EFFECTIVENESS OF WASTEWATER LAND APPLICATION: MONITORING AND MODELING

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ABSTRACT

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Wastewater land application has been used for decades because of its low cost, energy use, and maintenance requirements, compared to a conventional wastewater treatment system. The performance of treatment depends on the hydraulic and organic wastewater loadings, soil characteristics, and soil conditions. Understanding the complexity of soil is important. The aerobic or anaerobic condition of the soil may result in nitrate leaching and metal mobilization into groundwater, respectively. Currently, design criteria are generally based on empirical relationships, which do not adequately consider site and waste-specific conditions. Because organic and hydraulic loadings are generally fixed based on production, dosing is the only operational parameter that can be adjusted to enhance treatment for site-specific conditions. In this study, an evaluation of domestic and food processing wastewaters land application systems were performed including examining their benefits, effectiveness, and techniques for modeling. Monitoring strategies at the demonstration site showed the viability of using land application to treat food processing wastewater and helps in making an operation decision. The HYDRUS Constructed Wetland 2D (CW2D) model was successfully calibrated and validated using data from laboratory experiments. The modeling results showed that most of the COD removal in a domestic wastewater land application system occurs within a 30.5 cm (1 ft) depth for a sandy loam soil. Increasing the dosing frequency was effective in slightly reducing the COD effluent concentration. An increase in nitrate removal by changing dosing frequency while providing sufficient carbon was found to be possible.

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KEY TO ABBREVIATIONS

Abbreviation	Full Word
Avg	Average
BOD	Biological Oxygen Demand
COD	Chemical Oxygen Demand
E	Model Efficiency
GHG	Greenhouse gas
HYDRUS CW2D	HYDRUS Constructed Wetland 2D
IA	Index of Agreement
MW	Monitoring Wells
RMSE	Root Mean Squared Error
Std	Standard Deviation
TN	Total Nitrogen
TP	Total Phosphorus
WWTP	Wastewater Treatment Plant

Chapter 1. Introduction

Wastewater land application has been pertinent for years due to its low cost, energy use, and maintenance requirements. In a conventional activated sludge wastewater treatment system, aeration accounts for the majority of energy usage, requiring 50-70% of the facility energy needs (Environmental Dynamics International 2012). New York State Energy Research and Development Authority (NYSERDA) stated that the electricity requirement to remove a kg of BOD_5 ranges from 2.87 to 9.04 kWh, depending on plant size (NYSERDA 2007). Land application treatment systems reduce the energy/electricity cost, which further reduces greenhouse gas (GHG) emissions associated with the energy savings.

Land application has been used for various wastewater types such as domestic and food processing. More than 60 million people in the United States depend on individual onsite or small community cluster systems to treat their wastewater (USEPA 2017a). The density of septic systems varies nationwide, but in general it is higher in the eastern states. The largest density is in Vermont, with 55% of households relying on septic systems, whereas the lowest is California, with 10% of households depending on septic systems. Septic systems are used in about 33% of new development throughout the nation and continued growth is expected. Food processing wastewater also has been land-applied for treatment since 1947 (Dennis 1953), mainly in Minnesota, Michigan, Pennsylvania, Wisconsin, and Washington (CVRWQCB 2006; Dennis 1953). In California, approximately 70% of food processing wastewater is land-applied (Beggs et al. 2007). Specifically, over 640 food processing plants are in operation in Central Valley, California resulting in the application of approximately, 70% of wastewater annually

(CVRWQCB 2006). In summary, land application systems are commonly used to treat domestic wastewater and food processing wastewater.

Factors including hydraulic and organic loadings, frequency of loading, soil type, soil depth, temperature, and soil microbial communities play a significant role in the performance of land application treatment. Understanding the complexity of soil is important. The nitrification process converts ammonia to nitrites and then nitrate under aerobic conditions. Denitrification converts nitrate to nitrogen gas under anaerobic condition if an organic carbon source is available. Complete nitrification usually occurs within the first 30 cm (12 in) of the soil depth (Beach 2001; Fischer 1999). However, complete denitrification typically does not occur in land application systems for domestic and food processing wastewaters (Heatwole and McCray 2007; Redding 2012). Therefore, nitrate is a concern since it is highly mobile and can flow into groundwater. High levels of nitrate in groundwater causes methemoglobinemia, also known as blue baby syndrome (DEQ 2015). A case study found that a potato processing facility in Grant County, Washington applying approximately 5.3 million liters of wastewater per day (1.42 million gallons per day) year-round on 9.3 km^2 (2,301 acre) resulted in nitrate contamination in groundwater. The level increased from 1 to 20 mg/L-N in 1986 (Redding 2012). The United States Environmental Protection Agency (USEPA) set the maximum contaminant level for nitrate at 10 mg/L-N in groundwater, and have been strictly enforcing it. In 2017, a winery in California received a fine of $635,000 for land-applying wastewater that resulted in high levels of nitrates into groundwater (Cuff 2017). USEPA estimated that a small percentage of most state's groundwater is contaminated with nitrate at level above 5 mg/L-N. In Delaware, it is estimated that 53% of the groundwater has nitrate concentrations above 5 mg/L-N (USEPA 2017b).

Hydraulic loading, organic loading, dosing frequency, soil type, soil depth, and temperature determine the treatment effectiveness. Design procedures are generally based on empirical relationships that prevent water surfacing, which does not adequately account for site and waste specific conditions (Conn and Siegrist 2009; Leverenz et al. 2009; Siegrist 2007). Because organic and hydraulic loadings are generally fixed based on production, dosing is the only operational parameter that can be practically adjusted to enhance treatment for site-specific conditions. However, research in this area is lacking. To determine and optimize the dosing frequency based on the treatment performance, the complexity of soil treatment must be understood. In this study, a modeling effort was conducted using the finite element software, HYDRUS Constructed Wetland 2D (HYDRUS CW2D), to examine the impact of dosing frequency on treatment performance.

1.1. Hypothesis/research question

This research first verified that land application of wastewater is effective for site-specific condition. With this verification, the following hypotheses were researched.

- Wastewater land application, compared to conventional wastewater treatment, can save cost and energy usage, consequently, reducing GHG emissions; and provide resources, such as water and nutrients, for crop production while minimizing environmental pollution.
- Modeling can effectively simulate the wastewater land application treatment system to enable estimations of treatment performance.
- Increasing dosing frequency in wastewater land application system can maximize the denitrification process.

1.2. Objective

The above hypotheses and research questions lead to the following project objectives.

- Demonstrate the effectiveness of the land application of domestic wastewater by examination of literature.
- Evaluate the effectiveness of food processing wastewater land application by comprehensively monitoring an actual installation. The monitored parameters include hydraulic and organic loadings, soil conditions (including its physical characteristics, temperature, moisture content, oxygen concentration, irrigation uniformity, frequency of standing water, and crop growth), and local subsurface water quality.

- Compare the benefits of wastewater land application to conventional wastewater treatment systems in terms of energy saving, GHG reduction associated with the energy saving, and freshwater reduction and nutrient reuse for crop production.

- Develop a simulation approach for the wastewater land treatment system using the HYDRUS Constructed Wetland 2D model and calibrate and validate using laboratory experimental data.

- Analyze multiple scenarios using the above calibrated model to correlate operational parameter to treatment performance including carbon degradation, nitrification, and denitrification.

1.3. Dissertation framework

The chapters in this dissertation are, in order, introduction, literature review, general methodology, domestic wastewater land application, food processing wastewater land application, and conclusion. Each are summarized in the subsequent paragraphs.

Chapter 2 is a literature review on the following concepts.

- Wastewater land application
- Wastewater treatment technologies
- Environmental impacts from wastewater land application
- Nitrogen process in wastewater land application
- Impact of loadings for wastewater land application treatment
- Current design criteria
- Overview of subsurface flow soil modeling

Chapter 3 focuses on domestic wastewater land applications, addressing all the hypotheses and objectives except for the 2nd one. This chapter contains, in order, the introduction, methods, results and discussion, and conclusion. First, the performance of domestic wastewater land application systems was examined and the benefits were estimated, including energy conservation and GHG reduction associated with the energy savings. HYDRUS CW2D modeling of domestic wastewater land application was then discussed. The modeling approach was developed, and then calibrated and validated using laboratory experimental data. Using the model, multiple scenarios were examined to observe the capacity of wastewater land application systems and the enhancement of treatment performance by changing operation parameters.

Included is the assessment of soil depth requirements based on hydraulic and organic loadings and the impact of dosing frequency on the denitrification process.

Chapter 4 focuses on food processing wastewater land application, and addresses all the hypotheses and objectives except the 1st one. This chapter contains, in order, an introduction, methods, results and discussion, and conclusions. A comprehensive monitoring strategy for a long-term food processing wastewater land application sites is first discussed. Monitoring included tracking hydraulic and organic loadings, observing soil condition in real time by soil sensor clusters, and analyzing groundwater quality. Evaluation of non-optimal and optimal areas at the demonstration site were performed by visual observation, soil analysis, and uniformity of irrigation pivots. This monitoring strategy helps to safely operate the wastewater land application system. Monitoring the hydraulic and organic loadings and soil condition using soil sensor clusters also helped in determining the operational strategies. Next, HYDRUS CW2D modeling of food processing wastewater is discussed.

Chapter 5 concludes the dissertation by summarizing the effectiveness of domestic and food processing wastewater land applications. Thereafter, insights and recommendations for further research are provided.

Chapter 2. Literature review

This chapter contains background information on wastewater land application, wastewater treatment technologies, wastewater characteristics, nitrogen processes in wastewater land application, impact sof loadings for wastewater land application, environmental impacts by wastewater land application, current design criteria, and an overview of subsurface flow soil modeling.

2.1. Background of wastewater land application

Land treatment systems are commonly used to treat domestic and food processing wastewater. In 1980, approximately 25% of all housing units (18 million people) in the United States, disposed of wastewater using an onsite wastewater treatment. Septic tanks with a drain field were the most common (U.S. Census Bureau 2006; USEPA 1980). In 2017, approximately 60 million people depended on onsite wastewater treatment systems (USEPA 2017a). Use of onsite wastewater treatment system is expected to increase to an estimated one-third of all new housing development (USEPA 2017a).

In addition, land application treatment systems have been utilized for many years to treat food processing wastewater, which is highly variable in volume and composition. The first sprinkler irrigation system in the United States with food processing wastewater was demonstrated in 1947 (Dennis 1953). A 1964 national survey identified 844 operating land application systems applying food processing wastewater and it is estimated that over 70% of the wastewater produced by California food processors is applied to the land for the treatment

(Beggs et al. 2007). In summary, the users for domestic and food processing wastewater land application system will increase in the foreseeable future.

2.2. Wastewater treatment technologies

In general, wastewater treatment is divided into conventional treatment and land application systems. Each technology has advantages and disadvantages.

