDI system for use in a clay adsorption field. The intention of the research was to evaluate an alternative wastewater dispersal and treatment system for the study area. The experimental system was installed on a clay soil site of 500 m^2 and evaluated with respect to water and nutrient management over a two-year period from September 2006 to June 2008. The experimental system included an SDI wastewater dosing system controlled by volumetric soil moisture. A seasonal cropping rotation of sorghum-sudangrass (*Sorghum bicolor*) and winter wheat (*Triticum aestivum*) with rye (*Secale cereale*) mix was integrated into the wastewater dispersal and treatment system to enhance annual water and nutrient crop uptake. Wastewater was only dispersed in the drain field when field moisture content at monitored locations dropped below field capacity (0.40 m^3 m^{-3}), to limit health threats associated with overloaded drain fields.

6. 3 FIELD EVALUATION OF SYSTEM WATER AND NUTRIENT MANAGEMENT

Observed hydraulic dosing rates in the drain field varied from a high of 1.18 cm day^{-1} during drought conditions to a low of 0.0 cm day^{-1} during wet winter months. Estimated monthly water balance indicated that over 30% of dosed water percolated below 45 cm depth during the historic dry summer of 2007, presumably due to soil cracking. Although water percolation below 45 cm was not necessarily an environment concern at the site, preferential flows caused by soil cracking during dry soil conditions was unexpected. Over the two-year study, only 4 months out of 12 had an observed water balance in favor of soil adsorption. A minimum a two-month onsite wastewater storage requirement is estimated for the experimental system due to zero or low hydraulic disposal periods during typical wet winter months.

Observed seasonal and annual field nutrient loading and crop nutrient uptake in the drain field indicates the following nutrient imbalance. Nutrient supply from wastewater was at surplus levels for warm season crops, while cool season crops scavenged nutrients due to zero or almost zero wastewater disposal rates. Although soil nutrient profiles provided no direct evidence of drain field nitrogen or phosphorus accumulation or percolation below 100 cm depth in year two of the study, applied wastewater may have been transported into deeper soils (> 100 cm) by preferential flows caused by soil cracking if no restrictive layer existed at close to 100 cm to intercept percolated water and nutrients.

6.4 System Benefits and Limitations

Based on the two-year field experiment, the obvious advantages of the soil moisture controlled wastewater dispersal strategy are: 1) avoidance of soil moisture conditions above field capacity in the absence of consistent wet weather; and 2) seasonally increased wastewater dosing rates during favorable (dry) field conditions. In spite of these benefits, the experimental system cannot be recommended for direct field application due to the following limitations: 1) the extended zero or near zero hydraulic dosing rate (minimum 2-month) observed during wet winter conditions creates an impractical two-month minimum wastewater storage requirement; 2) the combined dosing control and SDI system resulted in substantial water percolation loss (over 30% of dosed water) during severe drought apparently due to seasonal soil shrinking; 3) the system dependence on soil moisture control for wastewater disposal created a nutrient imbalance by annually supplying approximately 70% more N and P than measured crop harvest); and 4) routine SDI and crop operation and maintenance requirements (three times per year of field crop harvest and planting) are not practical for land owners with limited resources.

Although the geologic status of the Alabama Black Belt area indicates that deep percolation not a serious environmental issue, effective control of soil water balance for environmentally safe dosing is one of the main goals still to be realized by this experimental system.

Based on the observed benefits and limitations of the experimental system, this study concludes that soil moisture controlled SDI disposal of wastewater in native clay soil of the Alabama Black Belt is not suitable as a stand-alone disposal and treatment method. The main limitation of the experimental system is the excessive (two-month minimum) wastewater storage requirement. Nevertheless, the system has potential as a supplement to certain existing municipal or decentralized community wastewater treatment facilities with adequate land, equipment, and labor resources. Due to the limited time frame of the field experiment and inadequate field sampling, extended field experiment is suggested to obtain more conclusive results than the current study in order to better address the applicability of the proposed soil moisture controlled SDI wastewater dispersal in the Alabama Black Belt area and other similar soil regions.

6.5 Suggestions for Future Research

The purpose of this study is to obtain quantifiable results regarding system water and nutrient management in a high clay drain field. System feasibility components including further quantification of wastewater storage requirement, surface ponding, and nutrient leaching are needed. Based on results of the present study, a long-term field experiment is recommended to include; 1) adequately replicated (three minimum) field treatments, similar to the current study: soil moisture controlled wastewater application, soil moisture controlled clean water application, agronomic control, and undisturbed control, 2) an improved field sampling schedule at deeper locations and higher frequencies. Since no significant percolation below the soil moisture controlling zone was observed during year two of the study (a normal rainfall year), SDI layout can be continued per the current study (61 cm of lateral and emitter spacing). Modification of SDI lateral and emitter spacing can be based on experimental results regarding system effectiveness in controlling soil water balance. Results will quantify system water and nutrient management in terms of hydraulic seasonal dosing pattern, monthly water percolation loss, wastewater storage requirement, drain field water and nutrient balance, monthly soil water nutrient levels, monthly nutrient leaching loss, and seasonal drain field nutrient balance. Conclusions can be made regarding system applicability based on observed season water and nutrient management cycles. Conclusive results from an improved long-term experiment are expected to provide a more thorough understanding of soil water balance control of wastewater dispersal on clayey soils with respect to the environmental challenges inherent when attempting to control leaching control while preventing surface ponding.

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APPENDIX I

SDI DRIP TUBE EMITTER HYDRAULIC UNIFORMITY TEST

System hydraulic performance was evaluated by drip tube emitter hydraulic uniformity and system flush velocity. To test emitter hydraulic uniformity, twenty emitter flow rates were measured at 240 kPa three times to determine average discharge per emitter. A coefficient of variation for the 20 emitters was calculated using Eq. I.1 (ASAE EP 405.1, 2003), with results compared to standard values.

$$C_v = \frac{\left(q_1^2 + q_2^2 + \ldots + q_n^2 - n\bar{q}^2\right)^{1/2}}{\bar{q}(n-1)^{1/2}} \qquad \text{(I.1)}$$

Where:

Cv = emitter coefficient of variation;

q1, q2, …, qn = discharge of emission devices tested (l/h);

$\bar{q}$ = average discharge of all emission devices tested (l/h);

n = number of emitters (n=20).

Measured flow rates of each emitter are plotted in Figure I.1 against the manufacturer's specification of 2 L h^{-1}. The calculated coefficient of variation (C_v) is 0.067, indicating that hydraulic uniformity performance is average (Table I.1) (ASAE EP 405.1, 2003). One of the SDI advantages in agricultural industry is that water can be distributed rather uniform in the field than surface overflow (Beggs et al., 2004). For OWTS, a uniform wastewater distribution in drain field can postpone local hydraulic and nutrient overloading in the field that causes most OWTS failures (US EPA, 2002). This average rated emitter hydraulic performance is expected for future field experiment.

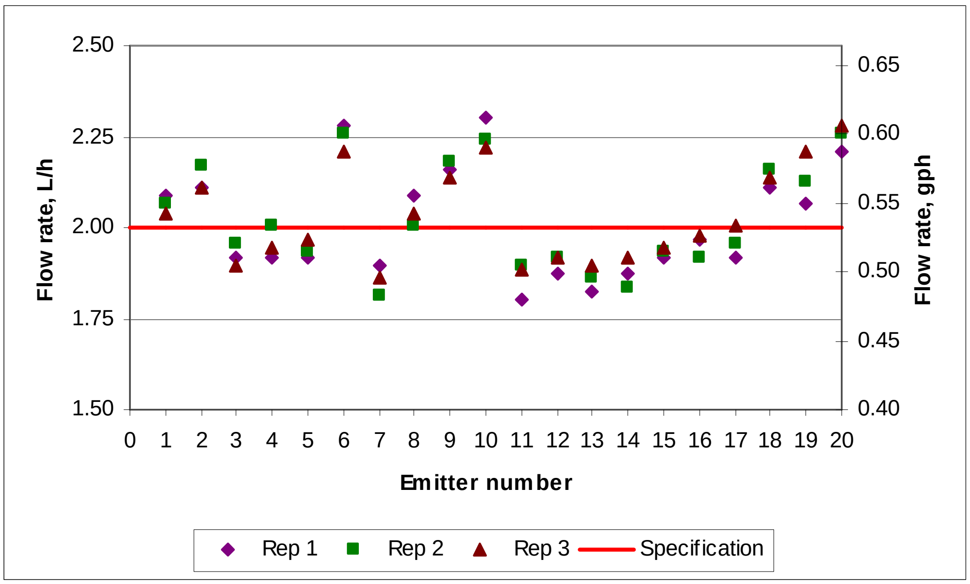

Figure I.1 Measured flow rates of each emitter. (Adapted from Ducote, 2006)

Table I.1 Recommended Classification of Manufacturer's Coefficient of Variation.

C_v range	Emitter Performance
< 0.05	Good
0.05 to 0.07	Average
0.07 to 0.11	Marginal
0.11 to 0.15	Poor
> 0.15	Unacceptable

Source: American Society of Agricultural and Biological Engineers (ASAE EP 405.1, 2003).

APPENDIX II

SITE CHARACTERIZATION

The site selected for the field study is in Marion Junction, Alabama, at the Alabama Black Belt Research and Extension Center (ABBREC), located approximately 10 miles west of Selma, Alabama. The unique value of the ABBREC is the proximity to the people, plants, weather, and soils of the region.

The test site was selected by field investigation after identifying five fields reasonably accessible to power, water, and transportation. Figure II.1 shows the boundaries and major soil series of the 450-hectare ABBREC and five potential fields for the study. A Houston clay soil at Field 21 was ultimately selected because of the fewest impediments to year-round SDI wastewater dosing, while providing low permeability and high shrink-swell features common to the region. Field 21 also has a low slope and relatively high fertility potential. A truck-mounted Giddings® probe and sleeve was used to retrieve core samples from Field 21 for soil profile characterization.

