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Numerical assessment of a soil moisture controlled wastewater SDI disposal system in Alabama Black Belt Prairie

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HIGHLIGHTS

- The system prevents hydraulic dosing during unfavorable drain field conditions.
- Bacterial growth around the drip emitters is crucial for wastewater treatment.
- Organic carbon and heterotrophs profiles mismatch the demand of denitrification.
- System performance is sensitive to soil permeability and feedback sensor position.

ABSTRACT

To promote the environmental sustainability of rural sanitation, a soil moisture controlled wastewater subsurface drip irrigation (SDI) dispersal system was field tested in the Black Belt Prairie of Alabama, USA. The soil moisture control strategy was designed to regulate wastewater disposal timing according to drain field conditions to prevent hydraulic overloading and corresponding environmental hazard. CW2D/HYDRUS simulation modeling was utilized to explore difficult-to-measure aspects of system performance. While the control system successfully adapted hydraulic loading rate to changing drain field conditions, saturated field conditions during the dormant season presented practical application challenges. The paired field experiment and simulation model demonstrate that soil biofilm growth was stimulated in the vicinity of drip emitters. Although biofilm growth is critical in maintaining adequate COD and NH₄⁺ – N removal efficiencies, the efficient removal of biodegradable COD itself by soil biofilm limits denitrification of formed NO₃⁻ – N. Furthermore, stimulated soil biofilm growth can create soil clogging around drip emitters, which was discerned in the field experiment along with salt accumulation, both of which were verified by simulation. Comparable modeling of system performance in sand and clay media demonstrate that the placement of soil moisture sensors within the drain field can have pronounced impacts on system hydraulic performance, depending on the soil permeability. Overall, the soil moisture control strategy tested is shown as a viable supplemental technology to promote the environmental sustainability of rural sanitation systems.

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1. Introduction

It is common sense that domestic wastewater should undergo proper treatment before discharge into the natural environment (Henze et al., 2008). The recent outbreak of viral pneumonia (COVID-19) further heightens the importance of safe disposal of household wastewater (Yeo et al., 2020). However, rural areas across the globe often lack necessary access to safe sanitation (Islam and Smith, 2019). Although the United Nations recognizes access to sanitation as a human right equal to safe drinking water (G.A. 2010, 2015), disparities, inequalities, and neglect are increasingly used to describe differences in access between rural and urban populations (Kaminsky and Javernick-Will, 2014). Society is built upon the efficient cooperation between urban and rural sectors, and thus rural sanitation deserves more targeted study with respect to its economic and environmental sustainability (Ji et al., 2018;
Kaminsky and Javernick-Will, 2014). Although the Human Rights Act does not identify a specific technology for sanitation in rural areas, natural-based onsite wastewater treatment systems (OWTS) have been widely used in rural areas of both developing and developed countries (Jantrania and Gross, 2006). The successful application of OWTS demands favorable site conditions such as good drainage and zero or rare flooding occurrence (NRCS, 1993).

The Black Belt Prairie in the Southern US, which runs through Mississippi and Alabama is approximately 310 miles long and up to 25 miles wide. This unique region showcases the geological and climate challenges inherent in many rural areas worldwide with similar limitations. Despite its historical glory before the US Civil War, the current economic and social development of the Black Belt Prairie falls behind the national standard (Wedgeuor and Brown, 2013). The dominant soils in this area are clay with slow water percolation and unique shrink-swell characteristics that are not favorable for soil-based OWTS. An early study based on regional soil survey indicates that over 50% of soils in the Alabama Black Belt are not suitable for traditional OWTS (He et al., 2011b). Unsuitable geology and soil challenges are further compounded by the sub-tropical climate that during wet winter months increases the risk of saturated soil overflow in Black Belt OWTS (Wedgeuor and Brown, 2013).

Most OWTS applications are based prescriptive design regulations that lack the flexibility to cope with large environmental variations. In contrast, adaptive design concepts are better able to cope with natural and societal variability to produce a more sustainable, site-specific design. The benefits of adaptive design have long been recognized by the irrigation industry and one particular case is soil moisture controlled irrigation scheduling (Nikkels et al., 2019). Soil moisture controlled irrigation scheduling (SMICS) holds promise to alleviate some of the geologic and climatic limitations faced by OWTS in the Black Belt Prairie and similar soils by managing hydraulic loading to disposal drain fields. Drain field soil moisture can be maintained within a relatively narrow range to limit the extent of counterproductive soil shrink-swell, which can open large fissures in the soil profile 2 m deep or more. More importantly, SMICS in a drain field can limit wastewater hydraulic disposal through an irrigation system during unfavorable drain field conditions. This responsive wastewater application system can create a cycle of alternating “flow/wet” and “drain/dry” phases that act as a passive pump to expel and draw air into the soil to facilitate both aerobic and anaerobic processes (Zhi and Ji, 2014).