Conventional wastewater treatment systems are complex mechanical systems that include activated sludge, aerobic lagoon, membrane treatment system, trickling filter, coagulation and flocculation, clarifier, and biological treatment (Tchbanoglous et al. 2003). These systems effectively treat the wastewater but have high capital and operation costs. Factors affecting operation costs include the size and loading of the plant, topography and geography of the site, wastewater characteristics, technologies associated with the treatment process, type of biosolids treatment, energy supply automation, and organization of the plant and management (Wendland 2005). If an activated sludge system is employed, the aeration tanks uses 50–73% of the total energy required for a typical wastewater treatment system (Bohn 1993; Environmental Dynamics International 2011). Approximately operation cost require $0.35 to treat a liter of wastewater (Balmer and Mattsson 1994; Big Fish Environmental 2010). For example, operation cost for a 1.89 million liter/day (500,000 gallon/day) wastewater treatment plant is estimated $672,000/year (Big Fish Environmental 2010). In addition, typical conventional wastewater treatment plant need to handle their biosolids. Approximately 0.94 kg (1.95 lbs) of dry solids per 3,785 liter (1,000 gallon) are produced from the primary and secondary processes (Tchbanoglous et al. 2003). A case study in New Hampshire found that 40%, 27%, 23%, and 16% of their biosolids were disposed of by land application (class A and B), landfilling, incineration (city of Manchester only), and out of state landfilling, respectively. The cost for biosolids disposal was estimated at \$75/wet ton, \$40/wet ton, \$71/wet ton, and \$77/wet ton for land application (class A and B), landfilling, incineration (city of Manchester only), and out of state landfilling, respectively (Wheeler et al. 2008). This energy requirement and other operational costs result in high reoccurring annual expenses for conventional wastewater treatment.

Wastewater land application treatment costs less, uses less energy and chemicals, and requires less maintenance, in comparison to traditional wastewater treatment. Specifically, land application typically costs 30–50% less to operate than a typical conventional wastewater treatment system (Charmley et al. 2006; Uhlman and Burgard 2001). Food processing wastewater is often irrigated on crop land to grow corn and alfalfa for animal feed, reducing the use of freshwater and nutrients. Water scarcity is a global issue and agriculture is the primary source of freshwater depletion in the United States (USDA 2016). In 2010, total irrigation water withdrawals were 435,275 million liter/day (115,000 million gallons/day), which was 38% of total freshwater withdrawals in the United States (Maupin et al. 2014). Therefore, wastewater land application system reduce the use of freshwater for crop production.

The land application of wastewater requires acceptable site conditions such as area of land availability, soil type, depth to the groundwater, and topography. Improper operation of land application systems can result in groundwater contamination. Soils that are either aerobic or anaerobic may result in nitrate leaching and metal mobilization into groundwater, respectively (Dong et al. 2017a; Julien and Safferman 2015). A balance is essential.

Selecting the best wastewater treatment technology is a complex process that requires accounting for site and waste-specific conditions. Included are parameters such as the volume and composition of wastewater, location, type of processing plant, availability of municipal treatment facility, soil type, cost, and state and local legislation (Harper et al. 1972). Both conventional wastewater treatment systems and wastewater land application systems can be effective in treating wastewater. However, conventional wastewater treatment systems are more suitable for urban area and land application system for rural areas. In regard to costs, wastewater land application system are generally more economical.

2.3. Environmental impacts from wastewater land application

Wastewater land application can damage the environment by leaching contaminants into the groundwater and/or cause run-off. Domestic wastewater land application systems are generally used in rural areas, representing one of the largest volumetric sources of effluent to groundwater (Koren and Bisesi 2003). If not properly designed and constructed, shallow, unconfined aquifers can become contaminated by nitrate, resulting in a significant public health risk (Robertson et al. 1991; Wilhelm et al. 1994). In fact, nitrate contamination (concentration in groundwater >10 mg/L) often occurs even in well-constructed and properly functioning domestic wastewater land application systems (Wilhelm et al. 1994). Nitrate contamination of groundwater has been found under drain fields in the valley soils of the northwestern United States (Ver Hey 1987). Similarly, nitrate contamination of groundwater has been documented in the South Valley of Albuquerque, New Mexico (Keleher 2008). If the nitrate is not denitrified, high levels enter in groundwater and can cause methemoglobinemia, also more commonly known as blue baby syndrome (DEQ 2015). A 1950 report listed 144 cases of infant methemoglobinemia with 14 deaths in Minnesota (Rosenfield and Huston 1950).

When the wastewater is applied to soil, soil microorganisms use the organic materials as a food source. During the process of oxidation and decomposition of organic materials, electrons are release. Oxygen is the most favorable electron acceptor (Tarradellas, Bitton, and Rossel 1997). When the oxygen is depleted, lower energy electron acceptors such as nitrate, manganese, iron, and sulfate are utilized (Haggblom and Milligan 2000; Matocha et al. 2005; Mokma 2006a). The low redox potential condition in soil may reduce metal species to be in a more mobile form (Safferman et al. 2011). Therefore, metal mobilization into groundwater is also a concern where nitrification is limited and nitrate is much less prevalent (McQuilan 2004).

2.4. Nitrogen processes in wastewater land application

Another concern regarding wastewater land application is nitrate leaching into groundwater (Cuff 2017; Redding 2012). Adriano et al. (1975) showed that 76% of total nitrogen from fruit and vegetable processing wastewater applied on the sandy loam soil leached into subsurface water (Adriano et al. 1975). Therefore, understanding the nitrogen processes in soil, as shown in Figure 1, is important to protecting the environment.

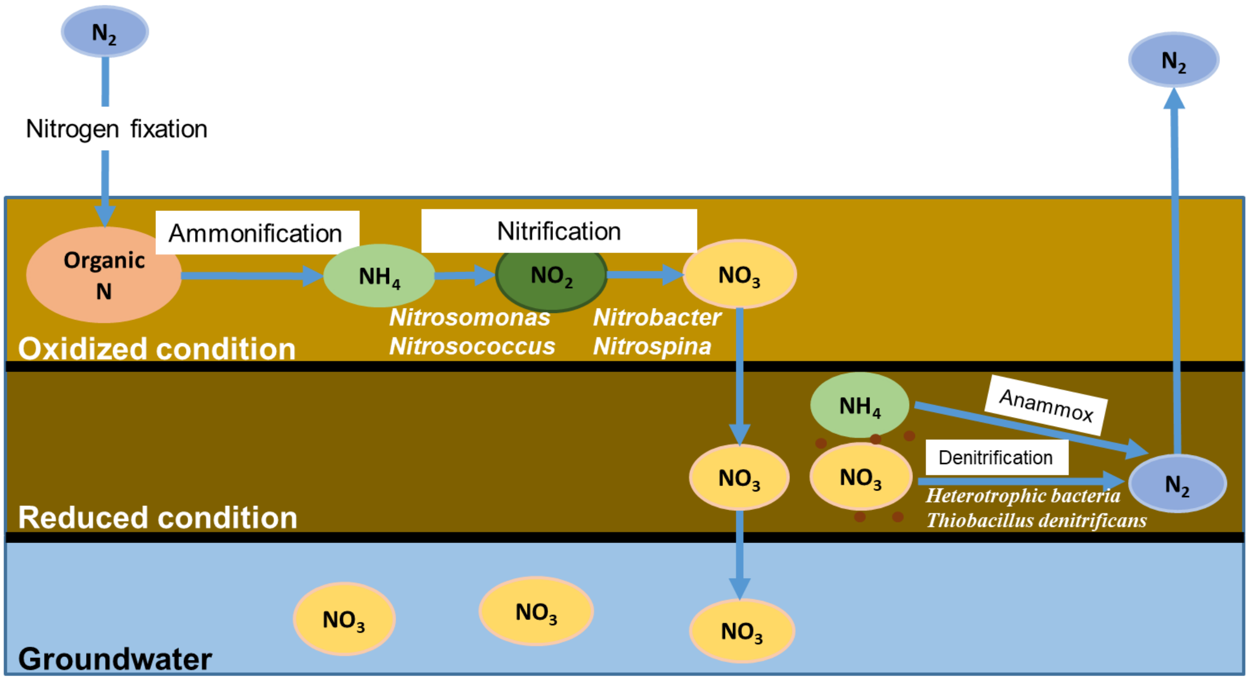


Figure 1. Nitrogen processes in soil

In soil, nitrogen is transformed by nitrogen fixation, ammonification, nitrification, denitrification, and anammox. Nitrogen fixation is the conversion of nitrogen gas to ammonium by microorganisms. Ammonification is the conversion of organic nitrogen to ammonium resulting from the decomposition of dead plant residual, animal tissue, and microbial biomass. Nitrification is the oxidation of ammonia to nitrite, and then nitrate, which is highly mobile. This is carried out by nitrifying bacteria under aerobic conditions. Nitrifying bacteria includes ammonia-oxidizing bacteria (*Nitrosomonas*, *Nitrosococcus*, and *Nitrosospira*) and nitrite-oxidizing bacteria (*Nitrobacter, Nitrospina*, and *Nitrococcus*) (Watson et al., 1981). Denitrification converts nitrate to nitrogen gas and is carried out by denitrifying bacteria such as *Heterotrophic bacteria*, *Thiobacillus denitrificans*, *micrococcus denitrificans*, *Pseudomonas*, and *Achromobacter*, under anaerobic condition (Carlson and Ingraham, 1983). Denitrification occurs when a carbon source is available for denitrification microorganisms, the soil is under anaerobic conditions, and temperatures are within an acceptable range. At greater soil depths, lower levels of oxygen are likely, which can promote denitrification. On the other hand, carbon is needed for denitrification. Typically, carbon is oxidized in the upper levels of soil that are often aerobic and, consequently, denitrification may not occur resulting in the nitrate leaching into groundwater. Anammox and anaerobic ammonia oxidation converts ammonia to nitrogen gas under anaerobic conditions. This process is driven by microorganisms such as *Candidatus Anammoxoglobu propionicus* and *Candidatus Brocadia* (Kartal et al. 2007).

2.5. Impact of loadings for wastewater land application treatment

Hydraulic and organic loadings have an important role when evaluating and designing wastewater land application systems. Typically, these loadings are not controllable at wastewater land application site. Related is the dosing frequency, which may be a critical, practical operational parameter as it can be altered without impacting the loadings. The impacts of loadings are discussed in below.

Hydraulic and organic loadings are the principal parameters in designing wastewater land application systems. Increasing the hydraulic loading increases the soil's moisture content, ultimate resulting in its porosity being potentially completely occupied by water (i.e., saturation). When the soil is saturated, oxygen cannot diffuse into its porosity resulting in anaerobic conditions (Erickson and Tyler 2000). In addition, when wastewater is applied to land, the particulate solids of the wastewater can remain near the surface, limiting oxygen transport to the soil and promoting anaerobic conditions (Beggs et al. 2007; Crites and Tchobanoglous 1998).

Excessive organic loading or long-term addition of organic loading can reduce the soil's hydraulic conductivity (McDaniel 2006). High organic loading may also enhance microbial activity because organic carbon is a substrate or food for microorganisms. Excess of microorganisms can clog the pore space in soil, which may reduce its hydraulic conductivity resulting in a lower redox potential (Hillel 2008). Furthermore, increasing hydraulic loading in a well-drained soil decreases retention time of the wastewater, which reduces the efficiency of treatment (Converse and Tyler 1998; Siegrist and Van Cuyk 2001). Converse and Tyler (1998) studied the treatment of fecal coliform concentrations in a well-drained soil with different hydraulic loadings at 40.75, 122.24, and 244.48 liters per day/m^2 (1, 3, 6 gallons per day/ft^2). Higher fecal coliform concentrations were found in effluent wastewater when the soil received 122.24, and 244.48 liters per day/m^2 (3 and 6 gallons per day/ft^2) instead of 40.75 liters per day/m^2 (1 gallon per day/ft^2) (Converse and Tyler 1998).

Dosing frequency is an operational parameter which may impact on the performance of treatment. A previous study discussed that a hydraulic resting period of 12 hours provided adequate time for a hydrodynamic "piston" effect to occur, which is when oxygen is drawn into the soil immediately after the addition of water. Doses given at a higher frequency, with less resting time, were shown to lead to anoxic conditions (Julien and Safferman 2015). Therefore, increasing dosing frequency may impact on soil reduction condition, which may promote denitrification process.