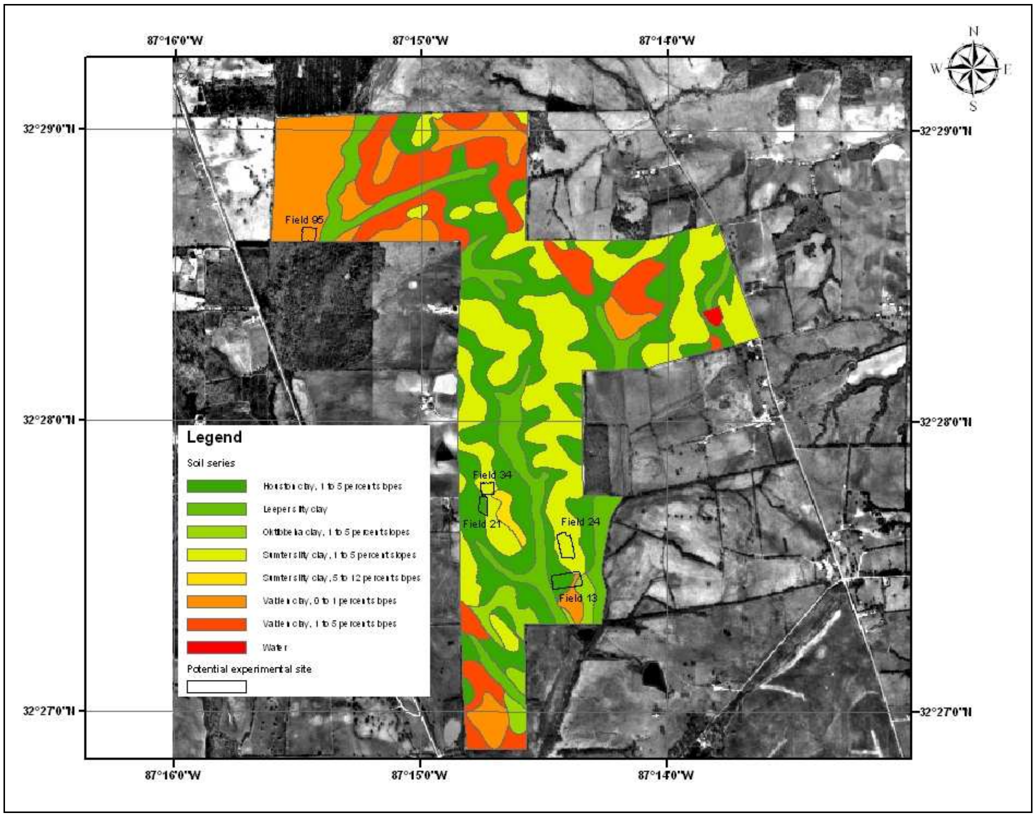

Figure II.1 General soil series map, Black Belt Research and Extension Center, Marion Junction, Alabama, showing location of study site.

Field and laboratory soil characterization for the Houston soil site at Field 21 was completed by the Auburn University Pedology Laboratory (Tables II.1-II.4). Five soil horizons were identified to 1.52 m and corresponding to Figure II.1. Table II.1 indicates that dark clay was prominent at the surface to approximately 42 cm depth, with evidence of redox at 88 cm indicating significant periods of saturated or anaerobic conditions during most years. Table II.2 indicates a pH favorable for agricultural production and a base saturation free of exchangeable acidity (H^{+1} and Al^{+3}).

Table II.1 Detailed soil description to 1.52m depth at Field 21, Black Belt Research and Extension Center, Marion Junction, Alabama. Side slope, 4%.

Horizons Depth (cm)	Color	Detailed description
Ap1 (0-23)	very dark gray (2.5Y 3/1)	clay
Ap2 (23-42)	dark grayish brown (2.5Y 4/2)	clay
BA (42-63)	olive brown (2.5Y 4/3)	clay; few fine black manganese oxide concretions; common medium and coarse calcium carbonate concretions and soft accumulations
Bkss1 (63-88)	olive brown (2.5Y 4/3)	clay; large wedge-shaped aggregates that are bordered by intersecting slickensides; very plastic, sticky; few fine black manganese oxide concretions; common medium and coarse calcium carbonate concretions and soft accumulations; calcareous
Bkss2 (88-152+)	light olive brown (2.5Y 5/6); common light brownish gray (10YR 6/2) redox depletions; common dark yellowish brown (10YR 4/6) redox concentrations	clay; large wedge-shaped aggregates that are bordered by intersecting slickensides; very plastic, sticky; few fine black manganese oxide concretions; common medium and coarse calcium carbonate concretions and soft accumulations; calcareous

Described: by P.G. Martin and J.N. Shaw (11/17/05).

Table II.2 Chemical properties and organic matter at Field 21, Houston soil field site, Black Belt Research and Extension Station, Marion Junction, Alabama.

Horizon	Lower depth	Base saturation	pH (1:1)	
	cm	NH_4OAc	H_2O	0.1N $CaCl_2$
Ap1	23	100.00	7.34	7.17
Ap2	42	100.00	7.47	7.23
BA	63	100.00	7.46	7.27
Bkss1	88	100.00	7.38	7.25
Bkss2	152	100.00	7.33	7.30

Source: Auburn University Pedology Laboratory.

Table II.3 indicates relatively high CEC, with clay particles predominantly of the 2:1, shrink-swell type widely associated with the Alabama Black Belt. Table II.4

confirms that the majority of the particle size distribution is in clay fraction, with increasing clay percentages with depth up to 71% at 152 cm.

Table II.3 Chemical characteristics at Field 21, Houston soil field site, Black Belt Research and Extension Station, Marion Junction, Alabama.

Horizon	Lower Depth	Exchangeable Cations					Cation Exchange Capacity			
		Ca	Mg	K	Al	Na	CEC-7	ECEC	ECEC	CEC-7
	cm	meq/100 g soil							meq/100 g clay	
Ap1	23	na	0.12	0.15	0.12	0.03	38.84	39.37	72.89	73.88
Ap2	42	na	0.04	0.12	0.08	0.03	34.15	38.42	63.60	71.55
BA	63	na	0.04	0.11	0.08	0.04	34.42	37.66	60.98	66.72
Bkss1	88	na	0.07	0.16	0.04	0.06	35.26	36.48	58.21	60.23
Bkss2	152	na	0.18	0.18	0.06	0.08	40.62	42.85	57.14	60.27

Source: Auburn University Pedology Laboratory

Table II.4 Particle size distribution at Field 21, Houston soil field site, Black Belt Research and Extension Station, Marion Junction, Alabama.

Horizon	Lower Depth	Particle Size Distribution			Sand Size Distribution				
		Sand	Silt	Clay	2-1	1-0.5	0.5-0.25	0.25-0.1	0.1-0.05
	cm	%							
Ap1	23	7.09	39.63	53.28	0.64	0.64	1.50	1.93	2.36
Ap2	42	8.27	38.04	53.7	1.56	1.34	1.56	1.79	2.01
BA	63	10.17	33.38	56.45	3.10	1.99	1.55	1.77	1.77
Bkss1	88	3.50	35.93	60.57	0.22	0.66	0.66	0.87	1.09
Bkss2	152	3.10	25.80	71.10	1.03	0.52	0.26	0.26	1.03

Source: Auburn University Pedology Laboratory

APPENDIX III

SIZING SDI DRAIN FIELD

The most basic design calculation needed to size any septic tank drain field, whether a conventional system or an engineered system, is the required disposal area. Table III.1, below, presents design data used to determine the field area for the experimental wastewater SDI dosing system. Alabama Department of Health (2006) regulations provide allowable hydraulic loading (dosing) rates for soils with different percolation rates. Percolation rate in the Houston clay soil is known to be much greater than 120 minutes per 25.4 cm, so the far right column in Table III.1 (bold) is used exclusively for the design of Field 21. What is needed to determine drain field area is an estimate of effluent volume generated per day.

Table III.2 provides flow and organic loading rates for residential generators in Alabama, indicating the general requirement to use 1136 liters per day as a flow minimum. The experimental disposal design completed for Field 21 uses a lesser quantity of 1022 liters per day because of the recognized allowance for lower generation rates when designing for decentralized systems typical of small subdivisions or other rural wastewater cooperators (Alabama Department of Public Health, June 14, 2005, personal communication).

Table III.1 Design data used to size drain field based on a household loading rate of 270 gpd. Adapted from manufacturer recommendations and State of Alabama Onsite Sewage Treatment and Disposal regulations (effective November 23, 2006).

Soil type	Group IV (Sandy Clay, Silty Clay, Clay)							
Percolation rate (min 25.4 mm^{-1})	90	95	100	105	110	115	120	**>120**
Drip field or hydraulic loading rate* lpd m^{-2} (gpd $sq.ft.^{-1}$)	4.075 (0.1)	3.056 (0.075)	2.037 (0.05)	2.037 (0.05)	2.037 (0.05)	2.037 (0.05)	2.037 (0.05)	**2.037 (0.05)**
Quantity of effluent to be disposed per day, liters (gallons)	1022 (270)	1022 (270)	1022 (270)	1022 (270)	1022 (270)	1022 (270)	1022 (270)	**1022 (270)**
Determine total area required m^2 (ft^2)	250.8 (2700)	334.5 (3600)	502 (5400)	502 (5400)	502 (5400)	502 (5400)	502 (5400)	**502 (5400)**
Spacing between wasteflow lateral lines, m (ft)	0.61 (2)	0.61 (2)	0.61 (2)	0.61 (2)	0.61 (2)	0.61 (2)	0.61 (2)	**0.61 (2)**
Spacing between wasteflow lateral emitters, m (ft)	0.61 (2)	0.61 (2)	0.61 (2)	0.61 (2)	0.61 (2)	0.61 (2)	0.61 (2)	**0.61 (2)**
Total length of lateral lines m (ft)	411.5 (1350)	548.6 (1800)	823 (2700)	823 (2700)	823 (2700)	823 (2700)	823 (2700)	**823 (2700)**
Total number of emitters	2700	3600	5400	5400	5400	5400	5400	**5400**

* Values in this row are set by Alabama state regulation (ADPH, 2006). Group I, II, and III soils having percolation rates less than 90 min 25.4 mm^{-1} have hydraulic dosing rates ranging from 8.149 to 18.34 lpd m^{-2} (0.2 to 0.45 gpd $sq.ft.^{-1}$) and corresponding area requirements ranging from 125.4 m2 (1350 ft^2) down to as low as 55.74 m^2 (600 ft^2).

Table III.2 Flow and organic loading, State of Alabama Onsite Sewage Treatment and Disposal regulations (effective November 23, 2006).