Acknowledging these potential benefits, a SMICS system using subsurface drip irrigation (SDI) for wastewater dispersal infiltration was field tested in the Black Belt Prairie. The installed system proved effective as designed to prevent wastewater disposal during unfavorable (i.e., saturated) drain field conditions (He et al., 2011a, 2013a). However, several aspects of system performance were difficult to assess due to the inherent limitations of the field study. For this reason, field study data was coupled with numerical simulation to reveal some of the more complex processes within the system. In this particular type of study, simulation through theoretical simulations can be especially helpful to enhance and further expand research findings (Henze et al., 2008).

Process-based biokinetic models such as CW2D (Langergraber and Simunek, 2005) and CWM1 (Langergraber et al., 2009), modified from the International Water Association (IWA) Activated Sludge Model (ASM) framework (Henze et al., 2000), have been successfully used for simulation of constructed wetlands (CWs) (Samsó and García, 2013). CWM1 is suitable for CWs with saturated flows, while CW2D is suitable for CWs with intermittent hydraulic loading patterns (Martí et al., 2018; Pucher et al., 2017). Therefore, CW2D is an appropriate model to study the current SMICS-SDI scenario which is likewise characterized by transient variably-saturated flows. To this end, the previous field experiment is revisited by CW2D to further explore the strengths and weaknesses of SMICS-SDI system performance in terms of environmental sustainability.

2. Materials and methods

2.1. Field study

The experimental site was a Houston clay soil (Very-Fine, Smectitic, Thermic Oxyaquic Hapluderts) located at the Alabama Black Belt Research and Extension Center in Marion Junction, Dallas County, Alabama. The experimental system consisted of 60 drip tubes (Geoflow, CA) of 27 m long at a lateral spacing of 61 cm installed approximately 20–25 cm deep. Two capacitance type volumetric soil moisture sensors (Delta-T, UK) were buried at two depths (20 cm and 46 cm) at one position in the middle of the site to control the SDI wastewater dosing pump. Further details of the experiment setup, field data sampling and analysis procedures can be found in He et al. (2011a, 2013b).

2.2. Model setup

CW2D/HYDRUS (PC-Progress, Inc. version 1.06) is used to simulate the reactive wastewater flows within the soil profile. HYDRUS is used to describe variably saturated water flows and CW2D is used to simulate the accompanied reactive transport of wastewater components. The details of CW2D can be referenced in Langergraber and Simunek (2005). The simulation domain is a rectangular soil profile (61 cm wide and 100 cm deep) representing the cross-sectional space between two SDI emitters on adjacent drip laterals (Figure S1). Drip emitter wetted perimeters are represented by two 16 mm semi-circles 20 cm deep at the side boundaries. Each emitter is given a time variable flux to represent daily wastewater application. The upper boundary of the simulation is set as a time-variable atmospheric surface associated with daily ET and precipitation. The bottom boundary of the simulation is set to allow free drainage, which matched conditions at the site. The side boundaries, excluding emitters, are set to no flux. Model input includes actual daily SDI hydraulic disposal rate (mm/d, from field records), precipitation (mm/d, from field records), and daily field Penman method ET (evapotranspiration) (mm/d, FAO, 2006). COD and nitrogen concentrations of the applied synthetic wastewater are set to 266 mg COD/L (with slowly biodegradable COD: readily biodegradable COD = 1:1) and 80 mg NH₄+/N/L, respectively. The detailed make-up of the synthetic wastewater can be referenced in He et al. (2013b).

The single porosity van Genuchten-Mualem model was used as the soil hydraulic property model, and the soil hydraulic parameters were obtained by the Rosetta method embedded in HYDRUS using the field measured particle distribution over the 5 horizons of the experimental site (Table S1). The initial conditions of the CW2D/HYDRUS were set as: water content: 0.25 v/v; dissolved oxygen: 0 mg/L; readily biodegradable COD: 20 mg/L; slowly biodegradable COD: 20 mg/L; inert organic matter: 0 mg/L; NH₄+ - N: 0 mg/L; NO₃- - N: 0 mg/L; NO₂- - N: 0 mg/L; NO₃- - N: 0 mg/L; Heterotrophs: 50 mg/L; Ammonium oxidation bacteria: 50 mg/L; Nitrite oxidation bacteria: 50 mg/L. Phosphorus was not included since the soil at the experimental site was rich of phosphorus.

In addition to simulate the experimental SMICS system in the field conditions, system design alternatives were also studied by theoretical simulations. The considered design parameters are: soil moisture sensor location (Figure S3), system run time setting (On
15min, no Off time; On 15min, Off 45 min; and On 5min, Off 55min) and emitter flow rate (0.1, 0.2, and 0.3 times of the saturated hydraulic conductivity of the medium (sand or clay)). The hydraulic performance and the removals of COD, \( NH_4^+ \) – N and \( NO_3^- \) – N are comparably studied under different system control settings in both clay and sand within the simulation domain of Figure S1.

