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Modeling Onsite Wastewater Treatment Systems in a Coastal Texas Watershed

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Abstract Onsite wastewater treatment systems (OWTSs) are commonly used to treat domestic wastewater in the Dickinson Bayou watershed, located between Houston and Galveston. The Dickinson Bayou is classified as "impaired" by the Texas Commission on Environmental Quality due to high levels of indicator bacterium, *Escherichia coli*. Failing OWTSs in the watershed are possible sources for the impairment of the bayou. Nearly all of the watershed is at risk to failing OWTSs due to high water table and clay content in the soil. The HYDRUS modeling software for water and solute flow through variably saturated media was used to simulate the performance of (1) conventional OWTSs, (2) aerobic treatment units (ATUs) with spray distribution, and (3) mounded OWTSs under conditions indicative of the Dickinson Bayou watershed. The purpose of the study was to simulate system performance under existing conditions. Simulation results indicated that both the conventional and ATU systems fail due to effluent ponding and *E. coli* transport to the land surface due to high water tables and clay soils in the watershed. Simulations indicated that conventional and ATU systems failed when rainfall intensity was greater than 0.25 cm/h. However, the model simulations indicate mound systems did not fail under existing conditions as they did not allow *E. coli* to reach the surface or ponding to occur. Consequently, mound systems can be considered as better systems in this watershed to minimize bacterial loadings.

Keywords Onsite wastewater treatment systems (OWTSs) · Septic system · Aerobic treatment unit (ATU) · HYDRUS · Vadose zone

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1 Introduction

Dickinson Bayou on the Texas coast has been found to be impaired due to higher than acceptable concentrations of *Escherichia coli*. Failing onsite wastewater treatment systems (OWTSs) may be contributing to the impairment through excess runoff. HYDRUS-2D (Simunek et al. 2011) was used to simulate an existing conventional septic system and potential alternatives (aerobic treatment unit (ATU) and mound system) under conditions representative of the watershed in order to determine which systems mitigate the risk of contributing to this impairment. OWTSs were considered to be in hydraulic failure when the soil saturated and surface runoff occurred. Treatment failure was marked when *E. coli* was transported to the surface or to the water table.

Approximately 25 % of the US population is serviced by onsite systems, most commonly by conventional septic tanks with soil absorption fields (USEPA 2002). According to a 2001 study, 1.5 million households in Texas use OWTSs (Reed Stowe and Yanke 2001). The majority of these systems are found in rural areas where 65 % of the households use OWTSs (Motz et al. 2011). OWTSs are effective wastewater treatment solutions that can adequately protect public health and the environment when placed in soils with appropriate treatment capabilities and designed and installed properly with regular maintenance (USEPA 2002).

OWTSs in the Dickinson Bayou watershed (DBW) include conventional septic systems with soil absorption fields (70 % of total OWTSs) and aerobic treatment units with spray distribution (hereon referred to as "ATUs") (30 %) (Highfield et al. 2011). Most systems installed since 2003 have been ATUs. There are no documented mound systems in the region, but they have been identified as appropriate alternatives to conventional systems. Conventional systems and mound systems use the same processes for treatment. Each has a septic tank for the removal of solids, fats, and oils and a soil absorption field to treat liquid effluent. The treatment capacity of conventional systems is highly dependent on the soil type and depth to the water table in the soil absorption field. Conventional systems do not function properly in clay or rocky soils with a high water table or soils saturated for long periods (Lesikar 1999a). A mound system can be used to mitigate problems associated with unsuitable conditions in the soil absorption field. This is accomplished by mounding soil on the surface to allow the drain lines to be buried in suitable soil and to increase the distance from the water table (Lesikar and Weynand 2002). ATU systems are an alternative to conventional systems that use aeration and disinfection, typically chlorination, to treat liquid waste before being sprayed on the surface.

EPA studies have found 10–20 % failure rates for OWTSs in the USA (USEPA 2002) due to age, siting, design, regulation and oversight, compliance, education, and maintenance. It is estimated that 148,573 (13 %) of total reported OWTSs in Texas were classified as chronically malfunctioning in a 2001 survey study (Reed Stowe and Yanke 2001). The major factors found to contribute to OWTS failure in eastern and coastal Texas (Region IV) (where the study site is located) were siting, age, and owner maintenance problems. The study found that soils were the leading cause for failure and 53 % of OWTSs in Region IV are placed in soils unsuitable for conventional systems (Reed Stowe and Yanke 2001). Additionally, the study found that OWTS owners do not receive adequate education for maintaining their systems (Reed Stowe and Yanke 2001).

In 1997, changes in Texas Commission on Environmental Quality (TCEQ) permitting for OWTSs required site evaluations of soils and water tables in Texas. This requirement disqualified new conventional OWTSs from being installed in many areas in the watershed. From 1995 to 2006, installations of new conventional OWTSs dropped from 84 to 23 % and more advanced systems, such as ATUs and mound systems, increased (DBWP 2009). However, the inadequate operation and maintenance of ATU systems has become a common problem (Reed Stowe and Yanke 2001). Disinfectants for ATU systems were either incorrectly added or not added at all, and many residents did not renew required maintenance contracts. Without necessary maintenance, ATUs cannot function as designed and result in spraying high concentrations of fecal contamination on the surface (Reed Stowe and Yanke 2001). Even with changes in regulations, the most recent TCEQ study estimates that 1546 of the 4857 OWTSs in the DBW are failing (TCEQ 2011).