Generator	Design Unit	Design BOD/TSS day^{-1}	Design Flow day^{-1}
Dwelling (8 bedrooms or fewer)	per bedroom	0.181 kg	568 liters
9 or more bedrooms to a single system	per person	0.091 kg	284 liters

Notes:
1. Organic loadings are prior to septic tank. It may be assumed that the tank will remove a maximum of 40% of the BOD & TSS load of sewage and 30% of high-strength sewage. This is an assumed loading rate for field sizing and should not necessarily be used for treatment design.
2. Estimated flows for residential systems assume a maximum occupancy of two persons per bedroom for systems handling fewer than 9 bedrooms. Large-Flow systems require engineer design, and documentation of occupant loading.

Using Table III.1, a total area of 502 m^2 (5400 ft^2) of drainage or disposal soil area is required, as follows; effluent generation/allowable hydraulic loading rate (based on percolation rate of the site)

(1022 liters per day) / (2.037 lpd/m^2) = <u>502 m^2 disposal area required</u>.

Based on standard practice of 0.61 m (2 ft) spacing between drip lines and 0.61 m (2 ft) spacing between emitters, a total of 823 m (2700 ft) of drip line is required, calculated as follows;

502 m^2 / 0.61 m = <u>823 m total length of drip line required</u>.

APPENDIX IV

FIELD CAPACITY MEASUREMENT

Soil cores (Ø: 5.5 cm, H: 6 cm) were taken from nine (9) different locations at a 20 cm depth within the SDI field (including treatment I (synthetic wastewater application) and treatment II (clean well water application)) on June 24, 2008 (Figure I.1). Soil cores were lab approximated for its field capacity at 1/3 bar air pressure. The measured volumetric moisture contents of the soil cores ranged from 37.19% to 44.39%, averaged at 40.60% with a standard deviation of 1.96% (Table I.1). The measured values together with their field sampling locations were imported into ArcMap 9.2 (ESRI, CA) and interpolated over the entire SDI site using inverse distance weight (IDW) method (Figure I.2).

Table IV.1. Soil volumetric moisture contents of the soil cores at 1/3 bar air pressure.

Soil moisture measurements, %, v/v									Average	STD
#1	#2	#3	#4	#5	#6	#7	#8	#9		
41.16	40.46	38.80	40.42	40.90	44.39	37.19	40.51	41.55	40.60	1.96

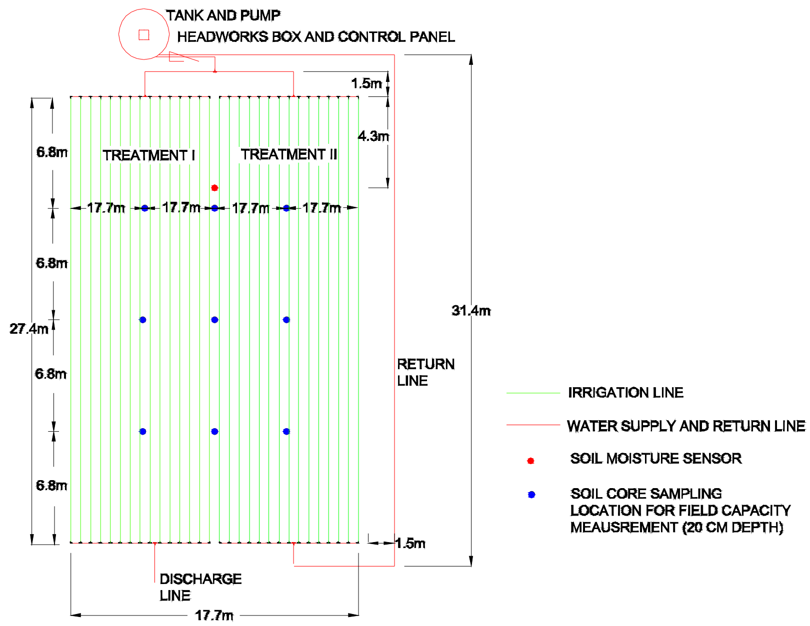

Figure IV.1 Field soil core sampling locations within the SDI site on June 24, 2008.

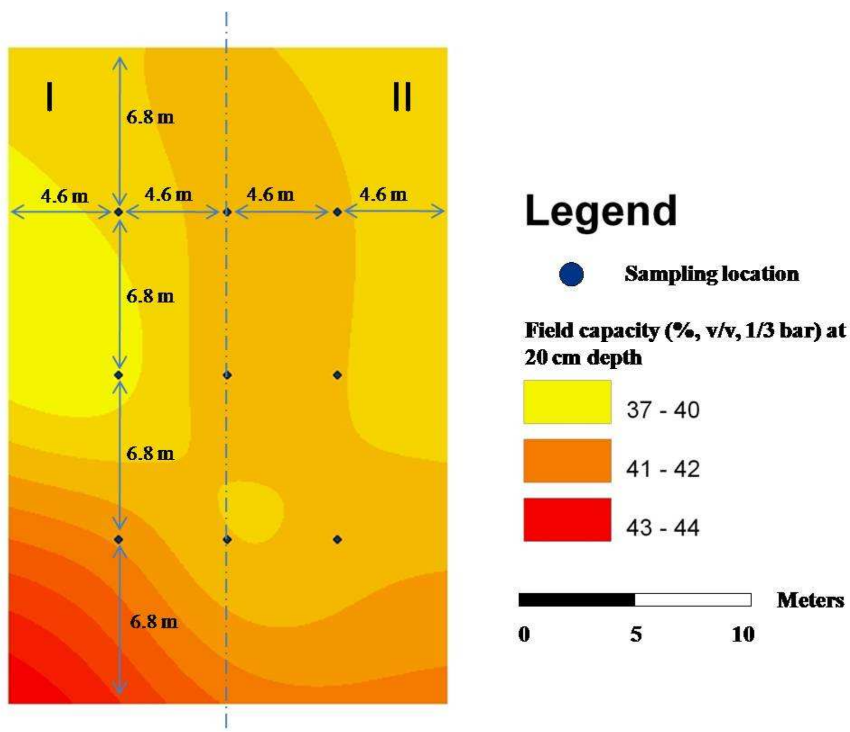

Figure IV.2 Interpolated site map based on the measured soil field capacities (1/3 bar) at 20 cm depth. (Sampled on June 24, 2008)

APPENDIX V

SOIL PHOSPHORUS ISOTHERM MEASUREMENT

The soil phosphorus isotherm of the experimental site was obtained by using composite soil samples with the method by Sevis-David (2002). The composite soil samples were prepared by mixing equal weight of subsamples from the soil cores of the four field treatments (I, II, III, and IV) (Figure V.1). Lab obtained isotherm data points are presented in Figure V.2. The data points of each soil depth were first fitted with Langmuir isotherm to determine maximum soil phosphorus adsorption capacity. Then, the adsorption coefficient, Kd, for HYDRUS 2D modeling was obtained by linear regression on the data points where water phase phosphorus concentration were lower than 20 mg P L^{-1}. This is because the maximum phosphorus concentration in the simulated wastewater was 10 mg P L^{-1} and the data points where water phase phosphorus concentration lower than 20 mg P L^{-1} showed a better linear relationship than that of the whole data set. The obtained coefficients for Langmuir type and linear type soil phosphorus adsorption isotherms are listed in Table V.1.

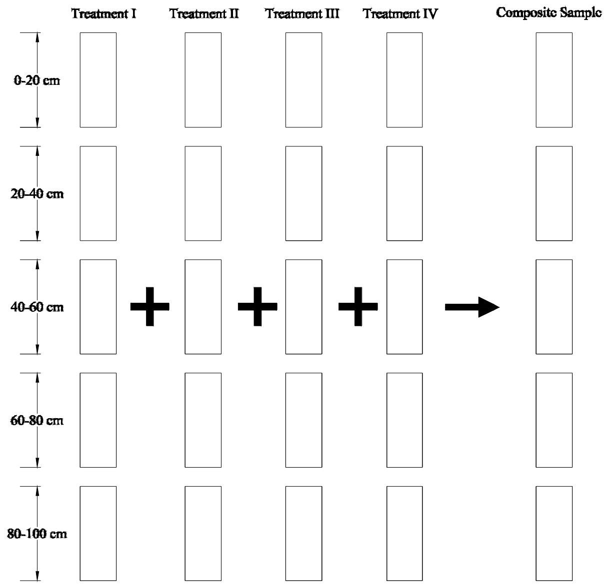

Figure V.1 Composite soil sample preparation for experimental site soil phosphorus adsorption isotherm determination.

Table V.1 Regression results of the soil phosphorus adsorption isotherm.

Constant / Soil horizon (cm)	Langmuir isotherm			Linear regression	
	K, L mg^{-1} P	S_{max}, mg P kg^{-1} soil	R^2	K_d, cm^3 g^{-1} soil	R^2
0-20	0.06	16667	0.804	300	0.51
20-40	0.06	16667	0.951	472	0.85
40-60	0.06	16667	0.923	1188	0.86
60-80	0.06	16667	0.917	793	0.82
80-100	0.06	16667	0.943	1055	0.81

Note: 1. Langmuir isotherm ($\frac{C}{S} = \frac{1}{K\,S_{max}} + \frac{C}{S_{max}}$), where S = P retained by the solid phase, mg P kg^{-1} soil; S_{max} = soil maximum P adsorption capacity, mg P kg^{-1} soil; C = concentration of P after 24 h equilibration, mg L^{-1}; K = a constant related to the bonding energy, L mg^{-1} P.

2. Linear isotherm (S=Kd×C), where S = P retained by the solid phase, mg P kg^{-1} soil; C = concentration of P after 24 h equilibration, mg L^{-1}; Kd= a constant related to the bonding energy, L kg^{-1} soil or cm^3 g^{-1} soil.