3. Results and discussion

3.1. Simulated system performance

The dynamic adaptation of the SMCIS-SDI wastewater application system (hereafter referred to as system) during the cooler dormant season significantly curtailed hydraulic application (Fig. 1). As expected, the system provided higher hydraulic application rates during warmer growing season months characterized by higher soil temperatures and relatively higher natural precipitations. Despite steadily increasing soil temperature and ET after the winter (November 2007 through February 2008), the system still required a period of months to recover desired, higher hydraulic application rates. Observations demonstrate the effectiveness of the system to regulate hydraulic application of wastewater. Although the system performs hydraulically as designed, the relatively long period of curtailed hydraulic loading during the cooler dormant season exposes a drawback in the system. The wastewater generated during the cool season will need to be stored or directed to other means of wastewater treatment. Therefore, this system appears to be inherently limited as a stand-alone sustainable wastewater technology for rural sanitation in an area such as the Alabama Black Belt Prairie.

Simulated 337-day spatio-temporal profiles of major soil bacterial and soil water chemical constituents (over the symmetry boundary of the simulation domain as indicated in Figure S1) are presented in Fig. 2. Profiles demonstrate the seasonal interaction between applied wastewater and soil microbial growth within the drain field, as follows. Stimulated soil microbial growth is significantly enhanced around drip line emitters as a result of wastewater application, which conforms to field observed levels of soil organic carbon. By the end of the year-long wastewater application, soil organic carbon had increased significantly in the upper soil layers (He et al., 2013a). Furthermore, the stimulated soil microbial growth zone is dominated by heterotrophs, and autotrophic nitrifiers (AOB and NOB) thrive outside the heterotrophs dominant

Fig. 1. The actual system hydraulic performance under the changing environment from June 2007 to June 2008.

Fig. 2. Simulated spatio-temporal profiles of biofilm composition and soil water quality over a 337-day wastewater application. (CR: readily biodegradable COD, CS: readily biodegradable COD, CI: biologically inert COD. Denitrification index \( = \frac{CR + CS}{CR + CS + CR} \times 100 \) where: \( K_{sat}^{CR} \) is the saturation coefficient for substrate CR, mg/L; \( \theta_s \) is the volumetric saturated soil water content; \( \theta_r \) is the volumetric residual soil water content).
zone. This outcome conforms to the rival growth of heterotrophs and autotrophic nitrifiers observed in reality that autotrophic nitrifiers thrive only when organic substrates are insufficient for heterotrophs (Zhang et al., 2015). Since heterotrophs are the major force for organic removal and biofilm formation (Ahmad and Husain, 2017), enhanced soil microbial growth indicates stimulated biofilm growth around drip emitters. Simulated spatial distribution of heterotrophs and autotrophic nitrifiers in Fig. 2 also explains corresponding COD removal profiles (Fig. 2E–G) as well as the predominance of nitrification in the soil profile (Fig. 2I and J).

Before soil biofilm growth is sufficiently increased to offer adequate utilization of applied COD, readily biodegradable COD (CR) is washed deep into soils. A plume of slowly biodegradable COD (CS) exists around the drip emitter since CS is hydrolyzed into CR for biological utilization. Consequently, no CS wash-off effect is observed. Over time, as soil biofilm gradually increases, CR is more adequately removed within the biofilm development zone and CR wash-off declines to zero. In contrast, the biologically inert COD (CI) produced from microbial decay has an observed wash-off effect that is intensified over time. This intensified wash-off of CI occurs because over time biofilm growth will lead to generation of CI generation. Overall, the presence and position of soil biofilm growth is demonstrated as crucial for adequate removal of biodegradable COD applied to a drain field as wastewater.

Despite the fluctuation of wastewater application, the \(\text{NH}_4^+ - N\) plume around the drip emitter gradually narrows as soil biofilm grows, indicating effective nitrification around the drip emitter-soil interface. In reality, nitrification can be well established to occur under both surface (Magalhaes et al., 2016) and subsurface wastewater applications (Pan et al., 2016). Comparable \(\text{NH}_4^+ - N\) and biodegradable COD (CR and CS) profiles indicate that the organic carbon limit on nitrification can be quickly attenuated by soil biofilm growth adjacent to the drip emitter. As a consequence, nitrification can be adequately carried out by nitrifiers within close proximity to each drip line emitter.

As soil biofilm grows, formed \(\text{NO}_3^- - N\) is not adequately removed because of low denitrification. A denitrification index that considers the impact of soil moisture content and organic carbon availability demonstrates that the main limitation for denitrification is the unavailability of readily biodegradable COD (CR) since soil moisture content is typically kept at relatively high levels to favor denitrification (Fig. 2L). Comparable to the \(\text{NO}_3^- - N\) profile (Fig. 2I), the denitrification index demonstrates that higher denitrification potentials occur during the early stage of soil biofilm development in deeper soil layers where CR is abundant. Over time, higher denitrification potentials are observed around the drip emitter where soil biofilm growth is promoted.