The purpose of this study was to evaluate through model simulations for water and *E. coli* transport in three different types of OWTSs located in a watershed with a biologically impaired body of water, clay soils, and high water tables for their possible contribution to that impairment. The hypothesis tested was that conventional systems would contribute to *E. coli* contamination due to restrictive soil filtration and/or high water tables; malfunctioning ATU systems would contribute to contamination due to surface application of highly concentrated waste; and mound systems would be an effective alternative to both systems where soil profiles are not suitable for conventional systems.

2 Materials and Methods

2.1 Study Area

The study area is a residential area between the cities of Dickinson and Santa Fe in the coastal plain of Texas. Figure 1S displays a map of DBW provided by the

Texas Coastal Watershed Program (Texas Coastal Watershed Program 2013) which contains known OWTSs located within DBW as well as the study location. The watershed containing the study site is drained by Dickinson Bayou. The bayou flows from west to east approximately 39 km and drains directly into Dickinson Bay which then flows into Galveston Bay. Dickinson Bayou was classified as "impaired" by the TCEQ in 1996 due to unacceptable levels of *E. coli* (more than 126 CFU/100 mL) (Texas Coastal Watershed Program 2010). Through total maximum daily load (TMDL) studies, one of the potential sources of contamination was septic waste from failing OWTSs reaching the bayou via runoff from surface discharge and storm water runoff (TCEQ 2011).

The study area focused on two neighboring subdivisions (Figure 1S), one with newer ATU systems and one with older conventional septic systems. Each subdivision had 16 homes adjacent to a drainage ditch flowing into the Dickinson Bayou. The layout of a typical system in the watershed used in the simulations was obtained from a Galveston County Health District Private Wastewater Disposal System Inspection Report. The simulated system modeled an actual system installed and approved in this subdivision and was representative of other systems. The conventional OWTS simulated served a four-person, three bedroom, 176 m^2 home on a 45 by 90 m lot. The system was composed of two septic tanks draining into six drainage trenches. The trenches were 22.86 m long, 91.44 cm wide, and 45.72 cm deep with 1.52 m between trenches. Each trench had a 10.16-cm-diameter PVC drainage pipe surrounded by washed gravel (1–6 cm diameter) and backfilled with native soil to ground surface as shown in Fig. 1. The total absorptive area of this designed system met TCEQ requirements for type II and type III soils, into which the soil of the study site (detailed below) fits. The simulated mound system used the same design for the drainage trenches but added 61.00 cm of soil above the surface for the mound. The simulated mounded OWTS and the ATU system with spray distribution were based on design specifications from Texas A&M AgriLife manuals (Lesikar and Weynand 2002; Lesikar 1999b, 2008) as shown in Fig. 1. The simulated ATU system used the same cross section as the conventional system in order to have comparable simulation geometries but did not include the drainage trench or drainage pipe because spray distribution is applied to the surface (Fig. 1).

The study site was located on Mocarey soils based on Natural Resources Conservation Service (NRCS) data with a water table depth of 61 cm (NRCS 2011). The Mocarey soil consisted of four layers: 28.00 cm loam, 28.00 cm clay loam, 76.00 cm loam, and 20.00 cm clay loam from the surface down. Characteristics of these layers are listed in Table 1.

2.2 Transport Modeling

The HYDRUS-2D model (Simunek et al. 2011) was used to simulate OWTS performance in the watershed under varying climatic events, system designs, and soil structures. HYDRUS is a finite element model used to simulate subsurface flow of water and solutes and was selected due to the model's ability to simulate complex soil processes including variably saturated flow in two dimensions. Since HYDRUS does not require steady state flow conditions, rapid changes in soil moisture associated with varying effluent discharges from OWTSs and rainfall events can be simulated. In addition, HYDRUS has already been widely used for solute transport in variably saturated media (Pang and Šimůnek 2006) including the modeling of OWTSs (Beach and McCray 2003; Beal et al. 2008; Pang et al. 2006) and *E. coli* transport (Bradford et al. 2006; Foppen et al. 2007; Pang et al. 2004).

Water transport in HYDRUS-2D is based on the Richards equation for saturated-unsaturated flow. Solute transport in HYDRUS-2D is based on the advection-dispersion equation based on Fick's law. Interactions between solid and liquid phases are described by non-linear non-equilibrium equations. HYDRUS-2D has three options for solute transport: equilibrium, chemical non-equilibrium, and physical non-equilibrium. Simulations for this project used chemical non-equilibrium transport considering attachment-detachment processes.