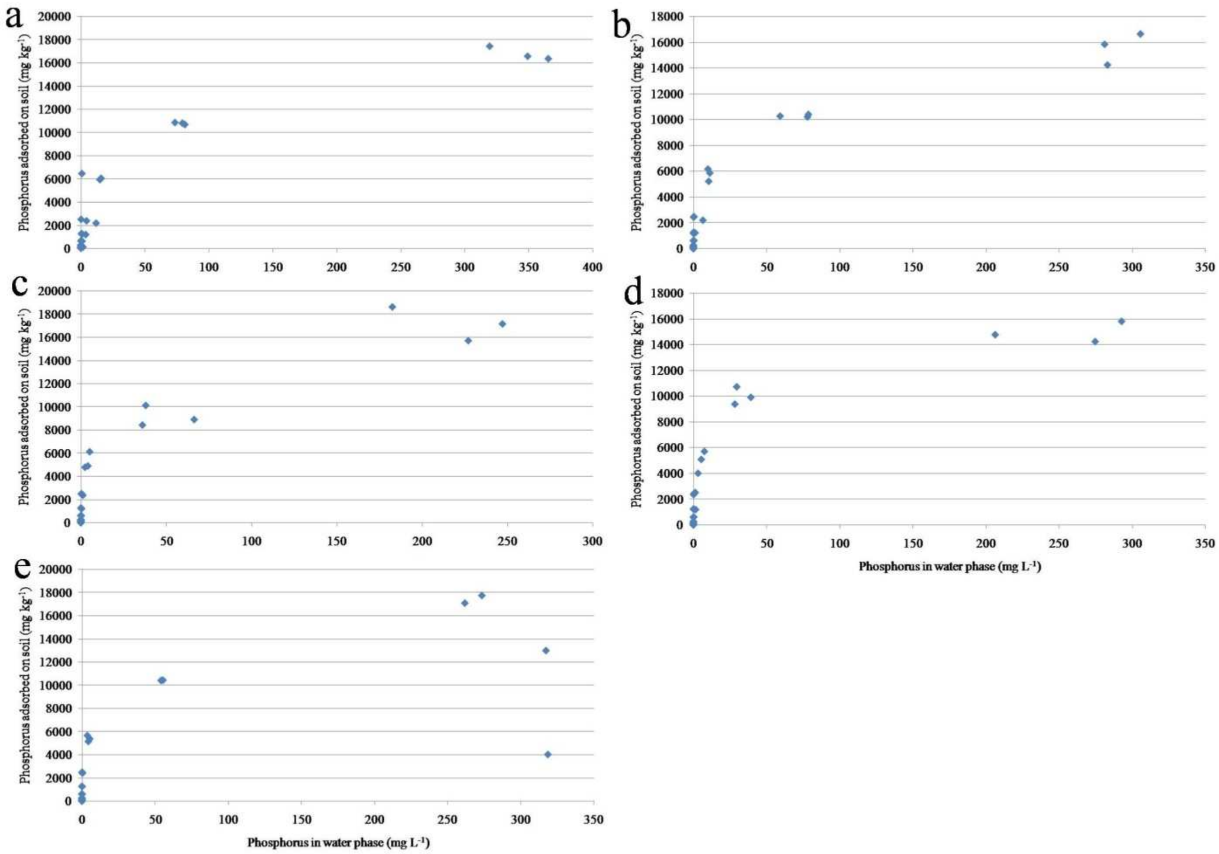

Figure V.2 Phosphorus adsorption on soil vs. phosphorus in water phase of the five soil horizons of the experimental site using composite samples from the soil cores taken from each field treatment. (a. 0-20 cm, b. 20-40 cm, c. 40-60 cm, d. 60-80 cm, e. 80-100 cm.)

APPENDIX VI

SOIL MOISTURE SENSOR VALIDATION

The ThetaProbe (ML2, Theta-T, Dynamax, TX) and PR2 (Theta-T, Dynamax, TX) soil moisture sensors are both capacitance type sensors and use the same technology to measure soil moisture content. It directly measures the soil dielectric constant (ε) and translates the readings into soil volumetric moisture content (θ) through a linear relation (Eq. VI.1). Since the soil moisture sensors were used as is during the laboratory and field experiment, soil moisture sensors were validated by calibrating the coefficients (a_0, a_1) for the ε and θ linear equation (Eq. VI.1) using intact soil cores from the experimental site. As recommended by the manufacture (Dynamax, TX), the PR2 can be calibrated with a pre-calibrated Theta-T sensor. Therefore, the calibrated coefficients for the ThetaProbe can be used with high confidence for the PR2 probe since they were monitoring the same soil on the same site. In this study, the calibration procedure was used as a validation for the used soil moisture sensors.

Since soil moisture content (θ) is proportional to the refractive index of the soil ($\sqrt{\varepsilon}$) as measured by the ThetaProbe and Profile Probe. The goal of calibration is to determine the two coefficients (a_0, a_1) that were used in a linear equation to convert probe reading (ε) into volumetric soil moisture: $\sqrt{\varepsilon} = a_0 + a_1 \times \theta$. ----Eq. VI.1

Two intact soil cores (Ø: 5.5 cm, H: 6 cm) were taken from besides the ThetaProbe at 1:30pm on May 09, 2008. The soil cores then lab measured for their volumetric moisture contents. The average reading was 39.0 % v/v and the $\sqrt{\varepsilon}$ was 4.98.

Another two intact soil cores (Ø: 5.5 cm, H: 6 cm) were taken from besides the ThetaProbe at 1:30pm on August 13, 2008. The soil cores then lab measured for their volumetric moisture contents. The average reading was 50.6 % v/v and $\sqrt{\varepsilon}$ was 6.01.

Using the readings of the two observations to solve the equation VI.1.

$4.98 = a_0 + a_1 \times 0.39$ (Eq. VI.2)

$6.01 = a_0 + a_1 \times 0.506$ (Eq. VI.3)

Resulting

$a_0 = 1.50, a_1 = 8.92$

The actual field volumetric moisture contents vs. the probe readings is plotted in Figure III.1 using the coefficients (before calibration) provided by Dynamax (TX) and the calibrated coefficients. It is necessary to emphasize that this calibration curve is suppose to work precisely only when the soil is not saturated. Free water that stays within soil empty pockets (e.g. water in cracks), if it stays close enough to the probe, can make the probe to give a reading higher than soil saturation level. Under such a circumstance, site condition where the probe is working should be explored before any solid conclusion can be made to interpret any unrealistic readings. Based on the above and Figure VI.1, the error between the pre-calibration and after calibration of the used soil moisture sensors (ML2, ThetaProbe and PR2) will be maximum 1.0 % v/v when the actual soil moisture content is 50%, v/v, and will decrease as actual field moisture content decreases.

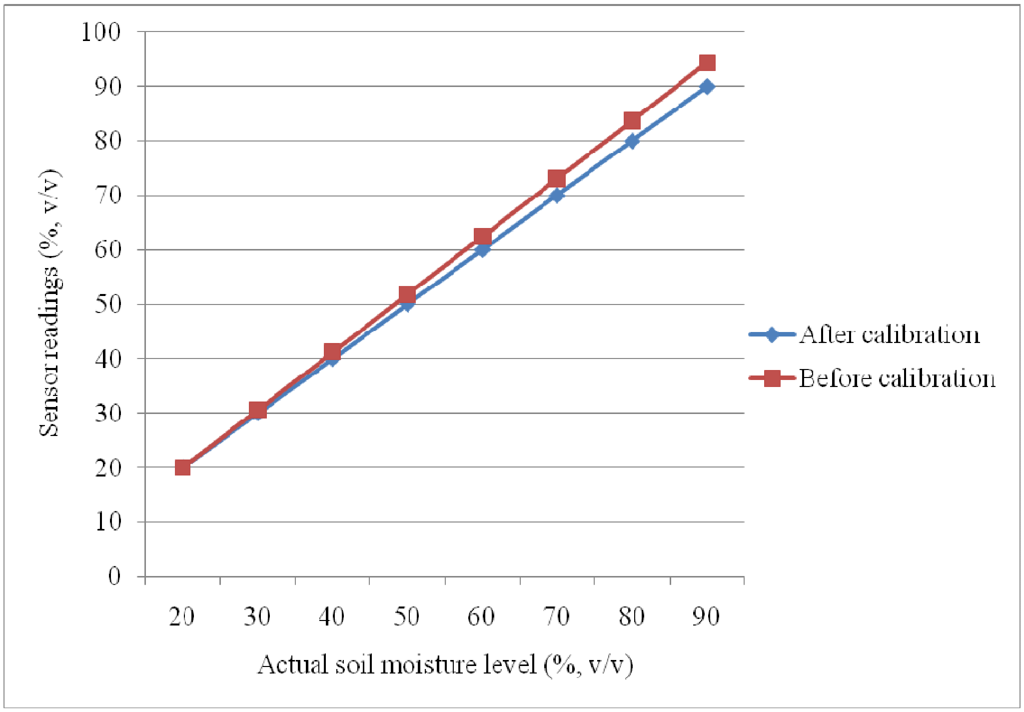

Figure VI.1 Sensor performance under calibrated coefficient and un-calibrated coefficient.

APENDIX VII

EVAPOTRANSPIRATION (ET) CALCULATION BY PENMAN-MONTEITH METHOD DEFINED BY FOOD AND AGRICULTURAL ORGANIZATION

The ET calculation equation used here is the FAO Irrigation and Drainage Paper 56 (FAO 56) version Penman-Monteith method (FAO, 2006) and is presented as:

$$ET_P^{day} = \frac{1}{L}\frac{\Delta(R_{ns} - R_{nl})}{\Delta + \gamma} + \frac{\gamma(e_s(T) - e_a)f(v)}{\Delta + \gamma} \quad \text{(Eq. VII.1)}$$

The data used for ET calculated was obtained from Alabama Mesonet's Agricultural Weather Information Service (AWIS). Since all necessary weather data at the Black Belt station was not available for the FAO 56 version Penman-Monteith method, weather data from Thorsby weather station approximately 77 km from the experimental site was used. The used parameters from AWIS are daily max (MAX), min (MIN), and average (AVG) air temperature (^oF) at 5 ft high, daily maximum (RHX) and minimum (RHN) relative humidity in percent at 5 ft high, daily solar radiation (SOLR) in watt-hours_per_square_meter at 5 ft high, daily maximum wind speed (HGU) in mph at 5 ft high. Once data was downloaded from AWSI, it was imported into a spread sheet and daily ET_0 was calculated using the automatic cell calculation function by defining the Eq 2.5 using the recorded daily weather information. Since the planting date and harvesting dates of the testing field crops are known, estimated ET (ET_c) at the experimental site was estimated by multiplying the reference ET_0 with single crop coefficient (K_c) as suggested in FAO 56 for Sorghum, Sudangrass, Rye, and Winter wheat (Table IV.1). One calculation example is provided for the month of September 2006 when the field was planted with Sorghum and Sudangrass (Table VII.2).