As the frequency of dosing increase, the retention time may increase, which may result in better treatment. In a sand filter treatment system, increasing the dosing frequency was found to improve the performance of treatment, but continuous heterotrophic bacterial growth was observed at the surface, which may result in clogging or premature of life of sand filter treatment system (Furman et al. 1955; Grantham et al. 1949; Leverenz et al. 2009). The optimal dosing frequency should be determined while considering both hydraulic and organic loadings to minimize clogging or premature of life.

2.6. Current design criteria

Currently, design criteria for wastewater land application systems differ by states. Siegrist (2007) discussed that hydraulic loadings for domestic wastewater land application systems are based on limited empirical evidence and vary widely from state to state (Siegrist 2007). However, most state regulations focus on a few specific wastewater disposal characteristics, the most important of which are hydraulic loading, organic loading, soil depth, and soil type. Table 1 shows the diverse design criteria for domestic wastewater land application system (Arkansas State board of health 2007; Colorado Department of public health and envrionment 2013; Michigan Department of Environmetnal Quality 2013; Nebraska Department of Environmental Quality 2007; New York State Department of Health 2016; Olivieri and Roche 1979; Oregon Department of Environmental Quality 2017; State of Kansas Department of Helath and Environment 1997; Tennessee State Government 2016). Many states do not have guideline for organic loading. States recommend between 45.72 cm (18 in) to 121.92 cm (4 ft) vertical separation between the bottom of the drain field and the water table. Regulations on wastewater hydraulic loading are even less uniform, relying on a combination of factors including hydraulic loading, soil type, organic loading, and treatment system size. Many rely on a flow rate per bedroom. Further stipulations are often imposed based on wastewater strength and soil profile.

Table 1. Design criteria for onsite wastewater land application system

State	Soil depth required for the drain field	Soil depth required between drain field and water table	Hydraulic loading	Organic loading	Reference
Arkansas	0.46 m (18 in)	0.6 m (24 in) – loamy soil, 0.9 m (36 in) – sandy soil	15 - 30 L/m^2/day (0.37 - 0.75 gal/ft^2/day) depends on percolation rate	N/A	(Arkansas State board of health 2007)
California	0.3 m (12 in)	0.91 m (36 in) – greater than 5 min/in, 6 m (240 in) – between 1 and 5 min/in, prohibited – less than 1 min/in	9 - 64 L/m^2/day (0.22 - 1.58 gal/ft^2/day) depends on percolation rate	N/A	(Olivieri and Roche 1979)
Colorado	N/A	1.2 m (48 in)	N/A	N/A	(Colorado Department of public health and envrionment 2013)
Kansas	N/A	1.2 - 1.8 m (48 - 72 in)	10 L/m^2/day (0.25 gal/ft^2/day) – sandy clay loam 16 L/m^2/day (0.4 gal/ft^2/day) – sandy loam 24 L/m^2/day (0.6 gal/ft^2/day) – loamy sand 37 L/m^2/day (0.9 gal/ft^2/day) – medium sand 45 L/m^2/day (1.1 gal/ft^2/day) – course sand	N/A	(State of Kansas Department of Helath and Environment 1997)
Maryland	0.15 to 0.3 m (6 to 12 in)	1.2 m (48 in)	Table is provided	N/A	(Maryland Deaprtment of the Environmemt 2010)

Table 1. Design criteria for onsite wastewater land application system (cont'd)

State	Soil depth required for the drain field	Soil depth required between drain field and water table	Hydraulic loading rate	Organic loading	Reference
Michigan	0.3 to 0.6 m (12 to 24 in)	0.46 m (18 in)	12 L/m2/day (0.3 gal/ft2/day) - sandy clay 24 L/m2/day (0.6 gal/ft2/day) - loam, sandy loam 40 L/m2/day (1.0 gal/ft2/day) - loamy sand 48 L/m2/day (1.2 gal/ft2/day) - fine sand 65 L/m2/day (1.6 gal/ft2/day) - coarse sand	$\frac{140 \frac{\mathrm{mg}}{\mathrm{L}} BOD_5}{Expected\ High\ Strength\ Waste\ (\frac{mg}{L} BOD_5)} * Soil\ Hydraulic\ Loading\ Rate$	(DEQ 2013)
Nebraska	N/A	1.2 m (48 in)	N/A	N/A	(Nebraska Department of Environmental Quality 2007)
New York	N/A	1.2 m (48 in)	416 - 568 L/day (110 - 150 gal/day)	N/A	(New York State Department of Health 2016)
Tennessee	N/A	1.2 m (48 in)	12 L/m2/day (0.3 gal/ft2/day) - sandy clay 24 L/m2/day (0.6 gal/ft2/day) - silt loam, loam 28 L/m2/day (0.7 gal/ft2/day) - sandy loam 40 L/m2/day (1.0 gal/ft2/day) – fine sand	> 150 mg/L BOD; 3 g/m2/day (27 lb BOD/acre/day) for clays, 4.6 g/m2/day (41 lb BOD/acre/day) for loams, 6.2 g/m2/day (55 lb BOD/acre/day) for sandy	(Tennessee State Government 2016)

The lack of design consistency is also observed for the land application of food processing wastewater. According to a literature review by Dr. Mokma's (2006), a wide range of hydraulic loadings are observed. Specifically, hydraulic loadings from 21 food processing facilities ranged from 1.96 to 140 liter/m^2/day (2,100 to 150,000 gal/acre/day) (Carawan et al. 1979). Hydraulic loading is limited by the organic loading. High organic loading can cause microorganisms to grow extensively, which can clog soils. When the soil is clogged, surface ponding or run-off may occur. Organic loading also has been roughly estimated based on empirical relationship, which result in a wide range of observations. In the state of New York, the organic loading was recommended at 56 g of BOD/m^2/day (500 lb of BOD /acre/day) (Crites et al. 2000). Spyridakis and Welch (1976) stated that organic loading from two food processing plants were 52 and 84 g of BOD /m^2/day (460 and 750 lb of BOD /acre/day) (Spyridakis and Welch 1976). Crarawan et al (1979) recommended the maximum organic loading of 22 g of BOD /m^2/day (200 lb/acre/day) (Carawan et al. 1979). For Michigan sandy soils, the rough limits, which have been observed by current wastewater application, are between 0.0056 and 0.0224 kg of BOD /m^2/day (50 and 200 lb/acre/day) with a hydraulic loading less than 3.74 liters/m^2/day (4000 gal/acre/day) (Mokma 2006). This wide range of hydraulic and organic loadings indicates that more research is needed and modeling can be beneficial to determine optimal hydraulic and organic loadings while considering site and waste specific condition.

2.7. Overview of subsurface flow soil modeling

Many models have been developed to quantify water flow and pollutant movement in soils. These models have been widely used in agriculture, constructed wetland, and septic soil treatment systems. The modeling approaches can be a simple analytical approach to a complex nonlinear process. Available models include HYDRUS, LEACHM, SWAP, VS2DT, and DRAINMOD. Details about these models are discussed below.

Quantification and visualization of pollutant flow patterns can be modeled using HYDRUS Constructed Wetland 2D (CW2D) software. HYDRUS CW2D simulates the complexity of water flow in unsaturated, partially saturated, and fully saturated soil by numerically solving the Richard equation and the convection dispersion equation (Šimůnek et al. 1999). This model considers chemical and physical processes of pollutants, soil properties, rainfall, and evapotranspiration, including the aerobic and anoxic transformation and degradation process for organic matter, nitrogen, and phosphorus (Šimůnek et al., 1999). This model has been widely used to simulate and understand the transport of pesticides, nitrate, phosphorus, and heavy metals in soil (Anwar and Thien 2015; Crevoisier et al. 2008; Dao et al. 2014; Freiberger et al. 2014; Honegger 2015; Mailhol et al. 2007; Nakamura et al. 2004; Naseri et al. 2011; Nohra et al. 2012; Shekofteh et al. 2013; Šimůnek et al. 2013; Sinclair et al. 2014; Srilert et al. 2012; Twarakavi et al. 2008; Vilim et al. 2013; Wang et al. 2016).

LEACHM (Leaching Estimation and Chemistry Model) is a one-dimensional finite difference model. The model can predict water and solute movement, transformation, plant uptake, and chemical reactions in an unsaturated soil by using the various subroutines. LEACHW describes water movement, LEACHP models pesticides, LEACHN models nitrogen and phosphorus, and LEACHC models salinity in calcareous soils. The model uses the Freundlich-Langmuir isotherm for sorption and desorption (Hutson 2000). The input of soil parameters, including soil physical properties (bulk density, particle size distribution, and water retention characteristics), are required. Previous studies have used the LEACHM model to predict pesticide, herbicide, and heavy metal transport through soil, as well as soil dynamics of nitrogen and nitrate (Hutson, 1991; Jemison et al., 1994; Khakural, 1993; Wagenet, 1989; Webb and Lilburne, 2000).

SWAP (Soil-Water-Atmosphere-Plant) is a one-dimensional model that solves multiple governing equations using finite difference numerical analysis. This model is used to simulate water flow, solute movement, heat flow, macropore flow, and crop growth in soils. It is designed to simulate water and solute movement processes at a field-scale with applications during both growing seasons and long-term time series. This model has been used for field-scale water and salinity management, irrigation scheduling, modeling transient drainage conditions, plant growth impacts from water and salinity, pesticide leaching into water sources, regional drainage from topsoil to different surface water systems, optimization of surface water management, and effects of soil heterogeneity (Van Dam et al. 2008; Kroes et al. 2017). In addition, the SWAP model can predict preferential flow, adsorption, and decomposition of nutrients and pesticides (Van Dam et al. 1997).

VS2DT (Variably Saturated 2D Flow and Transport) uses the finite difference technique to approximate the flow equation, developed using a combination of the law of conservation of fluid mass with a non-linear form of Darcy's equation. This model simulates water flow and nutrient transportation in variably saturated soil conditions. The model can simulate in 1-dimension and 2-dimensions with planar or cylindrical geometries. There are multiple options for boundary conditions for flow in unsaturated soil, including infiltration with ponding, evaporation, plant transpiration, and seepage faces. Options for solute transport include first-order decay, adsorption, and ion exchange. Previous studies used this model to predict pollutant transport to tile drainage, evaluate hydraulic properties of soils for irrigation strategies, and to evaluate groundwater transport of tracers (Constantz et al. 2003; Munster et al. 1994).

DRAINMOD is a hydrological model for simulating the performance of agricultural drainage and related water management systems. The model is effective for simulating the hydrology of poorly-drained, high water table soils on both short and long-term timescales. It predicts the effects of drainage and associated water management practices on water table depths, the soil water regime, and crop yields. Infiltration, subsurface drainage, surface runoff, evapotranspiration, vertical and lateral seepage, water table depth, and water-free pore space in the soil profile are considered (Skaggs et al. 2012). The current version of DRAINMOD simulate solely in 1-dimension flow. DRAINMOD has several modules, including DRAINMOD-S (salinity), DRAINMOD-NII (nitrogen), DRAINMOD-DUFLOW (linked to DUFLOW model), and DRAINMOD-W (watershed scale) In the past, this model has mainly been used for nitrogen transport (Salazar et al. 2009; Wang et al. 2005; Youssef et al. 2005) but a recent study used it for phosphorus (Askar et al. 2016).