The OWTSs in the watershed were simulated using soil and operation parameters representative of that area. Weather inputs to the model were representative of a typical year as explained in the next section. Initial conditions for water movement through the soil profile were based on pressure heads. Initial pressure heads in the soil were based on the distance to the initial water table depth. Initial *E. coli* concentrations were set to zero. Observation nodes in the model domain for monitoring pressure head, water content, and *E. coli*

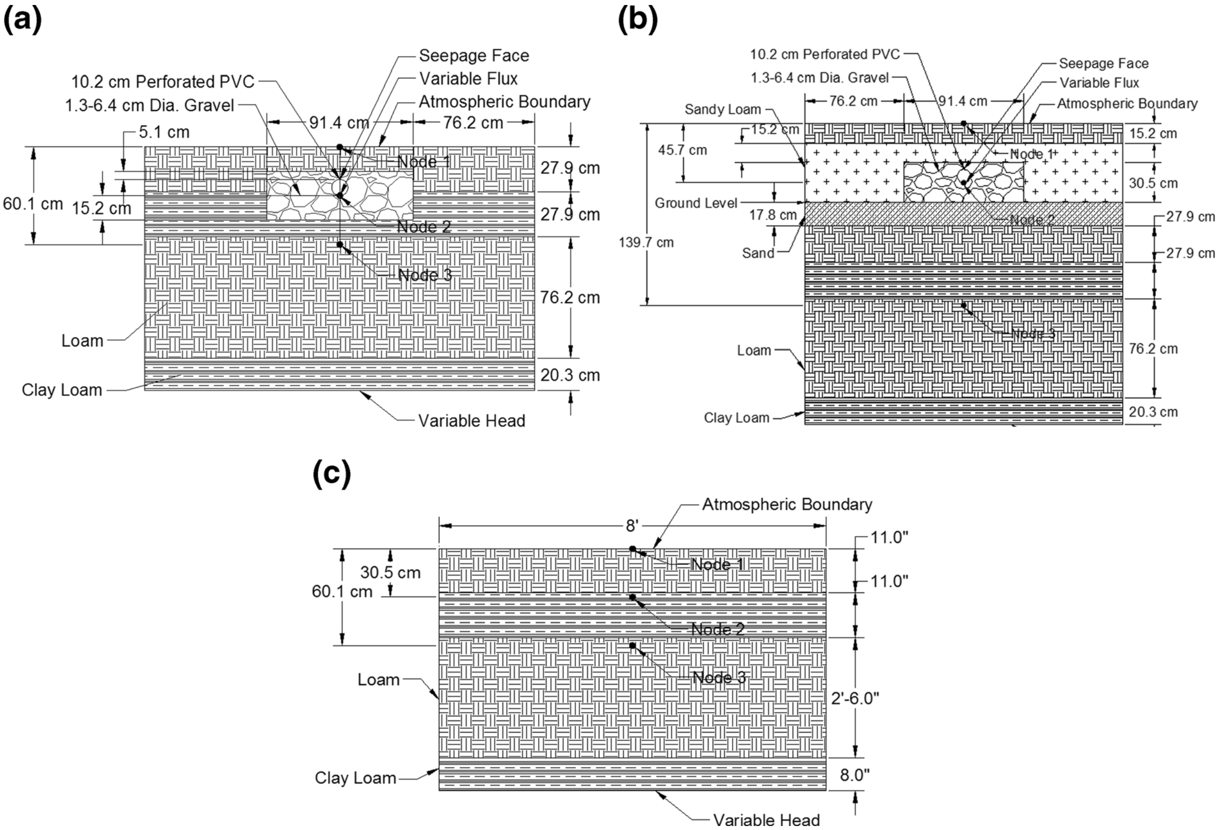

Fig. 1 Cross-sectional front views of simulated OWTS profiles and observation nodes. The conventional (**a**) and mound (**b**) systems are centered on the drainage pipe and extend on either side to the midpoints between adjacent drainage pipes. The ATU system (**c**) uses the same drainage field dimensions

concentration were placed on the surface (node 1), below the drainage pipe (node 2), and at the depth of the initial water table (node 3) of each soil profile, shown in Fig. 1.

2.3 Water Transport Simulations

Water fluxes into and out of the simulated system included wastewater flow, precipitation, and evapotranspiration. Precipitation and evapotranspiration are represented by an atmospheric boundary layer on the top surface. The atmospheric boundary for the ATU system added waste loadings to precipitation, and boundary concentrations were adjusted for dilution. Local data for daily rainfall and historical average monthly evapotranspiration values were used for simulations (Harris County Flood Warning System 2012; National Weather Service 2012; Texas AgriLife Extension 2005). Four years (2008–2011) of rainfall data from weather stations near the DBW were used to create a representative year of rainfall. One year's distribution was chosen and volume adjusted to create an average total, shown in Fig. 2. A variable flux was specified for the bottom third (cross section) of the PVC drain pipe using daily wastewater flow values. The upper two thirds (cross section) of the drainage pipe were set as a seepage face. This allowed HYDRUS to simulate flow into the drainage pipe from surrounding soil as it became saturated. The model simulated a central drain line and the soil domain extended halfway between the two adjacent drain lines. Therefore, symmetrical water flow was assumed on each side of the drain line, and a no flux boundary condition was used on the vertical sides of the soil profile. The model was run for 1 year to reach steady state and provided initial conditions prior to the study period. These initial conditions were input for variable head at bottom boundary.