Table VII.1 Lengths (days) of crop development stages and single crop coefficient (K_c) used in this study.

	Initial		Development to Middle		Late	
	Days	Kc	Days	Kc	Days	Kc
Sorghum-grain	20	0	75	1.00-1.10	30	0.55
Sudan Grass hay (annual)	25	0.50	40	0.90	10	0.85
Winter Wheat - with non-frozen soils	20	0.7	130	1.15	30	0.25-0.4
Rye Grass hay	N/A	0.95	N/A	1.05	N/A	1

Adapted from FAO Irrigation and Drainage Paper 56 (2006).

Table VII.2 ET calculation example for the experimental field during the month of Septmeber 2006 (FAO 56 method).

Date	Max. air temperature, F°	Min. air temperature, F°	Ave. air temperature, F°	Max. air humidity, %	Min. air humidity, %	Solar radiation, kJ m^{-2} d^{-1}	Max. wind speed at 5 ft height, mile hr^{-1}	ET0, mm d^{-1}	Kc	ETc, mm d^{-1}
9/1/2006	89	67	78	93	42	11502.00	15	5.13	1	5.13
9/2/2006	89	69	79	93	40	11527.20	17	5.39	1	5.39
9/3/2006	90	69	80	87	36	11998.80	12	5.31	1	5.31
9/4/2006	90	68	79	94	39	12124.80	12	5.15	1	5.15
9/5/2006	91	70	81	92	35	10580.40	11	4.82	1	4.82
9/6/2006	86	67	77	93	51	5572.80	12	2.92	1	2.92
9/7/2006	89	67	78	95	30	11714.40	12	5.25	1	5.25
9/8/2006	87	64	76	95	39	10036.80	12	4.44	1	4.44
9/9/2006	89	67	78	96	40	12106.80	12	5.03	1	5.03
9/10/2006	86	67	77	96	54	7520.40	9	3.07	1	3.07
9/11/2006	89	67	78	92	39	12067.20	14	5.31	1	5.31
9/12/2006	89	71	80	85	38	8107.20	16	4.66	1	4.66
9/13/2006	78	70	74	97	69	3078.00	20	1.85	1	1.85
9/14/2006	83	62	73	96	56	7315.20	13	3.14	1	3.14
9/15/2006	82	60	71	96	38	12744.00	14	5.01	1	5.01
9/16/2006	84	60	72	96	37	12776.40	11	4.91	1	4.91
9/17/2006	88	62	75	95	28	12891.60	9	5.19	1	5.19
9/18/2006	91	67	79	94	34	12272.40	9	5.04	1	5.04
9/19/2006	89	71	80	97	52	5587.20	31	4.26	1	4.26
9/20/2006	80	58	69	96	30	11181.60	19	5.19	1	5.19
9/21/2006	76	52	64	95	35	13122.00	15	4.85	1	4.85
9/22/2006	80	56	68	86	35	12171.60	16	5.14	1	5.14
9/23/2006	88	71	80	92	60	7329.60	22	3.80	1	3.80
9/24/2006	90	71	81	95	52	5778.00	20	3.69	1	3.69
9/25/2006	85	60	73	96	64	6811.20	16	3.00	1	3.00
9/26/2006	76	55	66	95	47	12769.20	17	4.52	1	4.52
9/27/2006	76	53	65	96	39	10623.60	17	4.32	1	4.32
9/28/2006	82	58	70	96	37	11631.60	13	4.69	1	4.69
9/29/2006	83	46	65	96	40	11491.20	24	5.49	1	5.49
9/30/2006	73	49	61	96	32	12304.80	11	4.25	1	4.25

APPENDIX VIII

NRCS WEB SOIL SURVEY

Method

NRCS Web Soil Survey (WSS) provides soil data and information produced by the National Cooperative Soil Survey. It is operated by the USDA Natural Resources Conservation Service (NRCS) and provides access to the largest natural resource information system in the world. To access, type the website below and follow the specific instructions shown on the web page to locate the area of interest and extract desired soil information from available options which is based on current SSURGO database (NRCS, 2007). The current soil area data is version 4, September 6, 2008.

Website: http://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx

For this study, site description and physical soil properties were extracted to assist understanding field soil moisture profile monitoring data carried from May 2008 to September 2008. The relevant information is listed below.

Site Description

Map Unit Setting

- Mean annual precipitation: 48 to 56 inches
- Mean annual air temperature: 61 to 68 degrees F

- Frost-free period: 200 to 250 days

Map Unit Composition

- Houston and similar soils: 85 percent
- Minor components: 1 percent

Description of Houston clay

Setting

- Landform: Hillslopes
- Landform position (two-dimensional): Footslope
- Landform position (three-dimensional): Base slope
- Down-slope shape: Linear
- Across-slope shape: Linear
- Parent material: Clayey marine deposits derived from chalk

Properties and qualities

- Slope: 1 to 5 percent
- Depth to restrictive feature: 48 to 72 inches to paralithic bedrock
- Drainage class: Moderately well drained
- Capacity of the most limiting layer to transmit water (Ksat): Very low to moderately low (0.00 to 0.06 in/hr)
- Depth to water table: About 48 to 72 inches
- Frequency of flooding: None
- Frequency of ponding: None
- Available water capacity: Moderate (about 8.4 inches)

Interpretive groups

- Land capability (nonirrigated): 2e

Typical profile

- 0 to 10 inches: Clay
- 10 to 42 inches: Clay

- 42 to 80 inches: Clay

Minor Components

Eutaw

- Percent of map unit: 1 percent
- Landform: Depressions
- Landform position (two-dimensional): Footslope
- Landform position (three-dimensional): Dip
- Down-slope shape: Concave
- Across-slope shape: Concave

Physical Soil Properties

Report—Physical Soil Properties

Physical Soil Properties– Dallas County, Alabama														
Map symbol and soil name	Depth	Sand	Silt	Clay	Moist bulk density	Saturated hydraulic conductivity	Available water capacity	Linear extensibility	Organic matter	Erosion factors			Wind erodibility group	Wind erodibility index
										Kw	Kf	T		
	In	Pct	Pct	Pct	g/cc	micro m/sec	In/In	Pct	Pct					
29—Houston clay, 1 to 5 percent slopes														
Houston	0-10	-16-	-26-	50-59- 67	1.10-1.45	0.01-0.42	0.12-0.16	6.0-8.9	2.0-5.0	.37	.37	5	4	86
	10-42	-12-	-27-	52-62- 71	1.10-1.45	0.01-0.42	0.12-0.16	9.0-25.0	0.0-2.0	.32	.32			
	42-80	-10-	-24-	53-66- 78	1.10-1.45	0.01-0.42	0.12-0.16	9.0-25.0	0.0	.32	.32			
Eutaw	0-4	-26-	-29-	40-45- 50	1.40-1.50	0.42-1.40	0.16-0.19	6.0-8.9	1.0-3.0	.32	.32	5	4	86
	4-60	-10-	-25-	60-65- 70	1.55-1.65	0.01-0.42	0.15-0.18	9.0-25.0	0.0	.28	.28			
55—Sumter silty clay, 1 to 5 percent slopes														
Sumter	0-10	- 8-	-51-	32-41- 50	1.30-1.60	0.42-14.00	0.12-0.17	6.0-8.9	2.0-5.0	.37	.37	3	4	86
	10-21	- 6-	-48-	35-46- 57	1.15-1.55	0.42-14.00	0.12-0.17	6.0-8.9	1.0-2.0	.37	.37			
	21-28	- 6-	-48-	35-46- 57	1.15-1.50	0.42-14.00	0.11-0.16	3.0-5.9	0.0-0.5	.32	.37			
	28-80	—	—	0- 0- 0	—	0.00-4.00	0.00	0.0	0.0					

Data Source Information

Soil Survey Area: Dallas County, Alabama
Survey Area Data: Version 4, Sep 16, 2008

FigureVIII.1 Screen capture of the soil report generated for physical soil properties of the experimental site.

APPENDIX IX

HYDRUS 2D NUMERICAL MODELING

HYDRUS 2D (Šimůnek et al., 1999), a PC based modeling environment for analysis of water flow and solute transport in variably saturated/unsaturated porous media, was used to simulate water and nutrient movement adjacent to drip emitters in treatment I using selected field data. The purpose of this simulation is to estimate water and nutrient short- and long-term moving trend under the soil moisture controlled SDI wastewater disposal over the testing Houston clay.

Model Setup

The boundary selected for numeric simulation was a rectangular soil profile, 100 cm deep by 60 cm wide. This control area represents the cross-sectional space between two emitters on adjacent drip laterals (Figure IX.1). Facing emitters are represented as two 16 mm semi-circles 20 cm deep at the side boundary. Each emitter is set with a time variable flux representing daily wastewater application. Soil horizons are characterized in the model following the site soil analyses (Table IX.1). The upper boundary of the model was set as a time-variable atmospheric surface associated with daily ET and precipitation. The bottom boundary of 100 cm was set to permit free drainage. The side boundaries, excluding emitters, were set to exclude lateral flux. Soluble nutrients, NH_4^+-N and phosphorus, entered the modeled soil profile as wastewater flux at the two emitters. Two hypothetical observation points for data summary were selected at 20 cm and 45 cm depths in the middle of the modeling space to compare with field observed soil moisture data (Figure IX.1). Two additional hypothetical observation points were selected at 45 cm and 100 cm depths directly beneath one emitter to monitor variation of water phase nutrient concentration over time. Required hydraulic parameters for numerical modeling, such as soil water retention function, were estimated using the neural prediction function embedded in HYDRUS 2D, based on the actual soil texture for each horizon (Table IX.1).