Comparison of HYDRUS CW2D, LEACHM, SWAP, VS2DT, and DRAINMOD-NII are shown in Table 2. All models can simulate water and solute flow in the soil and account for precipitation, evapotranspiration, plant uptake, and surface runoff. Only HYDRUS CW2D and SWAP consider macropore in the model. SWAP and DRAINMOD can provide estimated crop yield. HYDRUS CW2D was selected for this study, because it is one of the most comprehensive tools for modeling water and solute flow in soil. HYDRUS CW2D specializes in nutrient flow and it entails both aerobic and anoxic transformation and degradation processes for organic matter, nitrogen, and phosphorus. Nitrate is especially an issue in wastewater land application and HYDRUS CW2D has demonstrated capabilities to predict its fate. As a focus of this research is the impact of dosing frequency on treatment performance, it is important to note that several studies successfully used HYDRUS to observe its impact on the growth of heterotrophic bacteria, fecal coliform, and moisture content (Leverenz, Tchobanoglous, and Darby 2009; Radcliffe and West 2009; Hassan et al. 2005).

Once the HYDRUS CW2D was calibrated and validated using laboratory experimental data, multiple scenarios were run using different dosing frequencies in order to maximize the treatment performance while protecting environment. This modeling approach may allow for the determination if the operation parameter of dosing frequency can be set to achieve both the degradation of carbon and conversion of nitrate to nitrogen gas.

Table 2. Summary of HYDRUS CW2D, LEACHM, SWAP, VS2DT, and DRAINMOD-NII

Variable	**Model**				
	HYDRUS CW2D	**LEACHM**	**SWAP**	**VS2DT**	**DRAINMOD-NII**
Dimension	2D	1D	1D	2D	1D
Saturated/ Unsaturated flow	Yes	Yes	Yes	Yes	Yes
Solute flow	Yes	Yes	Yes	Yes	Yes
Hydraulic model	van Genuchten	Campbell	van Genuchten	van Genuchten	van Genuchten
Evapotranspiration	Yes	Yes	Yes	Yes	Yes
Surface runoff	Yes	Yes	Yes	Yes	Yes
Macropore flow	Yes	No	Yes	No	No
Plant uptake	Yes	Yes	Yes	Yes	Yes
Crop yield	No	No	Yes	No	Yes
Application	• Irrigation management • Tile drainage design • Drip irrigation design • Wastewater land application • Constructed wetland • Surface runoff • Nutrient transport • Seasonal simulation • Pesticide transport	• Irrigation water management • Nutrient transport • Pesticide transport • Surface runoff • Seasonal simulation	• Irrigation water management • Nutrient transport • Crop yield estimation • Surface runoff • Seasonal simulation • Snow • Freezing and thawing	• Irrigation water management • Drip irrigation design • Nutrient transport • Surface runoff • Seasonal simulation	• Irrigation water management • Surface runoff • Tile drainage design • Manure land application • Crop yield estimation • Nitrogen transport • Freezing and thawing • Seasonal simulation

Chapter 3. Domestic wastewater land application

This chapter discusses provides an introduction to domestic wastewater land application, including background, problem statement, and benefits. Then, the treatment performance of domestic wastewater land application systems are evaluated using the literature. HYDRUS CW2D modeling of domestic wastewater land application is then discussed.

3.1. Introduction

Wastewater land application has been used for many years to treat the domestic wastewater. The performance has been studied (Dong et al. 2017b; Gross 2004; Hammerlund and Glotfelty 2016; National Environmental Services Center 2013; Ronayne et al. 1982). The typical, least expensive configuration includes a 1892.7 – 3785.4 liter (500 – 1,000 gallon) septic tank and a subsurface soil distribution network. This network is referred to as drain field (USEPA 2017c), leach field (USEPA 2017c), septic soil treatment system (Dong et al. 2017b), septic field (USEPA 2017c), soil treatment unit (Wunsch et al. 2009), and soil absorption field (Lesikar 2008). In this study, domestic wastewater land application systems refer to a septic tank and drain field in series.

In comparison to a conventional wastewater treatment system, land application can save energy, consequently reducing GHG emissions. These benefits were estimated based on standard data on the production and characteristics of wastewater, population, and treatment requirements. Approximately 497 million kg (1,095 million lb) of BOD_5 per year from domestic wastewater is treated by wastewater land application systems in the United State. To remove 0.45 kg (1 lb) of BOD_5 at a traditional wastewater treatment facility where receiving less than 3.78 million liters/day (1 million gallon/day) requires 4.1 kWh of energy (NYSERDA 2007). Therefore, approximately 2,037,700 MWh electricity is saved annually. The cost for electricity was estimated at $326 million/year with an assumption 7.27 Cent/kWh. This result in a GHG reduction of 3.5 million metric tons/day, which is equivalent to GHG emission from 715,432 passenger vehicles driven 11,443 miles/year and a mileage of 22 miles/gallon and (USEPA 2016). GHG emission from phosphorus treatment in a typical wastewater treatment plant was also estimated. When 13,627 million liters (3,600 million gallons) of domestic wastewater/day is treated by onsite wastewater treatment system in the United States, approximately 29.8 million kg of phosphorus are treated annually. When 29.8 million kg of phosphorus are treated by wastewater treatment plant by a physical/chemical process, 476,800 metric tons of CO_2 and 646 metric tons of NO_x will be produced. Consequently, this amount of gases can be conserved by an onsite wastewater treatment system (Coats et al. 2011).

Although domestic wastewater land application systems have been widely used, design criteria are not fully developed and vary by state. The depth required for a soil adsorption field varies from 0.15 to 0.6 m (6 to 24 in). In addition, the depth required from the bottom of an adsorption field to the water table range from 0.46 to 6 m (18 to 240 in). Hydraulic loadings were mainly determined by soil type and many state do not have guideline for organic loadings. Siegrist (2007) discussed that allowable hydraulic loading for domestic wastewater land application systems are based on limited empirical evidence. The need for computer modeling efforts to design the treatment system is emphasized (Siegrist 2007).

For this research, HYDRUS Constructed Wetland 2D (HYDRUS CW2D), a finite element model, was selected for simulating the movement of water and multiple solutes in soil. This model was originally designed to simulate wastewater treatment in wetlands, but was also used in this research for wastewater land application. Previous studies have also used HYDRUS to simulate nutrient movement in soil (Crevoisier et al. 2008; Dao et al. 2014; Mailhol et al. 2007; Shekofteh et al. 2013; Vilim et al. 2013). This modeling approach may provide the minimum depth requirements for carbon degradation and allow for the understanding of how dosing frequency effects the conversion of nitrate to nitrogen gas. Laboratory column operation and chemical wastewater analyses were used to calibrate and validate the HYDRUS CW2D model.

This chapter discusses the effectiveness of domestic wastewater by an examination of the literature, develops a HYDRUS CW2D model, calibrates and validates the model using laboratory experiment data, and analyzes multiple scenario to observe the treatment performance including carbon degradation, nitrification, and denitrification.

3.2. Materials and methods

The method to achieve each objective for domestic wastewater land application are provided. Included is evaluation of domestic wastewater land application, the calibration and validation of HYDRUS CW2D modeling using laboratory data, and analysis of multiple scenario to provide a more accurate design approach and observe carbon degradation, nitrification, and denitrification.

3.2.1. HYDRUS CW2D modeling

HYDRUS CW2D is a finite element model for simulating two-dimensional water and solutes movement in soil. Included is the visualization of the transmission and degradation processes for organic matter and nitrogen under aerobic and anoxic conditions. In this section, the governing equation, HYDRUS CW2D component and processes, limitation of HYDRUS CW2D, laboratory experimental, calibration and validation procedures, input parameters, goodness of fit, and scenarios are discussed.

3.2.1.1. Governing equation

HYDRUS CW2D simulates the water and solute movement in two dimensions using the Richard and advection - convection dispersion equations (Šimůnek et al., 1999).

The HYDRUS model numerically solves the Richards' equation for water flow in unsaturated, partially saturated, and fully saturated soil (Šimůnek et al., 1999). The assumption was made that the air phase plays an insignificant role in the liquid flow process. The modified form of Richards' equation is described in Equation 1 (Šimůnek et al., 1999).

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x_i}\left[K\left(K_{ij}^{A}\frac{\partial h}{\partial x_j} + K_{iz}^{A}\right)\right] - S \qquad (1)$$

where θ is the volumetric water content [L^3L^{-3}], h is the pressure head [L], x_i (i=1, 2) are the spatial coordinates [L], t is the time [T], K_{ij}^A are components of a dimensionless anisotropy tensor K^A, K is the unsaturated hydraulic conductivity function [LT^{-1}], and S is a sink/source term [T^{-1}], which is considered here as the amount of water removed by plant roots.

The HYDRUS model solves the advection-dispersion equation (Šimůnek et al. 1999) for modeling transport of solute in a soil-air-water system. Equation 2 is the governing equation (Šimůnek et al., 1999)(Warrick 2002)

$$\frac{\partial(\theta c)}{\partial t} + \frac{\partial(\rho s)}{\partial t} = \frac{\partial}{\partial z}\left(\theta D \frac{\partial c}{\partial z} - qc\right) - \phi \tag{2}$$

where, c is solution concentration [ML^{-3}], s is adsorbed concentration [MM^{-1}], Θ is water content [L^3L^{-3}], ρ is soil bulk density [ML^{-3}], D is dispersion coefficient [L^2T^{-1}], q is volumetric flux [LT^{-1}], and $\boldsymbol{\phi}$ is the rate constant representing reaction [$ML^{-3}T^{-1}$]

3.2.1.2. HYDRUS CW2D component and processes

HYDRUS CW2D entails both aerobic and anoxic transformation and degradation processes for organic matter, nitrogen, and phosphorus (Šimůnek et al., 1999). There are 12 components (Table 3) and 9 processes (Table 4).

Table 3. Components in HYDRUS CW2D (Langergraber and Šimůnek 2006)

Symbol	Description
O_2	Dissolved oxygen (mg/L)
CR	Readily biodegradable COD (mg/L)
CS	Slowly biodegradable COD (mg/L)
CI	Inert COD (mg/L)
NH_4N	Ammonium-nitrogen (mg/L)
NO_2-N	Nitrite-nitrogen (mg/L)
NO_3-N	Nitrate-nitrogen (mg/L)
N_2-N	Dinitrogen gas (mg/L)
IP	Inorganic phosphorus (mg/L)
XH	Heterotrophic microorganisms (mg/L)
XANs	Nitrosomonas - autotrophic microorganisms (mg/L)
XANb	Nitrobacter - autotrophic microorganisms (mg/L)

Table 4. Processes in HYDRUS CW2D (Langergraber and Šimůnek 2006)

Processes	Description
Hydrolysis	Converts CS to CR, and small fraction being converted in to CI.
Aerobic growth of heterotrophic bacteria	Consumes O_2 and CR.
Anoxic bacteria growth using nitrite	Consumes O_2, CR, ammonium (NH_4-N), and IP, and produce N_2 due to denitrification on nitrite.
Anoxic bacteria growth using nitrate	Consumes O_2, CR, ammonium (NH_4-N), and IP, and produce N_2 due to denitrification on nitrate.
Lysis of heterotrophic organisms	Produces CR, CS, CI, ammonium (NH_4-N), and IP.
Aerobic growth of nitrosomonas	Consumes O_2 and ammonium (NH_4-N), and produce nitrite (NO_2-N).
Aerobic growth of nitrobacter	Consumes nitrite (NO_2-N) and nitrate (NO_3-N).
Lysis of nitrosomonas (XANs)	Produces CR, CS, CI, ammonium (NH_4-N), and IP.
Lysis of nitrobacter (XANb)	Produces CR, CS, CI, ammonium (NH_4-N), and IP.