Table 1 Soil and *E. coli* transport properties for soil, gravel, and fabric used in the HYDRUS simulations of the OWTSs in the watershed

Soil	Θ_r [cm/cm]	Θ_s [cm/cm]	Alpha	n	K_s [cm/h]	ρ [g/cm^3][e]	α_L [cm][f]	α_T [cm][f]	k_a [CFU/day][f]	k_d [CFU/day][f]	S_{max} [no./g][f]
Loam	0.078[a]	0.430[a]	0.036[a]	1.56[a]	1.04[a]	1.70	0.486	0.049	6.68	0.461	1000
Clay loam	0.095[a]	0.410[a]	0.019[a]	1.31[a]	0.260[a]	1.65	0.486	0.049	6.68	0.461	1000
Gravel	0.045	0.430	0.145	2.68	114[b]	2.00	0.486	0.049	0.000	0.000	10.0
Geotext	0.009[c]	0.224[c]	0.008[c]	1.92[c]	0.648[d]	2.00	0.486	0.049	0.000	0.000	10.0
Sandy loam	0.065[a]	0.410[a]	0.075[a]	1.89[a]	4.42[a]	1.75	0.486	0.049	6.68	0.461	1000
Sand	0.045[a]	0.430[a]	0.145[a]	2.68[a]	29.7[a]	1.80	0.486	0.049	6.68	0.461	1000

θ_r residual water content, θ_s saturated water content, *Alpha and n* soil constants, K_s saturated hydraulic conductivity, ρ density, α_L longitudinal dispersivity, α_T transverse dispersivity, k_a attachment coefficient, k_d detachment coefficient, S_{max} maximum amount of solute per site

[a] Schaap et al. 2001

[b] Brassington 1988

[c] Morris 2000

[d] Williams and Anwar Abouzakhm 1989

[e] NRCS 1996

[f] Bradford et al. 2006

Wastewater flow was assumed to be 0.265 m^3 (70 gal) per person per day in residential dwellings as per recommendation of the Texas Coastal Watershed Program. Another assumption made was that the flow in the drainage pipe would be on average one third full. Using these assumptions, wastewater flow into the conventional and mound systems was 72.6 mm/day of pipe length out of the bottom third along the full length of each of the six drainage pipes typically used in conventional systems. To simulate the timing of the outflow across the day, the wastewater distribution was based on a University of Wisconsin study (University of Wisconsin-Madison 1978).

The NSF International/American National Standards Code 40/245 requires that ATU systems are evaluated by a spray distribution of ATU effluent distributed throughout the day per the following percentage of total daily waste: 6:00–9:00, 35 %; 11:00–14:00, 25 %; and 17:00–20:00, 40 % (NSF International 2000a). The maximum ATU system dispersal rate for the DBW of 1.630 mm/day (Lesikar 1999b) and was distributed according to the NSF/ANSI standard as (1) 0.140 mm/

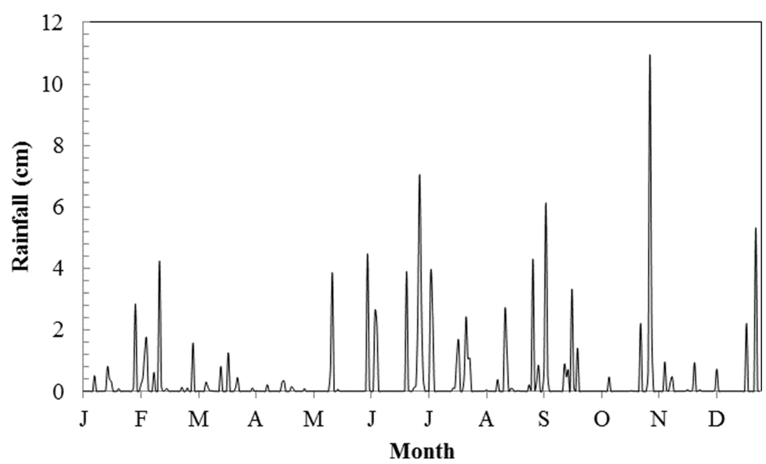

Fig. 2 Average daily rainfall in the watershed based on 4 years of data from nearby weather stations

h each hour from 6:00–9:00, (2) 0.100 mm/h from 11:00–14:00, and (3) 0.160 mm/h from 17:00–20:00.

Loam and clay loam soil was simulated in all systems as well as sand and sandy loam in the mound system. Other materials used in the simulation were geotextile fabric and gravel. The soil, gravel, and fabric properties used in the HYDRUS simulations are listed in Table 1. The soil properties include residual water content (θ_r), saturated water content (θ_s), soil constants α and n, and saturated hydraulic conductivity (K_s). The constants α and n are empirical coefficients in the soil water retention function for the van Genuchten equation. These values for soil were from Schaap et al. (2001). Values used for the van Genuchten equation were taken from Carsel and Parrish (1988), and values for the geotextile fabric were from Morris (2000) and Williams and Anwar Abouzakhm (1989). The saturated hydraulic conductivity value for gravel was from Brassington (1988).