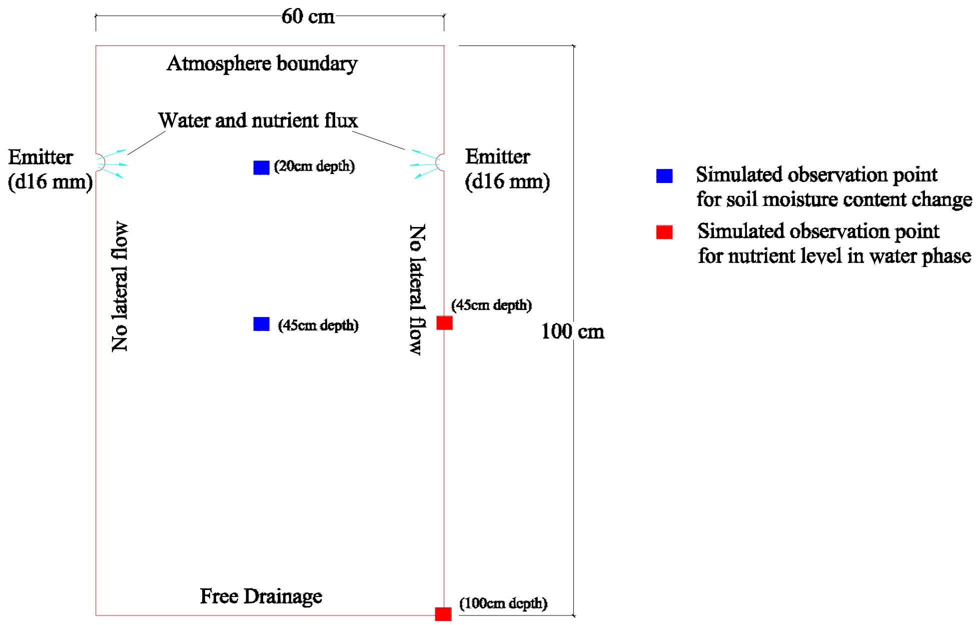

Figure IX.1 HYDRUS 2D simulation domain and boundary conditions.

Table IX.1 Particle size distribution of the soil horizons of the experimental site.

Horizon	Lower Depth	Particle Size Distribution		
		Sand	Silt	Clay
	cm	%		
Ap1	23	7.09	39.63	53.28
Ap2	42	8.27	38.04	53.7
BA	63	10.17	33.38	56.45
Bkss1	88	3.50	35.93	60.57
Bkss2	152	3.10	25.80	71.10

Source: Auburn University Pedology Laboratory

Model input included daily system hydraulic disposal rate (mm d^{-1}), precipitation (mm), nutrient (N, P) concentration of the applied synthetic wastewater (mg L^{-1}), and calculated daily field ET (mm d^{-1}). Initial soil and soil water nutrient levels were set to zero to illustrate the net movement of applied nutrients similar to Beggs et al. (2004). Nutrient concentrations of the disposed wastewater were set to 80 mg N L^{-1} for ammonium nitrogen (NH_4^+-N) and 10 mg P L^{-1} for total phosphorus to match the synthetic wastewater. Nitrate nitrogen (NO_3^--N) was simulated as the only daughter product of NH_4^+-N and was assumed not initially contained in the synthetic wastewater since chemical analysis of the used synthetic wastewater contained negligible NO_3^--N. Since crop uptake and soil adsorption for nitrogen and phosphorus are the most significant pathways for nitrogen and phosphorus in soils (Venhuizen, 1995), simulated chemical/physical reactions in this modeled soil matrix are nitrification/denitrification, soil adsorption, and crop uptake for nitrogen; soil adsorption and crop uptake for phosphorus. Water phase diffusion coefficients were set to 1.60 $cm^2\ d^{-1}$ for NH_4^+-N, NO_3^--N and phosphorus based on Beggis et al. (2004) <u>on their simulation on soiltesxture</u>. Soil adsorption coefficients were set to 6.0×10^{-6} $cm^3\ g^{-1}$ for NH_4^+-N and 1×10^{-7} $cm^3\ g^{-1}$ for NO_3^--N for the entire soil profile based on Beggis et al. (2004) on their simulation on loam and clay loam soils. Linear adsorption coefficients (K_d, $cm^3\ g^{-1}$) for phosphorus were assigned to soil according to lab measurement (Table 5.1); 300 for 0-20 cm, 472 for 20-40 cm, 1188 for 40-60 cm, 793 for 60-80 cm, and 1055 for 80-100 cm. The nitrification coefficient for NH_4^+-N was set at 0.72 d^{-1} and the denitrification coefficient for NO_3^--N was set at 0.072 d^{-1} as used by Misra (1974) for long column of silt loam soil, low water flow, at 0.5% (m/m) oxygen. Root mass (dimensionless) was simulated as linearly decreasing in the 100 cm depth, 1 at the top and 0 at the bottom. Root water uptake was modeled using the predefined HYDRUS 2D routine for forage.

Modeling scenarios included three one-year synthetic wastewater applications scenarios and one long term (10-year) synthetic wastewater application scenario (Table IX.2). The one-year (June 14, 2007 to June 24, 2008) synthetic wastewater application period was simulated with comparative scenarios including: 1) root water and nutrient uptake with nitrification/denitrification and soil adsorption for nitrogen, and soil adsorption for phosphorus; 2) root water and nutrient uptake only; and 3) No root water and nutrient uptake, has nitrification/denitrification and soil adsorption for nitrogen, and soil adsorption for phosphorus. A 10-year simulation (scenario 4) repeats the one-year synthetic wastewater application from June 14, 2007 to June 24, 2008 with both root water and nutrient uptake and solute reactions mode on, similar to scenario 1. This long-term simulation was used to assess long term water and nutrient movement in the drain field.

Table IX.2 HYDRUS 2D simulation scenarios.

Scenario #	Simulation period	Root nutrient uptake	Solute reaction[1]	Initial conditions	
				Water content[2]	Soil nutrient concentration
1	Annual[3]	Yes	On	40%, v/v of the entire soil profile	None
2	Annual	Yes	No		
3	Annual	No	On		
4	Ten years[4]	Yes	On		

Notes: 1. Solute reaction includes nitrification of NH_4^+-N, denitrification of NO_3^--N, soil adsorption for NH_4^+-N, NO_3^--N, and phosphorus.
2. Verified using HYDRUS 2D as the field capacity of the simulating soil matrix.
3. Duration from June 14, 2007 to June 24, 2008 when synthetic wastewater application experiment was carried.
4. Repeat scenario 1 consecutively ten times.

The HYDRUS 2D model was not calibrated over the 12-month simulation period because of the experimental setup and ambient environment. For a valid model calibration, the differences between simulated and observed data can only be ascribed and quantified to intrinsic limitations of field instruments (Šimùnek and Hopmans, 2002), and the applied process model and selected hydraulic relationships must be an exact description of the soil physical behavior (Šimùnek and Hopmans, 2002). For this particular study, one major difference between the actual field condition and the modeled soil profile is that soil cracking processes during hot, dry summer condition breaks soil continuity within the field, a scenario that the HYDRUS 2D model is not able to simulate. Therefore, a non-perfect 12-month simulation model performance is expected, especially, since spatial and temporal variability in parameter values such as K_{sat} will result in uncertain model outputs when describing or predicting a natural event (Shirmohammadi et al., 2006).

Simulation Result

Simulated Field Capacity

Since the soil moisture controlled SDI wastewater disposal system choose its operation window around field capacity of the experimental site, it is necessary to find out the simulated "field capacity" of the modeled soil matrix and use it as the initial status for the simulation. The defined condition for field capacity requires no evaporation occurs from the soil surface, no water table or slowly permeable barriers occur at shallow depths in the profile, and the soil profile should be let free drained until no substantial drainage occurs (SSSA, 1986). To mimic these conditions, the modeled soil profile (Figure IX.1) was initially assigned a volumetric soil moisture content of 50% and then let bottom free drained for 60 days with the soil profile of no root uptake, no precipitation, and no irrigation. The simulated time course of the volumetric soil moisture content of the soil profile is illustrated in Figure IX.2.

From the soil moisture content's dropping trend, "substantial drainage" stops at around 40%, v/v (Figure IX.2). Since this number is close to the lab approximated field capacity (37%-44%, v/v) of the experimental site at 20 cm depth, it was then used as the initial soil moisture content of the soil profile for scenario simulations.

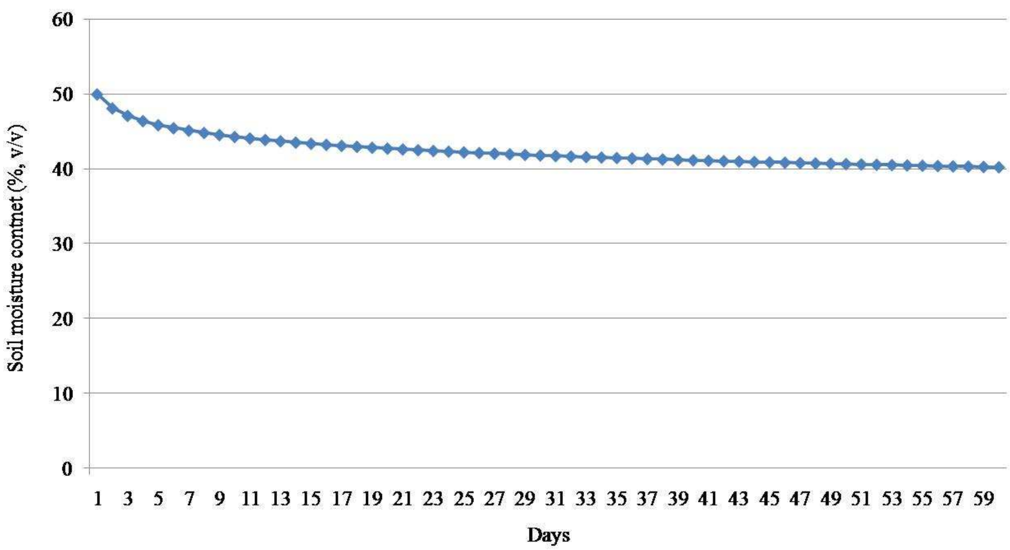

Figure IX.2 Soil moisture content (%, v/v) time course of the free drained simulating soil profile.

Model Evaluation in Terms of Soil Water Movement (June 14, 2007 to June 24, 2008)

Simulated soil moisture levels at 20 and 40 cm depth under scenario 1 were graphically compared first with field observed field data to illustrate the difference between simulation and field observation (Figure IX.3). It is graphically shown that the simulation process overestimates soil moisture levels from June 14, 2007 to November 14, 2008, slightly underestimates from November 15, 2007 to January 9, 2008, and obviously underestimates from January 2008 to June 2008.