The following assumptions are made in HYDRUS CW2D (Langergraber and Šimůnek 2006). Organic matter is present only in the aqueous phase and all reactions occur only in the aqueous phase. Adsorption is assumed to be a kinetic process and considered for ammonium, nitrogen, and inorganic phosphorus. All microorganisms are assumed to be immobile. Lysis in HYDRUS CW2D represent all decay and loss processes of all microorganism involved, and the rate of lysis does not represent the impact of environmental conditions. Heterotrophic bacteria of HYDRUS CW2D include all bacteria responsible for hydrolysis, mineralization of organic matter (aerobic growth), and denitrification (anoxic growth).

3.2.1.3. Limitations of HYDRUS CW2D

The limitation of HYDRUS CW2D include the following (Langergraber et al. 2003; Langergraber and Simunek 2005; Langergraber and Šimůnek 2006; Leverenz et al. 2009).

- Clogging can occur from particulate matters in the influent wastewater settling and excessive growth of bacteria (biofilm). The resulting pore size reduction is not considered in the model.
- Impact of environmental condition on pH are not considered in the model.
- Limited to a temperature range between 10 and 25 °C.

3.2.1.4. Model calibration and validation

This section focuses on the procedure to calibrate and validate HYDRUS CW2D. Included are a description of laboratory experiments, calibration and validation procedure, and goodness of fit.

3.2.1.4.1. Laboratory experiment

The original purpose of the laboratory experiment was to observe the impact of enzyme pretreated fast-food restaurant wastewater on the performance and life of a drain field. This laboratory experiment is similar to the current study and is suitable for model calibration and validation because it simulated wastewater land application system with multiple strengths of wastewater. Substantial details can be found in Dong et al. (2017). In order to calibrate and validate HYDRUS, data from laboratory experiment was used. This laboratory study measured the required parameters for calibration and validation for HYDRUS such as soil moisture content, chemical oxygen demand, ammonia, and nitrate. Bench-scale drain fields (trenches), including soil moisture sensors embedded within the soil, were designed and operated. All dimensions of the trenches were based on the Michigan Criteria for Subsurface Sewage Disposal (Michigan Department of Public Health 1994). Figure 2 is a photograph of the soil trenches used for this research. The feedstock flowed by gravity into each trench. At the bottom of the trench, the treated water exited through a water trap that did not allow air flow into the trench. The width of the trench was 60.96 cm (2 ft), selected to accommodate one inlet pipe. A typical septic soil treatment system had multiple inlet pipes with a maximum separation of 91 cm (3 ft) between the pipes. Its length was 121.9 cm (4 ft). The first layer of soil, before wastewater entered, contained 22.9 cm (9 in) of top soil. Wastewater was distributed in the next layer, having a 7.6 cm (3 in) depth of gravel followed by the inlet pipes and then 15.2 cm (6 in) of gravel. The depth of the sandy loam that served as the treatment media was 60.9 cm (2 ft). The loading required for the soil used in this research, sandy loam, is 10.2 L/day/m^2 (0.25 gal/day/ft^2) (Michigan Department of Public Health 1994) resulting in a flow rate of 7.57 L/day (2 gal/day) and an empty bed contact time (EBCT) of 60 days. Six CS616 soil moisture sensors, manufactured by Campbell

Scientific, were placed at two depths at 3 locations along the length of the trench. All soil moisture sensors were connected to a CR1000 data logger, manufactured by Campbell Scientific. Readings from the soil moisture sensors were monitored automatically using the CR1000 data logger.

Influent was fed three times every day to simulate the cleaning schedule at a typical fast-food restaurant. The influent and effluent were collected weekly and analyzed for COD, BOD_5, Total Phosphorus (TP), Total Nitrogen (TN), ammonia, and nitrate.

Figure 2. Photographs of soil trenches (Dong et al. 2017)

Each trench received only one of the following feedstocks:

1. Domestic wastewater (Domestic WW)—control that does not cause premature aging of the septic soil treatment system.

2. Domestic wastewater mixed with food wastewater treated with enzymatic pretreatment (Domestic/Food WW)—typical test condition.

3. Food wastewater treated with enzymatic pretreatment (Food WW)—high loading test condition.

3.2.1.4.2. Goodness of fit

The most common method to evaluate the performance of HYDRUS CW2D model are model efficiency (E), index of agreement (IA), and root mean squared error (RMSE) (Anlauf and Rehrmann 2013; Wallach 2006; Wegehenkel, M. Beyrich 2014) . Model efficiency (E), originally developed by Nash and Sutcliffe, is defined in Equation 3 (Nash and Sutcliffe 1970).

$$\mathrm{E} = 1 - \frac{\sum_{i=1}^{N}(M_i - P_i)^2}{\sum_{i=1}^{N}\left(M_i - \overline{M}\right)^2} \tag{3}$$

Index of agreement (IA) was proposed by Willmott (1981), as defined in Equation 4.

$$\mathrm{IA} = 1 - \frac{\sum_{i=1}^{N}(M_i - P_i)^2}{\sum_{i=1}^{N}\left(\left|P_i - \overline{M}\right| + \left|M_i - \overline{M}\right|\right)^2} \tag{4}$$

Root mean squared error (RMSE), is defined in Equation 5 (Anlauf and Rehrmann 2013).

$$\mathrm{RMSE} = \sqrt{\frac{1}{\mathrm{N}} \sum_{i=1}^{N}(M_i - P_i)^2} \tag{5}$$

Where M is measured value, P is predicted value, and N is the number of observations. A range of E lies between - ∞ and 1.0. Typically, a model efficiency value between 0 and 1 represents an acceptable level of performance.

An E value below 0 is considered an unacceptable level of performance (Moriasi et al. 2007). An E value of 1 indicates that simulated and predicted value are equal to observed value. Phogat et al. (2016) suggested E > 0.12 and Qiao (2014) recommended E > 0 for evaluating the performance of their HYDRUS model. Both Analuf and Rehrmann (2013) and Arora et al. (2011) discussed that the acceptable quality should have E > 0.5 (Anlauf and Rehrmann 2013; Arora et al. 2011; Qiao 2014). A range of IA lies between 0 and 1, and a value of 0 indicates no agreement between measured and simulated values. A value of 1 indicates a perfect fit of observed to simulated values. The higher value of IA indicates better agreement between observed and simulated values. Phogat et al. (2016) reported acceptable quality of HYDRUS model is IA > 0.8 (Phogat et al. 2016).

RMSE measures the difference between measured and predicted values. Arora et al. (2011) discussed that generally lower RMSE indicates better agreement between measured and predicted values. Shekofteh et al. (2013) reported a RMSE of 0.0135, Wang et al. (2016) reported a RMSE of 0.12, and Ramos et al. (2012) reported a RMSE of 0.030 for their satisfactory model performance. (Ramos et al. 2012; Shekofteh et al. 2013; Wang et al. 2016).

The criteria to evaluate satisfactory model performance should include both relative error indices, such as E or IA, and absolute error measured, such as RMSE (Legates and McCabe 1999; Wegehenkel, M. Beyrich 2014). Therefore, this study evaluated the quality of the model using the following criteria; E > 0.5, IA > 0.8, and RMSE < 0.014. The calculations for E, IA, and RMSE were performed using Rstudio software (Boston, MA). Details of the code are provided in Appendix B.

3.2.1.4.3. Calibration and validation procedure

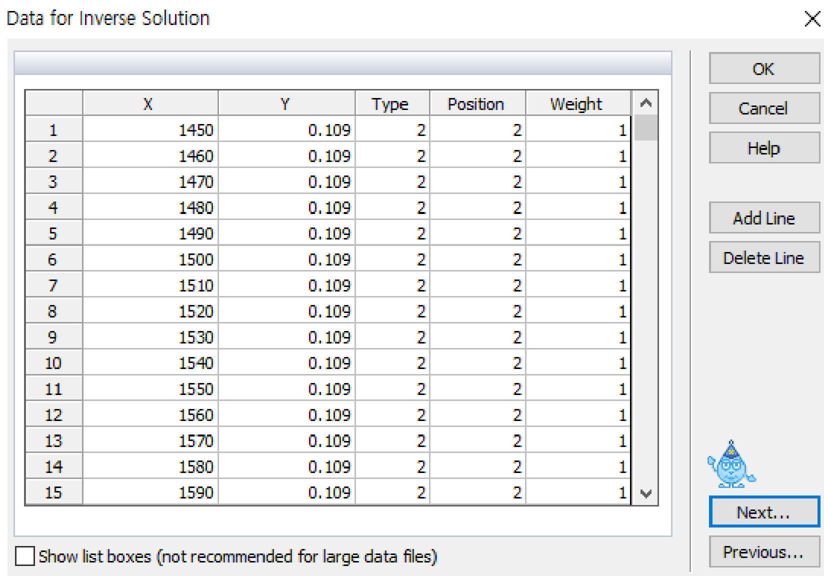
Calibration is described as the process of tuning by adjusting parameters and boundary conditions until the model result agrees with the experimental data. Validation is a process of quantifying the accuracy and credibility of the model (Šimůnek et al., 2012). Calibration and validation procedures are described below.

1. Calibrate the water flow of the model using measured volumetric water content data of the first two dosing periods of a day.
2. Validate the water flow of the model using the measured volumetric water content data of the last dosing period of the same day used in calibration.
3. Calibrate the solute flow of the model using measured COD, ammonia, and nitrate data in Domestic WW and Domestic/Food WW conditions from 108 days to 170 days (62 days). Initially, ammonia was not measured in the laboratory study. After 108th days, ammonia concentration was measured, which is needed for model calibration.

4. Validate the solute flow of the model using measured COD, ammonia, and nitrate data in Domestic WW and Domestic/Food WW conditions from 171 days to 225 days (54 days).

The soil saturated hydraulic and unsaturated hydraulic conductivity function are the most important hydraulic parameters in the Richards' equation (Radcliffe and Simunek 2010). In order to calibrate the water flow, the soil hydraulic parameters need to be optimal. Direct measurements of all the soil parameters are not always possible. An alternative indirect optimization using inverse modeling, as commonly used in hydrology modeling (Gupta et al. 2003). Inverse modeling in HYDRUS uses the initial estimate of the parameters to perform the simulation and compares the simulation results to the observed experimental data. The model is then re-run with modified set of parameter. The process is repeated until the modeled data closely match the observed experimental data (Rassam et al. 2003).

Data from a total of 144 measured volumetric water contents from a laboratory experiment were used for calibration. Figure 3 shows a screenshot of the inverse modeling routine in HYDRUS CW2D. The inverse solution function optimizes the following soil parameters: *Ks* (Saturated hydraulic conductivity), Alpha (Parameter in the soil water retention function), n (Parameter n in the soil water retention function), and I (Tortuosity parameter in the conductivity function).



	X	Y	Type	Position	Weight
1	1450	0.109	2	2	1
2	1460	0.109	2	2	1
3	1470	0.109	2	2	1
4	1480	0.109	2	2	1
5	1490	0.109	2	2	1
6	1500	0.109	2	2	1
7	1510	0.109	2	2	1
8	1520	0.109	2	2	1
9	1530	0.109	2	2	1
10	1540	0.109	2	2	1
11	1550	0.109	2	2	1
12	1560	0.109	2	2	1
13	1570	0.109	2	2	1
14	1580	0.109	2	2	1
15	1590	0.109	2	2	1

Figure 3. Inverse modeling data input in HYDRUS CW2D; X - time, Y - measured data; Type - pressure head (1), volumetric water content (2), solute concentration (4); Position - observation node number corresponding to where the volumetric water content is measured; Weight - weight associated with a particular data point

Unlike water flow calibration, HYDRUS CW2D solute flow does not provide inverse modeling. The HYDRUS CW2D solute process parameters need to be manually calibrated by using a trial and error approach (Dittmer et al. 2005; Langergraber et al. 2007; Palfy et al. 2016; Palfy and Langergraber 2014; Pucher 2015).