2.4 Solute Transport Simulations

E. coli was modeled as a solute in the system. Once in the soil, *E. coli* concentrations are affected by growth, die-off, and soil attachment and detachment. Microbial growth was not considered in the modeling study because replication processes are unlikely for *E. coli* in temperatures below 30 °C (Gallagher et al. 2012; Havelaar et al. 1991; Padia et al. 2012). Transport parameters are presented in Table 1. Soil bulk density values were obtained from the NRCS Bulk Density Fact Sheet (NRCS 1996). A study by Bradford et al. (2006) using sand columns found the following values based on 710 μm sized sand: longitudinal dispersivity (α_L) 0.486 cm, attachment (k_a) 6.6816 CFU/day, detachment (k_d) 0.4608 CFU/day, and maximum amount of solute per site ($S_{\max}$) 1000/g. The value for transverse dispersivity (α_T) is assumed to be one tenth of the corresponding longitudinal dispersivity (Pang and Šimůnek 2006). These values were assumed to be equal for all soil types except for the gravel and geotextile fabric. For these materials, k_a and k_d were set to zero while $S_{\max}$ was assumed to be 10. These values were used based on the assumption that *E. coli* would travel freely throughout the gravel drainage trench due to high concentrations, low attachment, and limited surface area. A diffusion coefficient of 0.415 cm^2/day was used in the simulations (Budrene and Berg 1991). *E. coli* decay coefficients in liquid and solid phases were set to 0.193 and 3.53 CFU/day, respectively (Pang et al. 2004). The concentration of *E. coli* from septic tank effluent was found to be 1.2×10^6 CFU/100 mL (Pang et al. 2004) and was used for the concentration in the conventional and mound system. A fully effective ATU system is designed to release at most 200 CFU of *E. coli*/100 mL in its effluent (NSF International 2000b). This concentration of *E. coli* is assumed to be reduced by surface soil sorption in order to enter water bodies below regulatory standards. However, when ATUs are not properly maintained, untreated effluent may be surface applied, resulting in *E. coli* concentrations equal to those in the conventional and mound system effluent (1.2×10^6 CFU/100 mL). In this study, solute transport simulations for non-maintained (hereon referred to as "malfunctioning") ATUs were compared to conventional and mound systems.

3 Results and Discussion

The model simulation results from each system were assessed under the same initial conditions as described in Section 2.3. Descriptive statistics of pressure head (nodes 1–3) and *E. coli* concentration results (nodes 1 and 3) for each OWTS are shown in Table 2. The STATA statistical software package (StataCorp 2011) was used to run the Kruskal-Wallis (K-W) rank sum analysis of variance for non-parametric data followed by the Mann-Whitney post hoc test with the Bonferroni correction to compare the three systems.

3.1 Water Flow

Water movement through the soil profile was assessed as a function of pressure head values from the HYDRUS simulations. Negative pressure heads represent unsaturated soils, positive pressure heads represent saturated soils, and a pressure head equal to zero is where the water table is located. A significant effect of system type was found at each node using K-W: node 1 ($\chi^2 = 588$, $p < 0.001$), node 2 ($\chi^2 = 540$, $p < 0.001$), and node 3 ($\chi^2 = 1087$, $p < 0.001$). Generally, the ATU system had the highest peak pressure head values (saturation zones highest in soil profile) and was most affected by rainfall events, the conventional system followed, and the mound system had the lowest values and was least affected by rainfall. These results are affected by point of application relative to ground surface (ATU discharge being on the surface) and distance to water table (mound having a further application from water table and below the surface).

Table 2 Descriptive statistics of pressure head and *E. coli* concentration results for each node in each OWTS

Node	System	Observations	Mean	St. dev.	Min	Median	Max
Pressure head (cm)							
1	Conventional	665	−211	325	-2.22×10^3	−79.21	−5.26
	Mound	596	-5.56×10^3	4.84×10^3	-1.00×10^4	-1.00×10^4	−5.28
	ATU	994	−266	1.18×10^3	-1.00×10^4	−59.2	−0.04
2	Conventional	747	−14.9	3.06	−18.9	−14.9	6.59
	Mound	596	−14.0	1.80	−18.9	−13.62	−11.42
	ATU	994	−16.9	5.96	−24.6	−18.28	23.49
3	Conventional	747	6.38	4.37	0.00	4.82	29.37
	Mound	596	7.25	3.11	0.00	6.24	31.47
	ATU	994	1.63	4.59	−2.58	0.13	33.17
Concentration (CFU/100 mL)							
1	Conventional	664	1131.43	986.94	−735	1115	3400
	Mound	596	0.02	0.02	−0.02	0.02	0.06
	ATU	994	1.60×10^5	2.00×10^5	0.00	83,850	1.20×10^5
3	Conventional	747	15.57	14.96	−2.96	14.4	195
	Mound	595	0.00	0.00	0.00	0.00	0.00
	ATU	994	0.00	0.00	0.00	0.00	0.00

The median pressure head at node 1 for the ATU system was greater than the conventional and mound systems ($Z = 12.0$, $p < 0.001$; $Z = 22.3$, $p < 0.001$, respectively). The median pressure head for the conventional system was greater than the mound system ($Z = 16.3$, $p < 0.001$). Node 1 on the surface of each system showed that neither the mound system nor the conventional system resulted in saturated conditions at the surface. However, the ATU system did produce saturated conditions throughout the soil profile and at the surface. The ATU system had daily water distributions to the surface, minimizing the impact of evapotranspiration and decreasing storage capacity between rainfall events, making saturation more common (Fig. 3).