Quantified comparison results using Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and root mean square error (RMSE)-observation standard deviation ratio (RSR) as described by Moriasi et al. (2007) are tabulated in Table IX.3. Over the 12 month simulation, each of the three indexes indicates a poor correlation with field data. NSE rated simulated results “Unacceptable”, PBIAS rated the simulated results “Underestimation”, and RSR rated the simulated results “Errors” (Table IX.3).

One explanation for the overestimation period is that the model failed to account for seasonal shrinking that caused preferential flows during dry summer conditions. Field observed system hydraulic disposal rates, in the actual cracking field conditions, substantial of those applied water will be conducted into deep soils much quicker due the cracking between soil aggregates. However, if applied in a more homogenized soil profile, these water would be expected to exert higher influence on the upper 45 cm soil.

For the underestimated period, it can be explained with the other feature of the clay soil, swelling during wet conditions. The clay soils naturally swelled in wet weather as well as in April and May of 2008 when local monthly precipitations were above average levels (Figure 5.5). With a negligible hydraulic disposal rate (0-2.44 cm month^{-1}) from the SDI dosing system, precipitation was responsible for field wetting. The swelling of the clay soils under wet conditions decreased soil water percolation rates (Bouma and Loveday, 1987; Weaver et al., 2005) and also held water longer. The HYDRUS 2D model did not accurately account for these physical and hydraulic property changes, resulting lower (underestimated) soil moisture content in the upper 45 cm.

Simulated results can be improved if related soil physical and hydraulic properties are properly reflected during the simulated time frame (Cabidoche and Guiliaume, 1998), and the current HYDRUS 2D simulation on SDI irrigation on clay soils is recommended for periods, if possible, when clay soil shrink/swell processes are at their minimum extent. Nevertheless, comparative results between simulation and observation in this study suggest that the experimental SDI system is disposing water faster than can be held by the soil during a hot, dry summer, more so during drought conditions. The current field experiment setup and field moisture sensing strategy does not adequately address this seasonal and spatial change in drain field hydraulic properties, and also soil cracking development seems not able to be deminished by those disposed water.

Table IX.3 Evaluation of the simulated (scenario 1) soil moisture levels compared to observed field data at 20 cm and 45 cm depths from June 14, 2007 to June 23, 2008.

Evaluation index / Time frame	Nash-Sutcliffe efficient (NSE)[1]		Percent bias (PBIAS)[2]		RMSE-observation standard deviation ratio (RSR)[3]	
	20 cm	45 cm	20 cm	45 cm	20 cm	45 cm
June 14, 2007- June 23, 2008 (whole year)	-2.126, Unacceptable	-0.167, Unacceptable	6.31, Underestimation	5.78, Underestimation	1.77, Errors	1.08, Errors
June 14, 2007- November 14, 2007	0.067, Unacceptable	0.44, Unacceptable	0.03, Overestimation	-3.80, Overestimation	0.06, Errors	0.75, Errors
November 15 2007- January 9, 2008	0.998, Unacceptable	0.999, Unacceptable	4.59, Underestimation	0.072, Underestimation	0.04, Errors	0.026, Errors
January 10, 2008- June 23, 2008	0.98, Unacceptable	0.977, Unacceptable	13.98, Underestimation	14.35, Underestimation	0.15, Errors	0.15, Errors

Notes: 1. Based on the calculated evaluation index (range from -∞~1), the ratings are categorized into Optimal (1), Acceptable (0-1), and Unacceptable (≤0).

2. Based on the calculated evaluation index (range from -∞~+∞), the ratings are categorized into Optimal, Underestimation (>0), and Overestimation (<0), with lower the magnitude of the index the better the simulation result.

3. Based on the calculated evaluation index (range from 0~+∞), the ratings are categorized into Optimal (0), Errors (>0), with lower the index the better the simulation result.

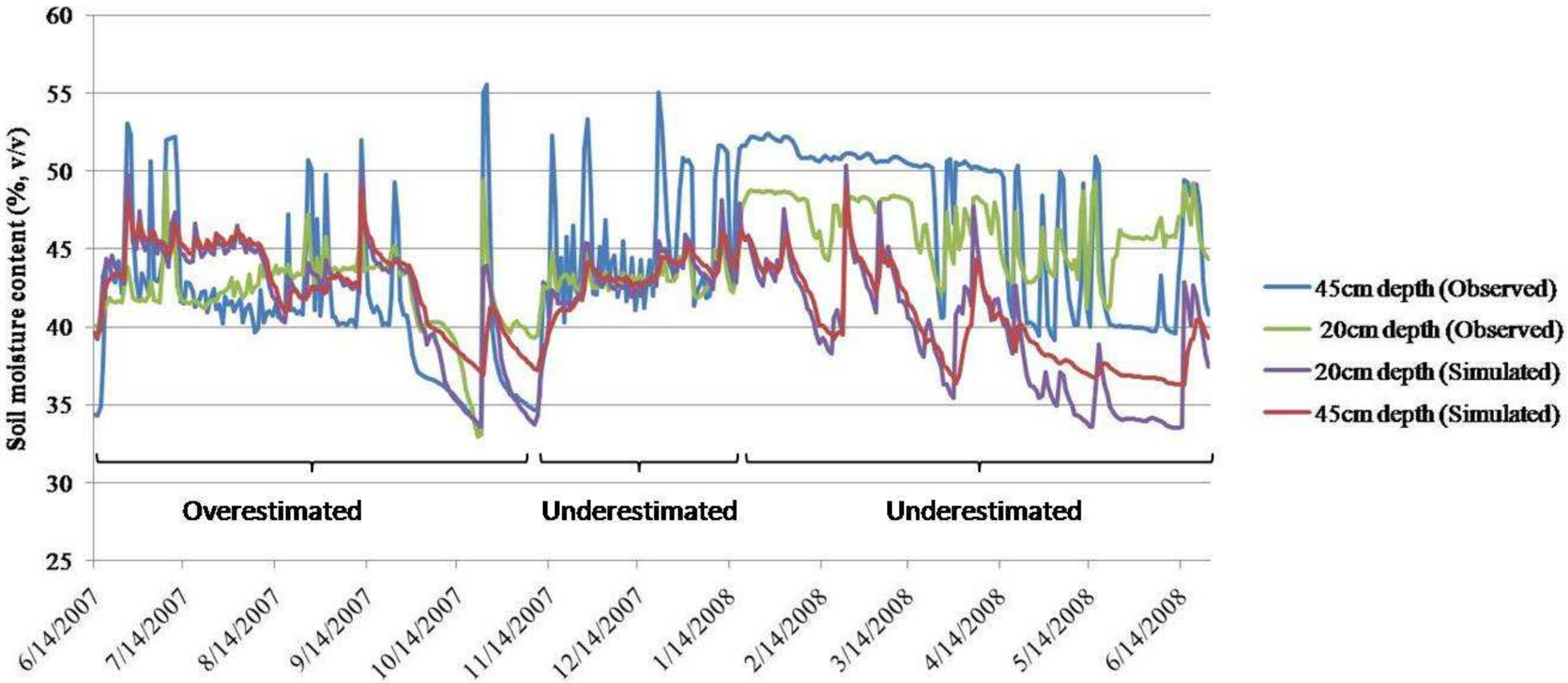

Figure IX.3 Simulated (scenario 1) soil moisture contents at 20 cm and 40 cm depths versus actual field monitoring in treatment I during the one-year synthetic wastewater application from June 14, 2007 to June 24, 2008.

Scenario Simulated Nutrient Output After One Year Synthetic Wastewater Application

Annual nutrient output from simulated soil profiles in scenarios 1, 2, and 3 are compared in Figure V.4. Simulated nitrification is more effective than crop uptake in preventing NH_4^+-N from leaching (below 100 cm depth) since NH_4^+-N leaching only occurs when there is no nitrification in the soil profile (scenario 2), while nitrification itself without crop uptake can prevent NH_4^+-N leaching (scenario 3). More than 30% of annually applied NH_4^+-N is lost to leaching under scenario 2 when there is no chemical mechanism to attenuate nitrogen. Results for NO_3^--N indicate that both denitrification and crop uptake are important in limiting NO_3^--N from leaching since denitrification cannot totally eliminate NO_3^--N leaching only by itself (scenario 3). Under scenario 2 when there is no denitrification but crop uptake for NO_3^--N, although there is no NO_3^--N input due to no nitrification of NH_4^+-N, it is also anticipated that NO_3^--N leaching issue can also be magnificent if applied at a comparable level as NH_4^+-N (80 mg L^{-1} in this study).

Although the nitrification/denitrification reaction coefficients are not directly field derived and won’t be consistent throughout seasons in a natural drain field, the results of the scenario comparison indicate the importance of maintaining intermittent nitrification/denitrificaiton conditions within the soil matrix to limit nitrogen deep percolation (Beggs et al., 2004; Venhuizen, 1995), especially in case where nitrogen is in surplus as a crop nutrient (Broadbent and Reisenauer, 1988).

Comparable results for phosphorus show that the high soil adsorption capacity for phosphorus retains applied phosphorus. As a major phosphorus retaining mechanism, phosphorus leaching would be expected to increase if the high soil adsorption capability were to become ineffective as in scenario 2 when there is only crop uptake to keep phosphorus from leaching.

The simulated phosphorus fate and transport pathways are much less complex than that in actual natural field conditions (Reneau et al., 1989, Venhuizen, 1995). Nevertheless, simulated results deliver the message that the length of time required to exhaust a soil's phosphorus adsorption capacity depends on the soil, disposal rate, and available phosphorus attenuation mechanisms such as adsorption/precipitation (Sawhney and Hill, 1975; Venhuizen, 1995).

Nutrient attenuation mechanisms that take effect within the same or different soil zones can decrease nutrient leaching even though deep percolation is substantial. On the other hand, poor nutrient removal, whether via crop uptake or chemical/physical attenuation, will lead to undesirable nutrient leaching when the field is consistently loaded above field capacity.