The first step to calibrate the solute flow of the model is to determine the characteristics of the wastewater. COD in HYDRUS CW2D model is divided into three fractions including readily CR, CS, and CI. Several approaches to fractionize COD into CR, CS, and CI are discussed. Palfy et al. (2016) reported CR:CS:CI ratio at 60:20:20, 40:40:20, and 30:60:10 for their study (Palfy et al. 2016). Dalahmeh et al. (2012) conducted a study assuming that CR is being measured as the influent BOD_5 concentration and CI is 0 % of the observed effluent COD level. The remaining COD was set to CS (Dalahmeh et al. 2012). Henrichs et al. (2007) assumed the percentage of COD for CR, CS, and CI were 5 - 20 %, 60 – 90 %, 5 – 19 % of influent COD concentration, respectively. The CI was also considered to be 80 – 90 % of the observed COD effluent concentration (Henrichs et al. 2007). Other studies set the CI value to 85% of the measured COD effluent concentration. The CR to CS ratio was then estimated to be approximately 2:1 (Dittmer et al. 2005; Henrichs et al. 2007; Toscano et al. 2009). A preliminary test to fractionate CR, CS, and CI was conducted and the best estimate for the CI is 85 % of the measured COD effluent and CR to CS ratio being 2:1 of remaining COD.

Wastewater composition such as COD (CR, CS, CI), ammonia-nitrogen, and nitrate-nitrogen were inputted in time variable boundary condition as a concentration (mg/L). Table 5 shows the values used to calibrate and validate HYDRUS CW2D. The calibration and validation values in Table 5 are average concentrations from 108 days to 170 days (62 days) and from 171 days to 225 days (54 days), respectively. Since the COD concentration in domestic wastewater is from 99 to 445 mg/L, this model was calibrated for two different COD values of wastewaters using laboratory experiments: Domestic WW (102.1 mg/L of COD) and Domestic/Food WW (519.1 mg/L of COD) (Brown et al. 1997; Dong et al. 2017; Hammerlund and Glotfelty 2016; Hossain 2008; Ronayne et al. 1982).

Table 5. Wastewater characteristic input parameters for HYDRUS CW2D calibration and validation

Parameter	HYDRUS CW2D Symbol	Calibration		Validation	
		Domestic WW	Domestic/Food WW	Domestic WW	Domestic/Food WW
CR (mg/L)	cVal1-2	51.5	385	48.9	470
CS (mg/L)	cVal1-3	25.7	193	24.4	235
CI (mg/L)	cVal1-4	24.9	41.0	27.4	41.9
Ammonia (mg/L-N)	cVal1-8	28.7	28.9	30.0	28.0
Nitrate (mg/L-N)	cVal1-10	5.18	1.00	5.01	1.10

Once the characteristics of wastewater were determined, adjustment of kinetic parameters were performed using trial and error. According to previous studies, hydrolysis rate constant, lysis rate for microorganisms (XH, XANs/b), maximum aerobic growth rate of XANs, maximum denitrification rate of XH, and fraction of CI generated in biomass lysis were adjusted using the calibration process. (Fuchs 2009; Heatwole and McCray 2007; Palfy et al. 2016; Pucher and Langergraber 2018). The procedure for adjusting the kinetic parameters is described below (Palfy et al. 2016; Pucher and Langergraber 2018).

1. Run the model using the standard parameters of the HYDRUS CW2D biokentic model (Langergraber and Simunek 2005).
2. Adjust the fraction of CI generated in biomass lysis value when the measured and simulated COD effluent concentrations are different.

3. Address the growth of bacterial groups, XH and XANs/XANb. using lysis rates (b_h, b_{ANs}, b_{ANb}) by adjusting each until steady state is reached (Palfy et al. 2016; Pucher and Langergraber 2017).
4. Modify the maximum aerobic growth rate, XANs, when measured and simulated ammonia effluent concentrations are different (Pucher and Langergraber 2017).
5. Adjust the hydrolysis rate and/or maximum denitrification rate for heterotrophic microorganisms when measured and simulated nitrate effluent concentrations are different (Pucher and Langergraber 2017). By decreasing the hydrolysis rate, less organic matter is degraded in the upper layer of soil and more is available for the denitrification process as an electron donor.

Time variable boundary condition was used for modeling domestic wastewater land application. This allowed the user to assign specific dosing times, dosing periods, and flux per day, and could repeat these conditions over the entire length of the simulation. In this study, dosing time followed a specific day in the laboratory experiment and the dosing period was set at 30 seconds. Flux was calculated as the total volume of applied wastewater divided by surface area. Table 6 shows the time variable boundary condition input values for calibration and validation. Detail of input parameters in HYDRUS CW2D is provided in Appendix A.

Table 6. Time variable boundary condition for calibration and validation

Time (min)	Flux (cm/min)
449.5	0
450	0.708
619.5	0
620	0.708
769.5	0
770	0.708
1440	0

3.2.1.5. Scenarios

Because the literature review showed that current design criteria for wastewater land application systems are generally based on limited empirical relationships, the calibrated and validated model was used to simulate multiple, common application scenarios. Included is the soil depth, hydraulic and organic loadings, and dosing frequency.

Nitrate contamination of groundwater (concentration in groundwater >10 mg/L-N) occurs even in well-constructed and properly functioning domestic wastewater land application systems (Wilhelm et al. 1994). Complete nitrification usually occurs within the first 30 cm of the soil depth, while complete denitrification typically does not occur in domestic wastewater land application systems (Beach 2001; Fischer 1999; Heatwole and McCray 2007). Therefore, the first scenario was conducted to evaluate operation parameters that may enhance nitrate removal. The denitrification process requires carbon and anaerobic condition for denitrifying bacteria such as *Heterotrophic bacteria*, *Thiobacillus denitrificans*, *micrococcus denitrificans*, *Pseudomonas*, and *Achromobacter* (Carlson and Ingraham 1983). Thus, increasing the dosing frequency may cause periodic saturated conditions that result in anaerobic soil conditions leading to the promotion of growth of denitrifying bacteria that will enhance nitrate removal. Multiple dosing frequencies including 3x dosing frequency, 6x dosing frequency, 10x dosing frequency, and continuous dosing were tested while maintaining constant hydraulic and organic loadings. Table 7 shows the time variable boundary condition for multiple dosing frequencies. The flux was divided into respective each dosing frequency.

Table 7. Time variable boundary condition for dosing frequency

Flux (cm/day)							
Time (day)	**3x dosing frequency**	**Time (day)**	**6x dosing frequency**	**Time (day)**	**10x dosing frequency**	**Time (day)**	**Continuous**
0.31215	0	0.31215	0	0.13889	0	1	1.062
0.31250	1019.52	0.31250	509.76	0.13924	305.86		
0.43021	0	0.43021	0	0.20833	0		
0.43056	1019.52	0.43056	509.76	0.20868	305.86		
0.53438	0	0.53438	0	0.31215	0		
0.53472	1019.52	0.53472	509.76	0.31250	305.86		
1	0	0.61111	0	0.43021	0		
		0.61146	509.76	0.43056	305.86		
		0.68056	0	0.53438	0		
		0.68090	509.76	0.53472	305.86		
		0.69445	0	0.61111	0		
		0.69479	509.76	0.61146	305.86		
		1	0	0.68056	0		
				0.68090	305.86		
				0.75000	0		
				0.75035	305.86		
				0.81944	0		
				0.81979	305.86		
				0.88889	0		
				0.88924	305.86		
				1	0		

3.3. Result and discussion

Results and discussion for monitoring, benefits, and modeling of domestic wastewater land application are presented in the following sections.

3.3.1. Monitoring of domestic wastewater land application

Domestic wastewater land application has been used for many years and its performance is documented in many previous studies. Hence experimentation to determine effectiveness was not part of this project as the literature was used. Table 8 summarizes several manuscripts. The average removal efficiency of COD, BOD_5, total phosphorus (TP), and ammonia are over 60%. Typical domestic wastewater land application nitrifies most of the ammonia to nitrate but does not denitrify nitrate to nitrogen gas (Beach 2001; Fischer 1999; Heatwole and McCray 2007). Since nitrate is highly mobile, nitrate leaching into groundwater is a concern.

Table 8. Typical domestic wastewater land application treatment performance

Parameter	Influent	Effluent	Reference
COD (mg/L)	99-445	29.3	(Brown et al. 1997; Dong et al. 2017; Hammerlund and Glotfelty 2016; Hossain 2008; Ronayne et al. 1982)
BOD_5 (mg/L)	32-217	0-17	(Dong et al. 2017; Hossain 2008; National Environmental Services Center 2013; Ronayne et al. 1982; Tchbanoglous et al. 2003)
Total Phosphorus (TP) (mg/L-P)	4.5-30	0.01-4.9	(Dong et al. 2017; Hammerlund and Glotfelty 2016; Hossain 2008; National Environmental Services Center 2013; Tchbanoglous et al. 2003)
Total Nitrogen (TN) (mg/L-N)	25-63.4	42.3-48	(Dong et al. 2017; Gross 2004; Hammerlund and Glotfelty 2016; National Environmental Services Center 2013; Ronayne et al. 1982)
Ammonia (mg/L-N)	20-60	0.03-0.1	(Cui et al. 2003; Dong et al. 2017; Hossain 2008)
Nitrate (mg/L-N)	0-10	39.1-42	(Brown et al. 1997; Dong et al. 2017; Hossain 2008; Ronayne et al. 1982; Tchbanoglous et al. 2003)

3.3.2. HYDRUS CW2D modeling

The results of the model's calibration, validation, and scenarios to determine the depth requirement for the wastewater land application systems and treatment enhancement are discussed below.

3.3.2.1. Model calibration and validation

Model calibration was conducted by inverse modeling using volumetric water content measurement data. Figure 4 shows the comparison of measured and fitted for the HYDRUS volumetric water content. The volumetric water content data measured in the first two doses were used for the calibration and the remaining data was used for model validation.

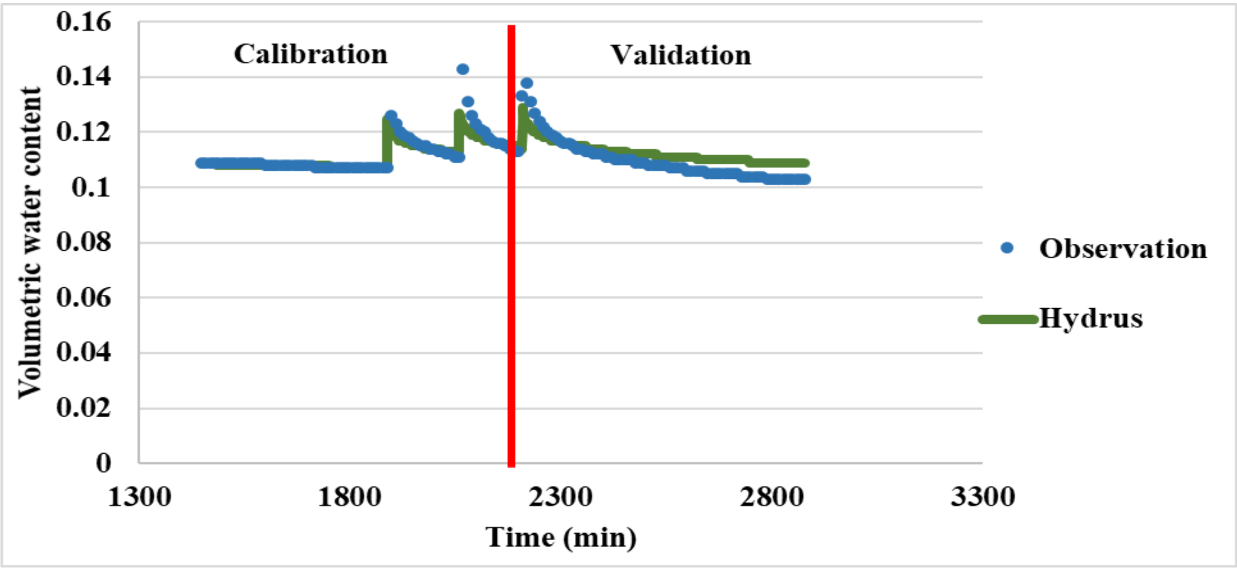


Figure 4. Fitted HYDRUS to measured value using volumetric water content from Domestic WW

Table 9. Goodness of fit result of calibration and validation

Model	E	IA	RMSE
Calibration	0.65	0.87	0.004
Validation	0.6	0.82	0.005

The calibrated and validated water flow model was evaluated by E, IA, and RMSE. As shown in Table 9, all of the values were met the quality of modeling criteria, E > 0.5, IA > 0.8, and RMSE <0.014.