Substrate diffusion and bacterial stratification within a biofilm matrix can result in different organic carbon and nitrogen usage efficiencies (Pan et al., 2019). In addition, simultaneous nitrification and denitrification (SND) can occur in biofilm systems due to biofilm stratification and diffusion differences between electron donors and acceptors (Liu et al., 2020; Pan et al., 2019). The SND effect is especially manifested in biofilm systems under low DO conditions or when thick biofilms can create DO deficiencies inside the biofilm matrix (Liu et al., 2020; Pan et al., 2019; Yan et al., 2019). CW2D simulation does not include substrate infiltration and competition within the biofilm matrix. Nonetheless, the relatively high denitrification potential simulated within the soil biofilm development zone still implies that SND potential adjacent to drip emitters, where the quantity of soil microbes is significantly promoted, further highlights the benefit of soil biofilm growth.

Nevertheless, the relatively high \(\text{NO}_3^- - N\) profile mirrors the common dilemma found in constructed wetland (CW) systems where denitrification is often limited by the lack of available biodegradable carbon after \(\text{NH}_4^+ - N\) nitrification (Pan et al., 2016; Petitjean et al., 2016). Even so, this relatively high \(\text{NO}_3^- - N\) risk can be managed by strategically addition of slow-release carbon materials to facilitate denitrification (Berger et al., 2019; Fang et al., 2020).

Experimental and simulated soil water \(\text{NH}_4^+ - N\) and \(\text{NO}_3^- - N\) are illustrated in Fig. 3. In comparison to the control which was irrigated with clean water, the wastewater application noticeably enhanced soil water \(\text{NH}_4^+ - N\) and \(\text{NO}_3^- - N\). Despite early soil

Fig. 3. Simulated and field measured soil water \(\text{NH}_4^+ - N\) and \(\text{NO}_3^- - N\) at selected, uniform depths in the drain field.
biofilm development when the biological effect on $\text{NH}_4^+ - \text{N}$ and $\text{NO}_3^- - \text{N}$ was not adequately established, the simulated trend generally conforms to field observations. The main exception is field measured soil water $\text{NH}_4^+ - \text{N}$ is higher than simulated soil water $\text{NH}_4^+ - \text{N}$ at 15 cm and 30 cm depths, which is likely caused by an overestimated nitrification rate in the simulation. Even so, field measured soil water $\text{NO}_3^- - \text{N}$ demonstrates insufficient denitrification of available $\text{NO}_3^- - \text{N}$. Based on simulation, soil water $\text{NO}_3^- - \text{N}$ below drip emitters is caused mainly by a lack of denitrification, while soil water $\text{NO}_3^- - \text{N}$ above drip emitters results from upward capillary water movement. Soil water $\text{NO}_3^- - \text{N}$ accumulation above drip emitters also suggest the potential for salt accumulation which was observed in the field experiment (Figure S4 and He et al., 2013a).

Despite the beneficial effects of soil biofilm, soil biofilm growth will inevitably lead to biological clogging around drip emitters; and the signs of potential system clogging were observed during the experiment (Figure S2). Although a low or zero hydraulic dosing period might reduce clogging by rest ing the system (de Matos et al., 2018), it is unlikely that a resting period can thoroughly restore the system since biologically inert substances formed by metabolic processes will accumulate in the soils (Samsø and García, 2014). Nonetheless, engineering measures such as chemical injection and flushing are available for SDI systems to control biofilm clogging (Katz et al., 2014; Yu et al., 2010). Even so, it is worthwhile to study the dynamics of soil biofilm formation under different soil moisture control settings in order to achieve an acceptable balance between soil biofilm formation and wastewater treatment efficiency.

By consensus, the appropriate complexity of any model depends on the modeling goals (van Loosdrecht et al., 2015). Actual processes within the drain field are far more complicated than those summarized in CW2D. In recent years greenhouse gas (GHG) emissions from wastewater application drain fields have received attention (Somlai et al., 2019; Truhlar et al., 2019). Although not directly measured during the experiment nor included in the CW2D structure, GHG emission potentials (e.g. $\text{N}_2\text{O}$) are suggested by experimental observations of $\text{NH}_4^+ - \text{N}$ and $\text{NO}_3^- - \text{N}$, along with simulated COD profiles. Theoretically, CH$_4$ emissions from anaerobic processes can occur in a drain field (Somlai et al., 2019). As for $\text{N}_2\text{O}$ generation, low DO soil layers can lead to hydroxylamine oxidation by AOB and nitrifier denitrification over $\text{NO}_2^-$ by AOB (Thakur and Medhi, 2019). Limited organic substrate supplies can also result in incomplete denitrification by heterotrophs (Thakur and Medhi, 2019). Repetitive wastewater application can cause wet-dry cycles in soil layers which alternatively stimulate CH$_4$ and N$_2$O emissions, and similar observations have also been made on sludge field application sites. On sludge fields, aerobic conditions before sludge application, characterized by low CH$_4$ emissions and high N$_2$O emissions, are altered by fresh sludge applications which enhance CH$_4$ emissions and decrease N$_2$O emissions (Uggetti et al., 2012). It has also been reported that ditch fringe areas cyclically saturated and unsaturated present as hotspots for N$_2$O emissions (Wang et al., 2018). Therefore, GHG emissions are a potential focus for future study on this proposed SMCIS-SDI wastewater disposal strategy. Overall, both experiment and simulation demonstrate that wastewater application significantly promotes soil biofilm growth around drip line emitters, which in turn remove most of the applied organic carbon and $\text{NH}_4^+ - \text{N}$.