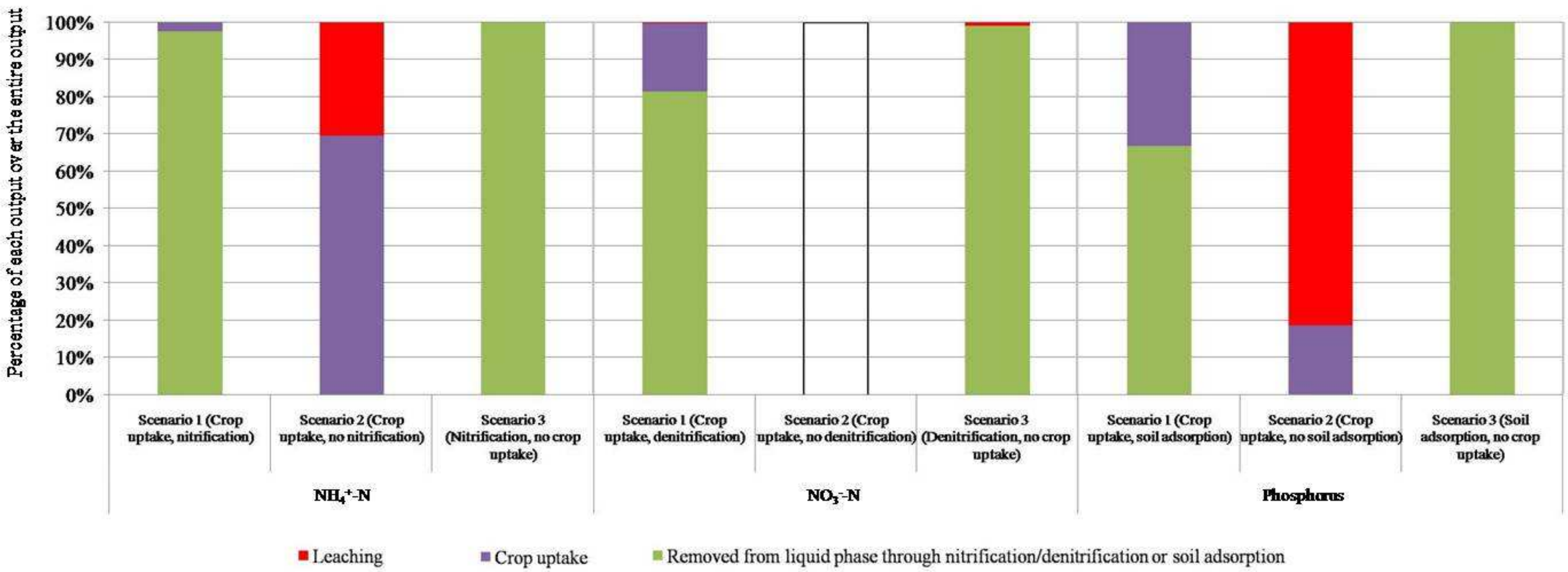

Figure IX.4 Simulated (scenarios 1, 2, and 3) NH_4^+-N, NO_3^--N, and phosphorus outputs within the simulated soil domain after one year synthetic wastewater application. (For scenario 2, no NO_3^--N presences in the soil domain since NH_4^+-N is not nitrified)

Long Term (10-Year) Simulation of Water and Nutrient Movement in the Drain Field

Water percolation at 100 cm depth over a 10 year simulation shows a repetitive pattern with substantial water percolation in both summer and winter (Figure IX.5a). Simulated soil moisture content at 20 and 45 cm depth show a regular fluctuation between 38%, v/v and 48%, v/v. This simulated result suggests that the hydraulic response to repeated wastewater disposal reached a "dynamic" steady state starting from the first year.

Over the entire 10 year period, in the simulated soil water phase there is no indication of NH_3^+-N presence at 50 cm and 100 cm depths. However, NO_3^--N concentration in percolates at the 45 cm depth show repetitive surges (Figure IX.5b) coincident with high and low water percolation cycles as indicated in Figure IX.5a. While NO_3^--N surges at 45 cm periodically reaches almost 10 mg L^{-1}, it does not extend to 100 cm depth where NO_3^--N consistently maintained below 0.05 mg L^{-1}. This result indicaties that the simulated denitrification and crop uptake process effectively attenuates NO_3^--N to a negligible level while hydraulic loading moves the NO_3^--N toward the 100 cm depth.

Simulated phosphorus levels in the percolates at 50 cm and 100 cm depths indicate accumulating phosphorus levels resulting from repetitive water loadings. Simulated phosphorus levels in the percolates are relatively stable at 9 mg P L^{-1} at 45 cm depth, and 8 mg P L^{-1} at 100 cm depth. Since there are only soil adsorption and crop uptake are simulated as the mechanisms to remove phosphorus from water phase, the initial rising then stabled phosphorus levels in the percolates indicates that consistent

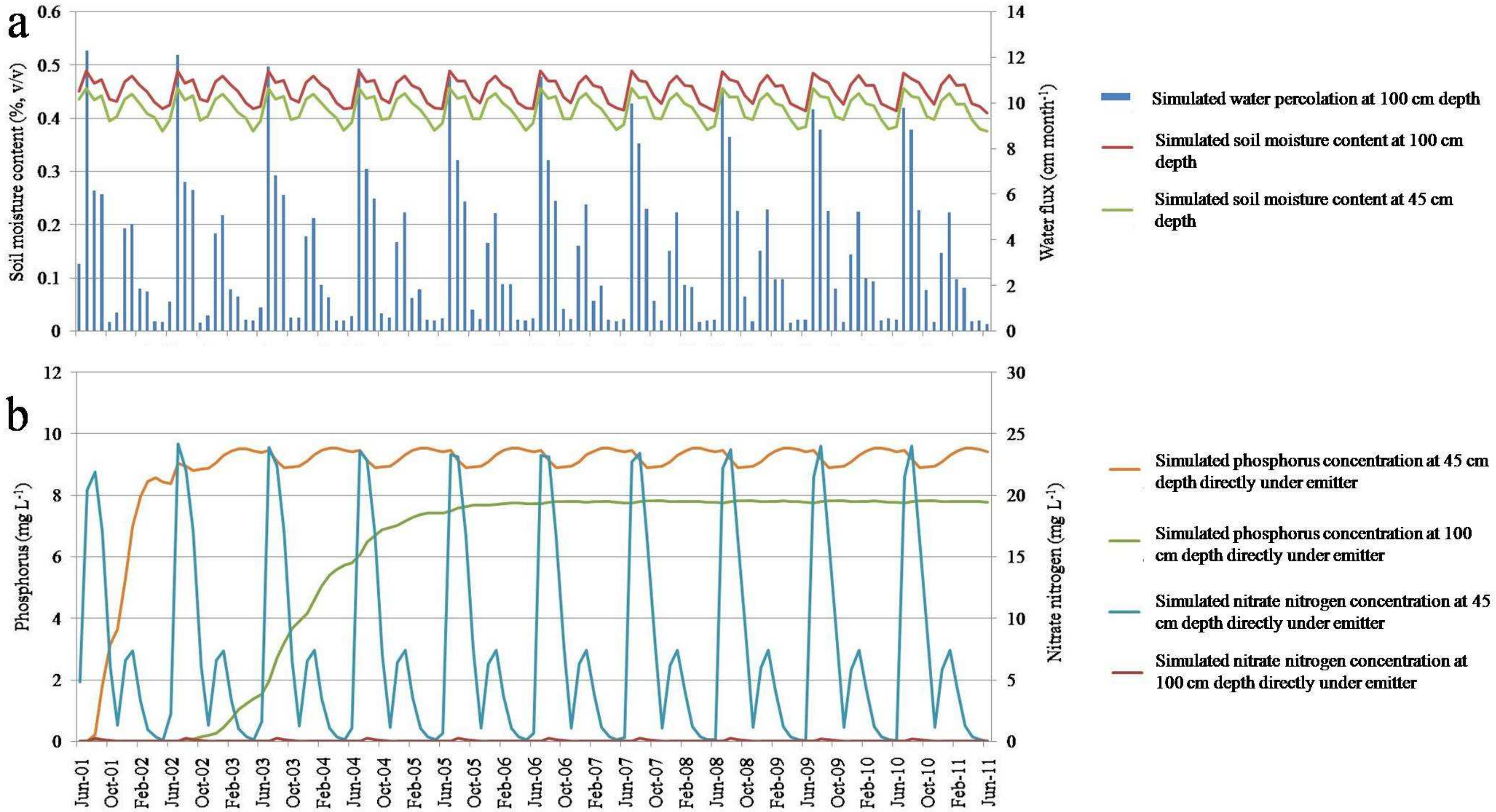

Figure IX.5 Simulated (Scenario 4) monthly water percolation and nutrient concentrations in water phase during the 10-year simulated period. (a. simulated monthly water percolation and soil moisture content change; b. simulated NO_3^--N and phosphorus concentration variations at simulated observation points)

phosphorus loading already exhausted the soil's phosphorus adsorption capacity at shallow soil depths, leaving crop uptake as the remaining removal mechanism. This simulated phosphorus movement indicates that yearly repetitive drain field hydraulic loading into the soil has the potential to carry phosphorus deeper into the soil.

However, phosphorus is not commonly found present in deep soils since there are numerous mechanisms to retain applied phosphorus (Reneau et al., 1989). Sawhney and Hill (1975) and Venhuizen (1995) reported phosphorus slowly creeping forward in onsite wastewater drain fields as soil phosphorus adsorption sites are depleted. Nevertheless, finding it present in deep clay soils due to ineffective phosphorus immobilization mechanism, such as preferential flows stimulated by soil cracking development is still possible (Larsson et al., 2007; Muukkonen et al., 2009), especially under the actual experimental site that is dominant with Huston clay, a Vertisol notorious for its cracking development during dry seasons (Amidu et al., 2007; Kishne et al., 2009).

Conclusion

Short term one year scenario comparisons suggest the necessary to maintain effective spatial and temporal nutrient attenuation mechanism along soil profile to prevent nutrient leaching. Long term 10-year simulation indicates the accumulated effect that repetitive hydraulic overloading can have on ground water nutrients. Even though short term adverse effect from wastewater drain field overload may not be discernable, cumulative impact has been shown can lead to a noticeable pollutants elevation in ground water. Based on simulation results, although less complex than actual field conditions, the importance of maintaining nutrient attenuation mechanisms such as nitrification/denitrification, crop uptake, soil adsorption, and immobilization in the soil to reduce nutrient level in percolates should be emphasized. On the other hand, even if drain field is hydraulically overloaded, a series of well maintained spatial and temporal nutrient attenuation mechanisms can still control nutrient percolation within a certain depth or even enhance the drain field capacity for sudden surge flows.