The median pressure head at node 2 of the mound system was greater than the conventional and ATU systems ($Z = 12.3$, $p < 0.001$; $Z = 19.3$, $p < 0.001$, respectively). The median pressure head of the conventional system was greater than the ATU system ($Z = 17.2$, $p < 0.001$). Although the pressure head at node 2 for the mound system was higher than the other systems, it never became positive, indicating that saturated conditions never existed between below the drainage pipe and the water table (Fig. 3). The pressure head for the conventional system had several peaks in positive pressure head values indicating that the water table has risen to the level of the drainage pipe. Under this condition, effluent is flowing directly into the groundwater without treatment and the drainage field is saturated.

The median pressure head at the initial depth of the water table (node 3) was greater in the mound system than in the conventional and ATU systems ($Z = 14.2$, $p < 0.001$; $Z = 26.9$, $p < 0.001$, respectively). The median pressure head for the conventional system was greater than the ATU system ($Z = 26.3$, $p < 0.001$). At node 3, the initial depth of the water table, the pressure heads for the conventional system fluctuate often where the mound system displays a consistent water table level around 6 cm above node 3 and only rises above 10 cm from node 3 in five occurrences (Fig. 3). Pressure head values in the conventional system were below those of the mound system for much of the year but became much higher than those of the mound system during peak times with only one exception during October (Fig. 3). An explanation for this observance is that evapotranspiration has a greater effect on the water table in the conventional system due to the water table in the conventional system being closer to the surface. The ATU system has similar but more dramatic fluctuations in the water table when compared to the conventional system.

The simulation results indicate that the mound system lessens the impact of rainfall on the drainage field.

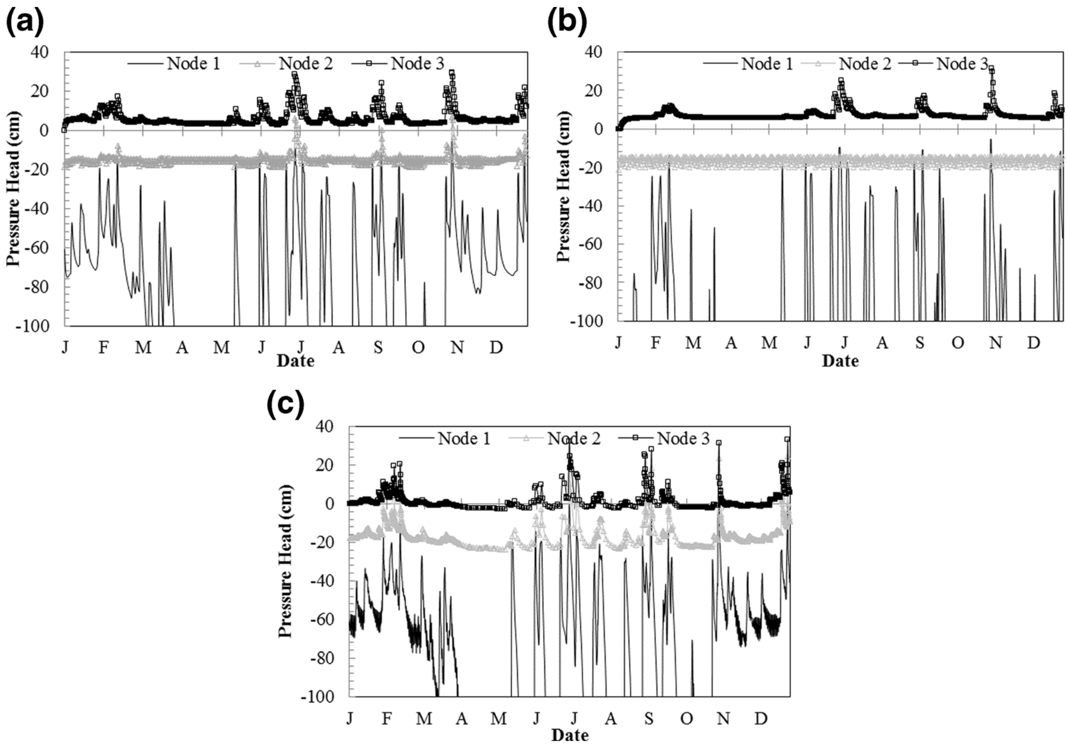

Fig. 3 Pressure head values measured in centimeters at each node in the **a** conventional system, **b** mound system, and **c** ATU system

During non-peak times in the mounded system, the pressure heads at nodes 2 and 3 are close to those of the conventional system. However, during rainfall events, the pressure head increases in the conventional and ATU system are much more dramatic than those for the mound system (Fig. 3). In contrast, the mound system often experiences no peaks due to rainfall as shown at node 2 in Fig. 3. The pressure head remains fairly constant at node 2 for the mound system ($\sigma^2 = 3.24$) but varies greatly in the conventional system ($\sigma^2 = 9.36$). Results at this node also show that the water table rises up to the drainage pipe of the conventional system. Under this condition, soil treatment of the effluent is severely decreased. System saturation reduces attachment capabilities of the soil and decreases retention times. With the mound system, saturated conditions never come near the drainage pipe.