Table 10 shows the adjusted HYDRUS CW2D parameters for domestic wastewater land application modeling. Based on literature review, six parameters were considered in the calibration process for the solute flow. After trial and error, a maximum aerobic growth rate of XANs, maximum denitrification rate of XH, and fraction of CI in biomass lysis were adjusted.

Table 10. Adjusted HYDRUS CW2D parameters for domestic wastewater land application modeling

Parameter	**Description**	**Unit**	**Standard**	**Adjusted**
k_h	Hydrolysis rate	1/d	3	-
b_h	Lysis rate for XH	1/d	0.4	-
$b_{ANs/b}$	Lysis rate for XANs/b	1/d	0.15	-
μ_{ANs}	Maximum aerobic growth rate of XANs	1/d	0.9	0.45
μ_{dn}	Maximum denitrification rate of XH	1/d	4.8	3.0
$f_{BM,CI}$	Fraction of CI in biomass lysis	1/d	0.02	0.01

Table 11 shows the simulated effluent concentrations before and after the fitting process. The table contains averages and standard deviations of measured influent, effluent, and simulated values using standard and adjusted parameters from the calibration process. The measured values from day 108 to 170 (62 days) in Domestic WW and Domestic/Food WW were used.

Table 11. Simulated effluent concentrations before and after the fitting process for calibration

Condition	**Inf/Eff**	**Type of value**	**HYDRUS CW2D Parameters**	**COD (mg/L)**	**Ammonia (mg/L)**	**Nitrate (mg/L)**
Domestic WW	Influent	Measured avg. (Std.)		102.3 (14.1)	28.7 (3.2)	5.18 (3.5)
	Effluent	Measured Avg. (Std.)		29.4 (5.4)	0.2 (0.1)	38.4 (3.3)
		Simulated	Standard	28.7	0.1	35.6
		Simulated	**Adjusted**	**26.8**	**0.2**	**35.5**
Domestic/Food WW	Influent	Measured Avg. (Std.)		619.0 (142.3)	28.9 (3.2)	1.0 (0.6)
	Effluent	Measured Avg. (Std.)		48.1 (7.2)	0.5 (0.2)	29.4 (4.6)
		Simulated	Standard	59.5	0.1	21.2
		Simulated	**Adjusted**	**50.3**	**0.25**	**29.6**

Table 12 shows the relative differences between the measured and simulated values in calibration process. The stimulated COD, ammonia, and nitrate values using standard parameters in Domestic WW conditions are not significantly different with the measured values. However, the stimulated COD, ammonia, and nitrate values using standard parameters in Domestic/Food WW conditions were -19.2 %, 400 %, and 38.7 % different than measured values, respectively. In order to address these differences, the maximum aerobic growth rate of XANs, maximum denitrification rate of XH, and the fraction of CI in biomass lysis were adjusted. After adjustment, the relative difference of COD, ammonia, and nitrate in Domestic/Food WW decreased from -19.2 % to -4.37 %, from 400 % to 100 %, and from 38.7 % to -0.68 %, respectively. The relative difference of COD and nitrate in Domestic WW condition was increased after an adjustment was made, but not significantly. This was a compromise to maximize the fit for both Domestic wastewater and Domestic/Food WW conditions. Although the relative difference value for ammonia was high, the predicted model value was trace level, therefore, the model concentration was well predicted even for the standard parameters.

Table 12. Relative difference between measured and simulated values for calibration

Condition	**HYDRUS CW2D**	**COD**	**Ammonia**	**Nitrate**
Domestic WW	Standard	2.44 %	100 %	7.87 %
	Adjusted	9.70 %	0 %	8.17 %
Domestic/Food WW	Standard	-19.20 %	400 %	38.70 %
	Adjusted	-4.37 %	100 %	-0.68 %

Table 13 shows the simulated effluent concentrations before and after the fitting process for validation. The table contains average and standard deviations of measured influent, effluent, and simulated values using standard and adjusted parameters in the validation process. The measured values from day 171 – 225 (54 days) in Domestic WW and Domestic/Food WW wastewater were used.

Table 13. Simulated effluent concentrations before and after the fitting process for validation

Condition	**Inf/Eff**	**Type of value**	**COD (mg/L)**	**Ammonia (mg/L)**	**Nitrate (mg/L)**
Domestic WW	Influent	Measured Avg. (Std.)	100.8 (6.9)	30.0 (4.4)	5.01 (4.8)
	Effluent	Measured Avg. (Std.)	32.3 (8.3)	0.15 (0.1)	39.4 (3.3)
		Simulated	29.2	0.2	36.7
Domestic/Food WW	Influent	Measured Avg. (Std.)	746.5 (111.1)	28.0 (4.5)	1.1 (0.9)
	Effluent	Measured Avg. (Std.)	49.3 (13.1)	0.1 (0.1)	30.9 (3.3)
		Simulated	52.3	0.26	28

Table 14 show the relative differences between measured and simulated values in the validation process. Except for ammonia, the largest relative difference between the measured and simulated values were 10.6%. Although the large number of relative difference was shown among ammonia, the difference between 0.1 and 0.26 is not significant with respect to field conditions. The model was successfully calibrated and validated for both water and solute flow using the laboratory experimental data.

Table 14. Relative difference between measured and simulated values for validation

Condition	COD	Ammonia	Nitrate
Domestic WW	10.60 %	15.40 %	4.23 %
Domestic/Food WW	6.09 %	-61.50 %	10.40 %

3.3.2.2. Scenario – capacity of wastewater land application

With the model calibrated and validated, observations on the effects of multiple hydraulic and organic loadings on COD treatment performance were conducted. The COD treatment performance was observed at depths of 15.24, 30.48, 60.96, 91.44, and 121.9 cm (0.5, 1, 2, 3, and 4 ft) with 1x, 2x, 3x, 4x, and 5x strength of hydraulic and organic loadings. This simulation was conducted with 3 dosing frequencies. A COD of 102.3 mg/L and COD effluent concentration at multiple depths of 150th days were observed. The 150th day was selected for the comparison because the COD effluent concentration did not significantly change after 60 days. Figure 5 shows the COD treatment performance at multiple depths with different strengths of hydraulic and organic loadings. The efficiency of COD removal at 15.24 cm (0.5 ft) and 30.48

cm (1 ft) started to decrease after 2x and 3x strength of hydraulic and organic loadings, respectively. However, COD concentrations below 60.96 cm (3 ft) did not significantly change as loading strength increased. This result shows that most of the COD treatment was within a 30.48 cm (1 ft) depth of soil, as confirmed by others (Guilloteau et al. 993; Pan et al. 2017). Consequently, sandy loam at a soil depth of 60.96 cm (2 ft) can treat 3 times higher hydraulic and organic loadings without decreasing the COD treatment performance. The state of Maryland requires a minimum soil depth of 15 cm (6 in) for drain field. According to the model result, 15 cm (6 in) can adequately treat the typical domestic wastewater loading. However, for households that produce higher strength wastewater, which can be caused by large amount of food waste and more frequent laundry, a 60 cm (24 in) soil depth is recommended to ensure the treatment while minimizing environmental impact.

HYDRUS does not consider the growth of biofilm. Biofilm is a combination of microbial cells and an extra-cellular polymer matrix (Lazarova and Manem 1995). The biofilm is often referred to as the clogging zone (Siegrist and Van Cuyk 2001), crust development (Magdoff et al. 1974), biofilm (Dong et al. 2017b; Siegrist and Gujer 1985), biomat, or biozone (Beach et al. 2005; Siegrist and Van Cuyk 2001). Biofilm creates a hydraulic barrier, which encourage the distribution of wastewater throughout the field (Beach 2001). However, excessive growth of biofilm can restrict the flow and decrease the hydraulic conductivity. Currently, the growth of biofilm and the change of the hydraulic properties by the biofilm are not considered in HYDRUS CW2D. In order to address this limitation, understanding the thickness, hydraulic conductivity, and development rate of the biofilm is needed. High strength domestic wastewater may stimulate the growth of biofilm because of its high carbon content. To prevent clogging, organic and hydraulic loading should be reduced.

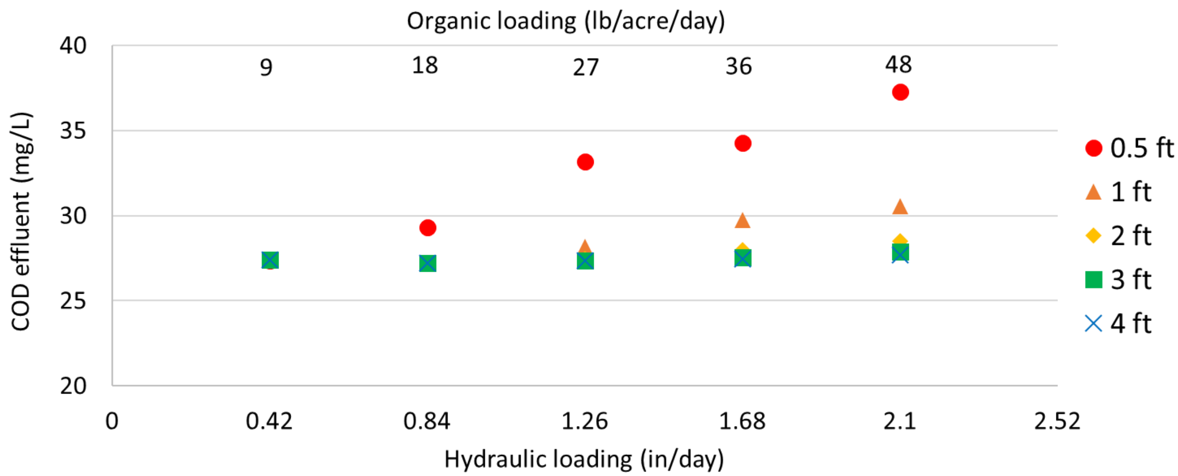


Figure 5. Simulated COD effluent concentration at 15.24, 30.48, 60.96, 91.44, and 121.9 cm (0.5, 1, 2, 3, and 4 ft) depths as increasing loading strength (1x-5x) on 150th days of operation

3.3.2.3. Scenario – treatment performance enhancement

Incomplete nitrate treatment in domestic wastewater land application system is already known. HYDRUS CW2D was conducted to observe the potential impact of dosing frequency along with the strength of the influent COD concentration on nitrate treatment performance. These two parameters may reduce oxygen levels in the soil creating anaerobic zones and increase the available carbon sources for denitrifying bacteria. A nitrate effluent concentration on the 150th day at 60.96 cm (2 ft) depth of soil was observed. As Figure 6 shows, the nitrate effluent concentration from the HYDRUS CW2D simulation did not significantly change after 100 days, which was mathematically proven by a slope of -0.00001. In Figure 7, the growth of heterotrophic bacteria supports the above since it did not significantly change after 100 days.