### 3.2. Simulated system design alternatives

Selected system design and hydraulic control settings that influence hydraulic performance including single emitter flow rate, system runtime setting (on/off time), and soil moisture sensor position are provided as simulated parameters in Table S2 (sand) and S3 (clay). The result shows that higher emitter flow rates, a design parameter, provide higher system hydraulic application rates. In a simulated sand profile the soil moisture sensor position (i.e., the physical proximity of the soil sensor to the emitter) has a greater impact on system hydraulic application rate than in simulated clay. In sand, under a given emitter flow rate, the hydraulic application rate is increased as the distance from the soil moisture sensor to the emitter increases, with horizontal location more influential than vertical direction. This intuitive finding is due mainly to the influence of gravity on the more highly permeable sand media. Similarly, in sand a longer system run time (on-time) increases the impact of soil moisture sensor proximity on system hydraulic application rate, but no such response is reflected in clay. This is caused by the different soil vertical and horizontal movement potentials between sand and clay.

The placement of soil moisture sensors balances the need to minimize water leaching loss and adequately hydrate soils for plant uptake. A larger horizontal distance between the soil moisture sensor and the emitter will favor wetting soils between drip laterals at the cost of greater leaching. A smaller horizontal distance between the soil moisture sensor and the drip emitter tends to reduce water leaching, but may not adequately wet all soil between drip laterals. This inverse relationship is more pronounced in soils with higher permeability, indicating that soil permeability has a profound impact on system hydraulic application rate based on quantifiable relationships between emitter flow rate, soil moisture sensor location, and system run time settings. Even so, it should be emphasized that these simulations are made without considering the influence from weather and crops, which will have noticeable impact on system performance (Roberts et al., 2009).

Simulated spatio-temporal water quality profiles during a typical 24-h period are illustrated for sand and clay in Fig. 4. For both media, readily biodegradable COD (CR) is sufficiently removed within a limited distance around the drip emitter. In the meantime, the spread of slowly biodegradable COD (CS) is larger than CR since CS needs to be converted into CR before biological utilization. Even so, the differences between CR and CS are wider in the sand than in the clay. Different from CR and CS, biologically inert COD (CI) accumulation is found above the drip emitter in sand, but below the drip emitter in clay. $\text{NH}_4^+ - \text{N}$ removal is complete within a limited distance from the drip emitter in both sand and clay. However, the apparent diffusion of $\text{NH}_4^+ - \text{N}$ is wider in sand than in clay, which is due to the good permeability of sand.

The cumulative profile of $\text{NO}_3^- - \text{N}$ is found similar to CI in both sand and clay media. The high permeability of sand favors leaching and also has the tendency to bring salts up to surface layers under high hydraulic loading rates (Roberts et al., 2009), while the low permeability of clay favors upward capillary water movement but restricts leaching. Therefore, $\text{NO}_3^- - \text{N}$ and CI accumulations are more pronounced in clay.

The accumulation of CI and $\text{NO}_3^- - \text{N}$ also suggest the possibility of salt accumulation which was noted during observations at the end of the one-year-long wastewater application (Figure S4 and He et al., 2013a). Since irrigation normally aims for water conservation, the irrigation leaching requirement to flush soil salt accumulation is often inadequately considered and implemented (Duan and Fedler, 2013). For soils with high permeability, higher hydraulic application rates applied through SDI tend to bring salts up to surface layers (Roberts et al., 2009). Likewise, the simulation demonstrates that sand has a more pronounced surface accumulation of CI and $\text{NO}_3^- - \text{N}$ than clay. Typically, a wastewater disposal
system that is designed to maximize leaching applies wastewater that contains more salts than required for agricultural irrigation. As a result, salt accumulation should be more pronounced for wastewater disposal systems.

Soil salt accumulation is common in agricultural practices, and it is influenced by weather conditions as well. Strong precipitation can leach accumulated salts from the soil but at the cost of underground water quality (Raine et al., 2007). On the other hand, a high ET can be conducive to salt accumulation (Raine et al., 2007). Therefore, salt accumulation might be only a temporal issue for areas with changing season and weather patterns. The subtropical climate of the Black Belt Prairie at times forces the system to rest during any rainy season, and system resting can alleviate biofilm clogging as well as leaching of accumulated salts from the drain field. However, this premise requires additional field study for verification.