3.2 *E. coli* Transport in Soil Profile with Different OWTSs

A significant effect of system type was found on simulated *E. coli* concentrations at the soil surface ($\chi^2 = 1598$, $p < 0.001$) and at the initial depth of the water table ($\chi^2 = 1735$, $p < 0.001$). At the soil surface, the median *E. coli* concentration was greater in the malfunctioning ATU system than in the conventional and mound systems ($Z = 30.9$, $p < 0.001$; $Z = 31.6$, $p < 0.001$, respectively). Median *E. coli* concentration was higher in the conventional system than mound system ($Z = 23.5$, $p < 0.001$). Only the conventional system allowed *E. coli* to reach the initial depth of the water table, and the median *E. coli* concentration was greater in the conventional system than in the mound and ATU systems ($Z = 28.9$, $p < 0.001$; $Z = 32.8$, $p < 0.001$, respectively). In the solute transport simulations, *E. coli* concentrations reached the soil surface and the water table in the conventional system (Fig. 4). Both are unacceptable situations for water quality. The worst case scenario for groundwater contamination occurs when the water table rises to the outlet (Fig. 4). In this case, the untreated *E. coli* concentrations in the effluent are discharged directly into the ground water. *E. coli* concentrations reaching the surface are also a major concern for surface runoff. The

Dickinson Bayou is classified as impaired by *E. coli*, and OWTSs that allow *E. coli* to reach the surface have the potential to contribute to that impairment by facilitating pathogen transport in surface runoff. The simulated mound system does not allow *E. coli* to reach either the surface or the water table depth (Fig. 4). With a functional ATU, the surface concentration of *E. coli* is minimal and does not go beyond the top loam surface (Fig. 4).

The water table never rises to the surface of the conventional system, but simulated concentrations of *E. coli* are still found at the surface. This might be due to *E. coli* transported to the surface in conventional treatment systems in unsaturated flow induced by evaporation at the surface. In addition, dispersion may also play a role in *E. coli* transport to the surface. Saturated soil conditions with *E. coli* concentrations from 50 to 100 count/100 mL in the conventional system come within 15 cm of the surface. The upward hydraulic gradient induced by evaporation along with dispersion may account for *E. coli* at the surface of the soil profile as has been seen with other solutes (Mohamed et al. 2000; Nakagawa et al. 2010; Öztürk and Özkan 2004).

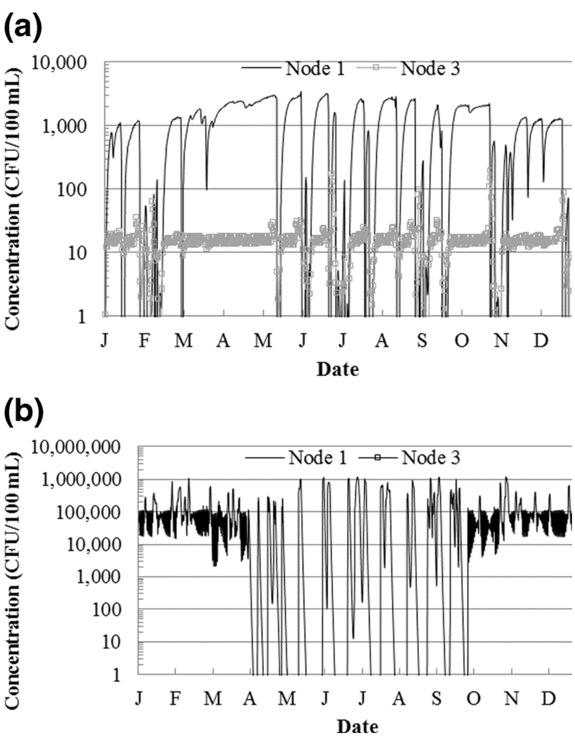

Fig. 4 HYDRUS simulation results for solute transport of *E. coli* in OWTSs in the watershed for **a** the conventional system and **b** the ATU system (mound not shown because concentration remained <0.10 CFU/100 mL throughout simulation). Improper maintenance of the ATU and untreated effluent surface application was assumed

3.3 Simulated Surface Runoff in the Watershed with OWTSs

Runoff results from the HYDRUS simulations are displayed in Table 3. Results are given for the total runoff created by each OWTS in 1 year and the peak *E. coli* concentration at the surface for each simulation. Runoff was generated once in the conventional system and three times in the ATU system as shown in Table 3. For reference, total wastewater flow was 1,060,000 cm^3 each day. These occurrences were during the peak rainfall events of January, August, and October with the one occurrence for the conventional system in October. HYDRUS simulations indicated that the system can have a negative pressure head at the surface and still generate runoff. The pressure head at node 1 of the conventional system reached a maximum −6.3 cm during the runoff generating period. The maximum pressure heads at node 1 of the ATU system during runoff generating periods were −0.04, −7.97, and −0.07 cm.

The effect of soil type on runoff was evaluated through simulation results of the conventional system with three soil profiles (Table 3). A soil profile with all loam generated no runoff while the initial profile and profile with clay instead of clay loam generated 62,303 and 95,297 cm^3 of runoff, respectively, over the 4050 m^2 surface area (Table 3). Increased clay content increased runoff. The slow infiltration rates through the clay soil keep water content values above the clay layer higher and increased runoff potential. The amount of runoff was much greater for the ATU system than for the conventional system (1,019,375 to 62,303 cm^3). The runoff event on 10/31 contributed to 80 % of total runoff (815,500 cm^3) with the ATU which was significantly greater than events on the same day with other systems. As shown in Table 3, ATU systems generate more runoff than conventional systems while the mound system had no runoff. The increased distance to the water table in the mound system provided more pore volume for water

Table 3 HYDRUS simulation results for surface runoff and *E. coli* concentrations in the runoff from OWTSs in the DBW

System	Runoff events [month/day]	Total runoff per OWTS [cm^3]	Peak *E. coli* concentration at surface [CFU/100 mL]
Conventional—loam	10/31	0	3430
Conventional	10/31	62,303	3400
Conventional—clay	10/31	95,297	2840
Malfunctioning ATU	6/19, 9/5, 10/31	1,019,375	1,200,000
Mound	–	0	0

absorption. This allowed the water content in the soil layers close to the surface to be much lower between storm periods. The increase in storage capacity prevented runoff from ever occurring in all simulations of the mound system. The conventional system had less runoff than the ATU because the wastewater application was below surface which prevented surface saturation from occurring as quickly as with the ATU system.

However, runoff alone is not the major concern. *E. coli* concentrations in the runoff are the major concern. Results for the conventional system show that, with original soil conditions, the maximum concentration at the surface was 3400 CFU/100 mL (Table 3). The runoff from an effectively operating ATU system is of no concern since pathogens are eliminated by the chlorine treatment process. However, a malfunctioning ATU system has the potential to contribute more to *E. coli* pollution than conventional systems. The level of *E. coli* concentration in runoff from an ineffective ATU system (1,019,375 cm^3 runoff and 1,200,000 CFU/100 mL) would be much higher than *E. coli* in runoff from a conventional system, as shown in Table 3. The simulated mound systems, however, did not release *E. coli* because no runoff was generated and *E. coli* was not observed at the surface (Table 3).

4 Conclusion

Results from the simulations of conventional septic systems with soil absorption fields in the DBW showed four hydraulic failures and many treatment failures where simulated *E. coli* could be released to the water table or surface runoff. Three hydraulic failures were due to the water table rising to the drainage pipe during peak rainfall events in January, August, and October and one from runoff generation in the same October rainfall event. Maximum pressure head values at node 1 during runoff-generating events lead to the conclusion that during these periods, the surface was near saturation so that the infiltration capacity was exceeded by rainfall intensity and resulted in runoff. The high water table and clay content were the largest contributors to hydraulic failure. Treatment failure in the simulated systems occurred when *E. coli* reached the surface and/or the initial depth of the water table. Runoff from conventional systems is of concern to Dickinson Bayou since it is classified as impaired, and one possible reason is runoff from conventional OWTSs. This assumption appears to have some legitimacy from observations in this research.

ATUs with spray distribution prevent contamination of both ground and surface waters when the system is used and maintained properly. However, when these systems are not properly maintained, the model results indicate that contamination from ATUs to surface waters would be greater than that of the conventional systems. Due to surface applications of effluent, more runoff is generated in ATU systems than conventional systems. In addition to greater runoff, there is a much higher *E. coli* concentration applied to the surface from improperly maintained ATU systems. This finding indicated that ATUs are more likely to contaminate surface waters with *E. coli* than conventional systems.

Model simulations of mound systems showed no *E. coli* transport to either surface runoff or water table. The increased amount of soil between the drain field and water table allows for increased removal of *E. coli* before reaching the water table and also prevents the drainage pipe from becoming saturated. *E. coli* did not reach the surface or the initial depth of the water table in the simulated system. The model results display that a mound system will improve hydraulic conditions in the soil profile while the ATU system often caused saturated soil condition at or near the soil surface. Problems associated with the high water table and clayey soils are mitigated by the added soil layers in the mound system. Mound systems may provide a more suitable alternative to conventional systems than ATUs as they proved to be more effective in this simulation study and have less maintenance requirements than ATUs.

This project demonstrates the feasibility of OWTSs by successfully comparing three types of OWTSs in the Dickinson Bayou watershed under existing conditions. Comparing the results of the different OWTSs using uniform conditions shows that mound systems are the best OWTS option in the Dickinson Bayou watershed. Mound systems operated more effectively than conventional systems and present a suitable alternative that requires less oversite and potentially more long-term proper operation than an ATU system.

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