Fig. 4. Comparable simulated spatio-temporal soil water quality profiles during a typical 24-h period for sand and clay. (Single emitter flow rate: 0.3Ks of the soil, and Ks is the soil saturated hydraulic conductivity; soil moisture sensor position: G in Figure S3; system runtime setting: 15-min on-time/45-min off-time).
4. Conclusions

In an attempt to enhance the sustainability of rural sanitation, a SMCS-SDI wastewater disposal system was implemented in an experimental high clay site to adapt system hydraulic disposal timing to drain field moisture conditions. After demonstrating system effectiveness to prevent hydraulic loading during unfavorable drain field conditions, a biokinetic process model CW2D was utilized to help evaluate the benefits and limitations of the system in terms of environmental sustainability for rural sanitation.

System performance is found similar to other soil-based on-site wastewater treatment systems in terms of nutrient management. Experimental and simulated results converged to demonstrate that soil biofilm growth was promoted around drip emitters as a natural response to wastewater application. Biofilm benefits include adequate removal efficiency of applied biodegradable COD and NH$_4^+$ - N. In short, biofilm formation is the result of wastewater application, and the removal efficiency of COD and NH$_4^+$ - N is the result of biofilm formation. Nevertheless, the spatial distribution of biodegradable COD and heterotrophs does not provide sufficient carbon to denitrify formed NO$_3^-$ - N, leading to an excess of NO$_3^-$ - N with the potential for salt accumulation, as observed in the field experiment.

Subsequent modeling of system performance in sand and clay media demonstrate that the placement of soil moisture sensors in the drain field is sensitive to soil permeability, and thus crucial for optimal system performance. With respect to hydraulic and nutrient management, the soil moisture control strategy evaluated in this study is demonstrated to be a viable technology that promotes the environment sustainability of rural sanitation systems.

Author statement

Jiajie He, is responsible for the simulation, data treatment, manuscript draft and coordinates among the other two authors. Mark Dougherty contributed in manuscript draft and proof read, and was also the PI during the field experiment of this study. Zhongbing Chen contributed in simulation.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.chemosphere.2020.128210.

References

Ahmad, I., Hussain, F.M., 2017. Biofilms in Plant and Soil Health. Wiley-Blackwell, USA. https://onlinelibrary.wiley.com/doi/book/10.1002/9781119194632.
Berger, A.W., Valencas, R., Miao, Y., Ravi, S., Mahendra, S., Mohanty, S.K., 2019. Biochar increases nitrate removal capacity of woodchip biofilters during high-intensity rainfall. Water Res. 165, 115008. https://doi.org/10.1016/j.watres.2019.115008.
de Mato, M.P., von Sperling, M., de Mato, A.T., 2018. Clogging in horizontal subsurface flow constructed wetlands: influencing factors, research methods and remediation techniques. Rev. Environ. Sci. Biotechnol. 17, 87–107. https://doi.org/10.1007/s11157-015-9458-1.
Duan, R., Fedler, C.B., 2013. Salt management for sustainable degraded water land application under changing climatic conditions. Environ. Sci. Technol. 47 (18), 10113–10114. https://doi.org/10.1021/es403619m.
Fang, D., Wu, A., Huang, S., Shen, Q., Zhang, Q., Ju, L., Ji, F., 2020. Polymer substrate reshapes the microbial assemblage and metabolic patterns within a biofilm denitrification system. Chem. Eng. J. 387, 124128. https://doi.org/10.1016/j.cej.2020.124128.
Food and Agricultural Organization (FAO), 2006. Crop Evapotranspiration-Guidelines for Computing Crop Water Requirements. FAO Irrigation and Drainage Paper 56. http://www.fao.org/home/en/.
General Assembly of the United Nations, 2010. The human right to water and sanitation, 64/292 Res. https://www.un.org/en/ga/.
General Assembly of the United Nations, 2015. The human rights to safe drinking water and sanitation. 70/169 Res. https://www.un.org/en/ga/.
He, J., Dougherty, M., Arriaga, EJ., AbdelGadir, A.H., 2013a. Impact of a real-time controlled wastewater subsurface drip disposal system on the selected chemical properties of a vertisol. Environ. Technol. 34 (9–12), 1341–1347. https://doi.org/10.1080/09593330.2012.746737.
He, J., Dougherty, M., Fulton, J.P., Wood, C.W., Shaw, J.N., Lame, C.R., 2013b. Short-term soil nutrient impact in a real-time drain field soil moisture-controlled SDI wastewater disposal system. Irrig. Sci. 31 (1), 59–67. https://doi.org/10.1007/s00271-011-0292-2.
He, J., Dougherty, M., Shaw, J., Fulton, J., Arriaga, F., 2011a. Hydraulic management of a soil moisture controlled SDI wastewater disposal system in an Alabama Black Belt soil. J. Environ. Manag. 92 (10), 2479–2485. https://doi.org/10.1016/j.jenvman.2011.05.009.
He, J., Dougherty, M., Zehner, R., Martin, G., 2011b. Assessing the status of onsite wastewater treatment systems in the Alabama Black Belt soil area. Environ. Eng. Sci. 28 (10), 693–699. https://doi.org/10.1089/ees.2011.0047.
Henze, M., Gujer, W., Minou, T., van Loosdrecht, M., 2000. Activated Sludge Models ASM1, ASM2, ASM2d and ASM3. IWA Publishing, UK. https://doi.org/10.1201/9781780404366.
Henze, M., van Loosdrecht, M.C.M., Ekama, G.A., Brdjanovic, D., 2008. Biological Wastewater Treatment: Principles, Modelling and Design. IWA Publishing, UK. https://doi.org/10.2166/wst.2008.101.
Islam, S., Smith, K.M., 2019. Interdisciplinary Collaboration for Water Diplomacy: A Principled and Pragmatic Approach. Routledge. https://www.routledge.com/.
Jantrania, A.R., Gross, M.A., 2006. Advanced Onsite Wastewater Systems Technol. CRC Press, USA.
Ji, X., Ren, J.Y., Ulgati, S., 2018. Towards urban-rural sustainable cooperation: models and policy implication. J. Clean. Prod. 213, 892–898. https://doi.org/10.1016/j.jclepro.2018.12.090.
Kaminsky, J.A., Javernick-Will, A.N., 2014. The internal social sustainability of sanitation infrastructure. Water Sci. Technol. 64 (17), 10028–10035. https://doi.org/10.2166/wst.2014.050.
Katz, S., Dosoretz, C., Chen, Y., Tarchitzky, J., 2014. Fouling formation and chemical control in drip irrigation systems using treated wastewater. Irrig. Sci. 32 (6), 459–469. https://doi.org/10.1007/s00271-014-0442-4.
Langergraber, G., Rousseau, D.P.L., García, J., Mena, J., 2009. CWM1: a general model to describe biokinetic processes in subsurface flow constructed wetlands. Water Sci. Technol. 59 (9), 1687. https://doi.org/10.2166/wst.2009.111.
Langergraber, G., Simunek, J., 2005. Modeling variably saturated water flow and multicomponent reactive transport in constructed wetlands. Vadose Zone J. 4 (4), 924–938. https://doi.org/10.2136/vzj2004.924.
Liu, T., He, X., Ji, A., Xu, J., Quan, X., You, S., 2020. Simultaneous nitrification and denitrification process using novel surface-modified suspended carriers for the treatment of real domestic wastewater. Chemosphere 247, 125831. https://doi.org/10.1016/j.chemosphere.2020.125831.
Magalhães, T.M., Tonetti, A.L., Bueno, D.A.C., Tonon, D., 2016. Nitrification process modeling in intermittent sand filter applied for wastewater treatment. Ecol. Eng. 93, 18–23. https://doi.org/10.1016/j.ecoleng.2016.05.003.
Martí, A.C., Pucher, B., Hernández-Crespo, C., Moneris, M.M., Langergraber, G., 2018. Numerical simulation of vertical flow wetlands with special emphasis on treatment performance during winter. Water Sci. Technol. 78, 2019–2026. https://doi.org/10.2166/wst.2018.479.
Natural Resources Conservation Service (NRCS), 1993. Part 620-soil interpretations rating guides. In: National Soil Survey Handbook. https://www.nrcs.usda.gov/wps/portal/nrcs/site/national/home/.
Nikkels, M.J., Kumar, S., Meinke, H., 2019. Adaptive irrigation infrastructure-linking insights from human-water interactions and adaptive pathways. Curr. Opin. Env. Sust. 40, 37–42. https://doi.org/10.1016/j.cosust.2019.09.001.
Pan, J., Yuan, F., Yu, L., Huang, L., Fei, H., Cheng, F., Zhang, Q., 2016. Performance of organics and nitrogen removal in subsurface wastewater infiltration systems by intermittent aeration and shunt distributing wastewater. Biosour. Technol. 211, 774–778. https://doi.org/10.1016/j.biortech.2016.03.113.
Pan, Y., Liu, Y., Peng, L., Ngo, H.H., Guo, W., Wei, W., Wang, D., Ni, B.J., 2019. Substrate diffusion within biofilms significantly influencing the electron competition during denitrification. Environ. Sci. Technol. 53, 261–269. https://doi.org/10.1021/acs.est.8b05476.
Petrieanu, A., Forquet, N., Boutin, C., 2016. Oxygen profile and clogging in vertical flow sand filters for on-site wastewater treatment. J. Environ. Manag. 170, 15–20. https://doi.org/10.1016/j.envman.2015.12.033.
Pucher, B., Ruiz, H., Paing, J.L., Chazarenc, F., Molle, P., Langergraber, G., 2017. Using numerical simulation of a one stage vertical flow wetland to optimize the depth of a zeolite layer. Water Sci. Technol. 75, 650–658. https://doi.org/10.2166/wst.2016.545.

Raine, S.R., Meyer, W.S., Rassam, D.W., Hutson, J.L., Cook, F.J., 2007. Soil–water and solute movement under precision irrigation: knowledge gaps for managing sustainable root zones. Irrig. Sci. 26 (1), 91–100. https://doi.org/10.1007/s00271-007-0075-y.

Roberts, T., Lazarovitch, N., Warrick, A.W., Thompson, T.L., 2009. Modeling salt accumulation with subsurface drip irrigation using HYDRUS-2D. Soil Sci. Soc. Am. J. 73 (1), 233–240. https://doi.org/10.2138/sssaj.2009.0033.

Rams/C19o, R., García, J., 2013. BIO_PORE, a mathematical model to simulate biofilm growth and water quality improvement in porous media: application and calibration for constructed wetlands. Ecol. Eng. 54, 116–127. https://doi.org/10.1016/j.ecoleng.2013.01.021.

Rams/C19o, R., García, J., 2014. The Cartridge Theory: a description of the functioning of horizontal subsurface flow constructed wetlands for wastewater treatment, based on modelling results. Sci. Total Environ. 473–474, 651–658. https://doi.org/10.1016/j.scitotenv.2013.12.070.

Somlai, C., Knappe, J., Gill, L., 2019. Spatial and temporal variation of CO2 and CH4 emissions from a septic tank soakaway. Sci. Total Environ. 679, 185–195. https://doi.org/10.1016/j.scitotenv.2019.04.449.

Thakur, I.S., Medhil, K., 2019. Nitrification and denitrification processes for mitigation of nitrous oxide from waste water treatment plants for biovalorization: challenges and opportunities. Bioresour. Technol. 282, 502–513. https://doi.org/10.1016/j.biortech.2019.03.069.

Truhlar, A.M., Ortega, K.L., Walter, M.T., 2019. Seasonal and diel variation in greenhouse gas emissions from septic system leach fields. Int. J. Environ. Sci. Technol. 16 (10), 6043–6052. https://doi.org/10.1007/s13762-019-02314-6.

Uggetti, E., García, J., Lind, S.E., Martikainen, P.J., Ferror, L., 2012. Quantification of greenhouse gas emissions from sludge treatment wetlands. Water Res. 46 (6), 1755–1762. https://doi.org/10.1016/j.watres.2011.12.049.

van Loosdrecht, M.C.M., Lopez-Vazquez, C.M., Meijer, S.C.F., Hooijmans, C.M., Brdjanovic, D., 2015. Twenty-five years of ASM1: past, present and future of wastewater treatment modelling. J. Hydron. Ecol. 17 (5), 697–718. https://doi.org/10.2166/hydro.2015.006.

Wang, S., Wang, W., Liu, L., Zhuang, L., Zhao, S., Su, Y., Li, Y., Wang, M., Wang, C., Xu, L., Zhu, C., 2018. Microbial nitrogen cycle hotspots in the plant-bed/ditch system of a constructed wetland with N2O mitigation. Environ. Sci. Technol. 25 (11), 6226–6236. https://doi.org/10.1021/acs.est.7b04925.

Wedgeworth, J.C., Brown, J., 2013. Limited access to safe drinking water and sanitation in Alabama’s Black Belt: a cross-sectional case study. Water Qual. Expos. Heal. 5 (2), 69–74. https://doi.org/10.1007/s12403-013-0088-0.

Yan, L., Liu, S., Liu, Q., Zhang, M., Liu, Y., Wen, Y., Chen, Z., Zhang, Y., Yang, Q., 2019. Improved performance of simultaneous nitrification and denitrification via nitrite in an oxygen-limited SBR by alternating the DO. Bioresour. Technol. 273, 153–162. https://doi.org/10.1016/j.biortech.2018.12.054.

Yeo, C., Kaushal, S., Yeo, D., 2020. Enteric involvement of coronaviruses: is faecal-oral transmission of SARS-CoV-2 possible? Lancet Gastroenterol. 5 (4), 335–337. https://doi.org/10.1016/S2468-1253(20)30048-0.

Yu, Y., Shihong, G., Xu, D., Jiandong, W., Ma, X., 2010. Effects of Treflan injection on winter wheat growth and root clogging of subsurface drippers. Agric. Water Manag. 97 (5), 723–730. https://doi.org/10.1016/j.agwat.2010.01.003.

Zhi, W., Ji, G., 2014. Quantitative response relationships between nitrogen transformation rates and nitrogen functional genes in a tidal flow constructed wetland under C/N ratio constraints. Water Res. 64, 32–41. https://doi.org/10.1016/j.watres.2014.06.035.