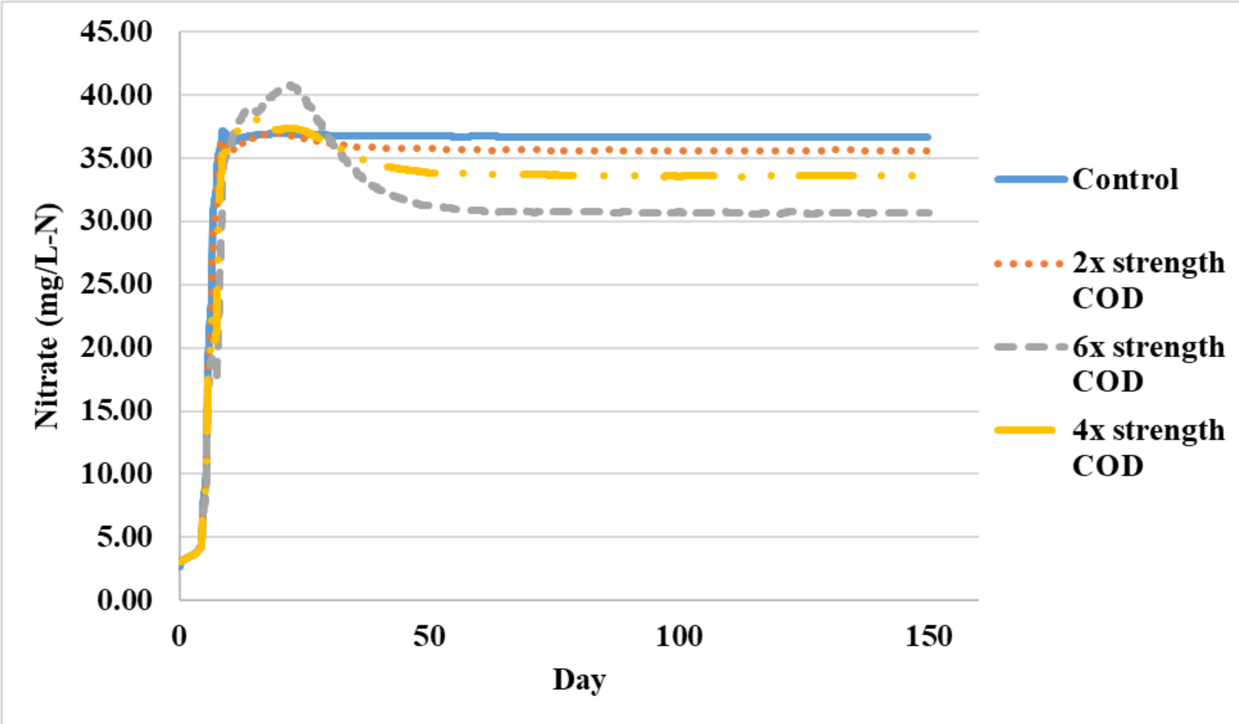


Figure 6. Simulated nitrate effluent concentrations with multiple influent COD strength

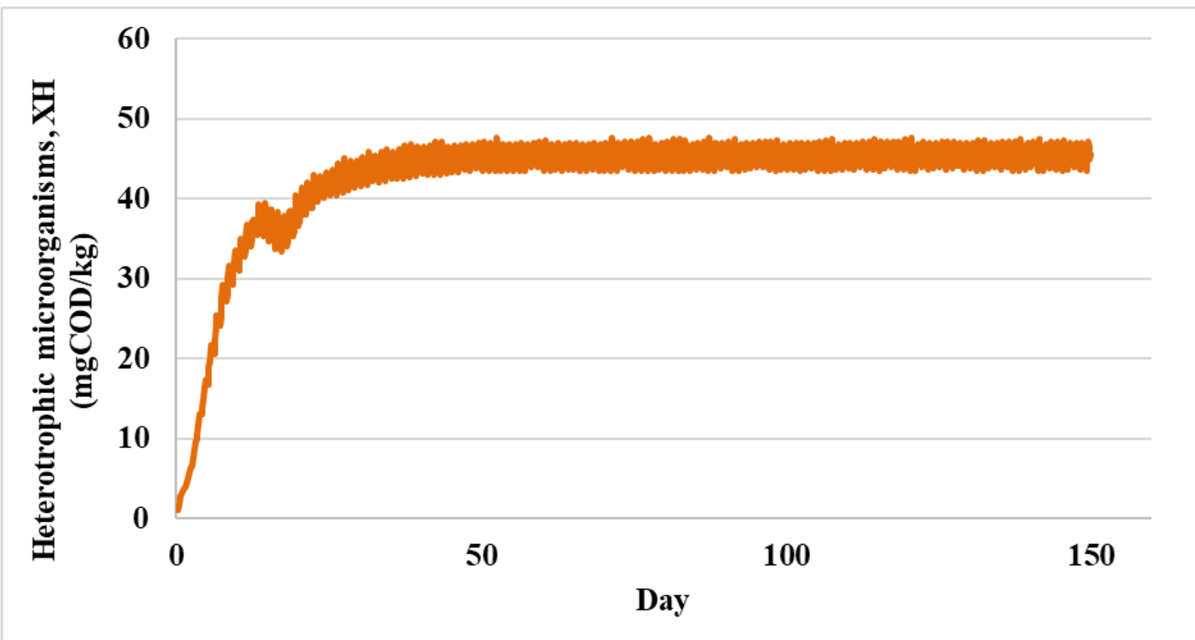


Figure 7. Steady state condition of heterotrophic microorganism (XH) growth

Comparison of COD and nitrate effluent concentrations effected by dosing frequency and influent COD strengths were observed. Figure 8 shows the impact of the dosing frequency and the strength of influent COD concentration on its effluent concentration. Table 15 shows the relative differences from the control, which had a 3X dosing frequency with 102 mg/L of COD. The removal efficiency in Figure 8 is calculated by subtracting continuous from 3 dosing frequency to show the effect of dosing frequency on COD removal. As observed, there is a direct relationship between COD influent and effluent concentration - as the influent COD concentration, increases the COD effluent concentration also increases. Increasing the dosing frequency was effective in reducing the COD effluent concentrations by a maximum of 5.6 mg/L of COD. However, the removal of COD at an influent COD concentration of 102 mg/L was not impacted by an increase in dosing frequency. The removal efficiency increase as COD influent concentration increases. COD in the effluent might be the inert form, which is non-biodegradable. A possible explanation for a lower COD effluent concentration with higher dosing frequency is an increase in retention time for treatment correlated with the higher dosing frequency. Because the removal efficiency was not significant, increasing the dosing frequency to improve COD treatment is not recommended.

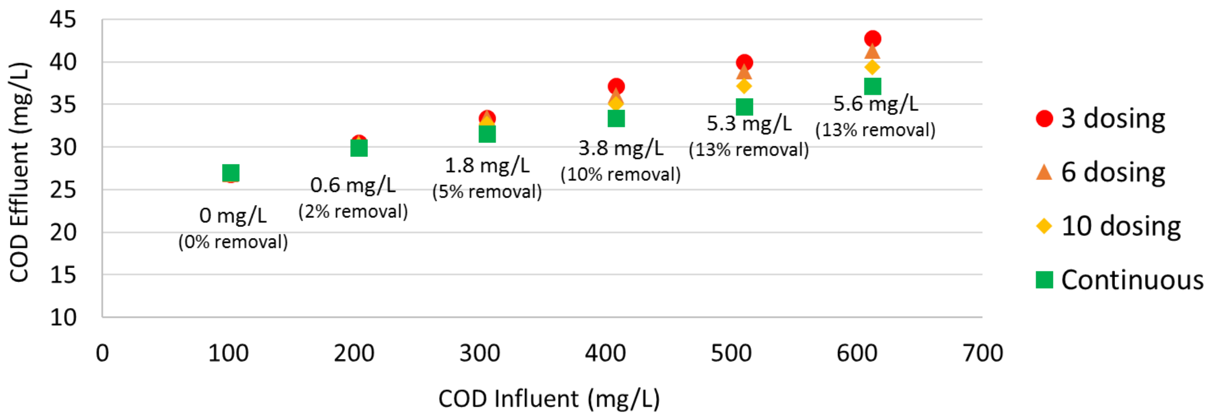


Figure 8. Impact of dosing frequency and influent COD concentration on COD effluent concentration on the 150th day at 60.96 cm (2 ft) depth of sandy loam soil

Table 15. Relative difference from control for COD simulations (3 dosing frequency with 102 mg/L of COD)

Influent COD Concentration (mg/L)	**Relative difference from control (mg/L)**			
	3 Doses	**6 Doses**	**10 Doses**	**Continuous**
102	0	0	0	0.2
204	3.7	3.7	3.4	3.1
306	6.6	6.7	5.9	4.8
408	10.4	9.3	8.2	6.6
510	13.2	12.1	10.4	7.9
612	16.0	14.5	12.6	11.2

Figure 9**Error! Reference source not found.** shows the impact of dosing frequency and COD concentration on nitrate effluent level estimation at 60.96 cm (2 ft) depth of sandy loam soil. Table 16 shows the relative difference from the control (3 dosing frequency with 102 mg/L of COD). The removal efficiency in Figure 9 was calculated by subtracting continuous from 3 dosing frequency. At an influent COD concentration of 102 mg/L, the nitrate was not impacted significantly by an increase in dosing frequency. This may indicate that the influent COD concentration of 102 mg/L is not provide enough carbon for the denitrification process. However, the nitrate effluent concentration was lower when the influent COD concentrations was above 204 mg/L and the dosing frequency was more frequent.

The most nitrate removal was observed with the highest concentrations of COD (612 mg/L) with 10 dosing frequency and continuous loading. There was no significant difference observed in nitrate effluent concentrations between a 10 dosing frequency and continuous condition. Higher dosing frequency may increase the moisture content of the soil, leading to a reduction in the oxygen content, providing optimal conditions for denitrification. Figure 10 shows higher soil water content was observed in 10 dosing frequency than 3 and 6 dosing frequencies. Additionally, a higher influent COD concentration provides more substrate for microorganisms, which may stimulate the denitrification process. For households with typical domestic wastewater, increasing the dosing frequency does not have a significant impact on nitrate removal. However, for households and facilities that produce higher COD strength wastewater, than 102 mg/L COD, 10 dosing frequency or continuous dosing can reduce effluent nitrate levels. Increasing dosing frequency may require upgrading the distribution system including pumping capacity an economic analysis.

Currently, the model did not consider factors such as precipitation, seasons, topography, and growth of biofilm. However, the model result shows estimated treatment performance affected by dosing frequency and influent COD concentration. When the model considers the above factors, the predicted value of the model may be different but might not be significant. Further, calibration and validation of the model using different soil type, precipitation, seasons, topography, and growth of biofilm are needed.

Seasons can impact nitrate removal. As the soil temperature increases, microorganism activity also increases, which can promote nitrification and denitrification processes. As the soil temperature decreases, microbial activity decreases, which may slow the nitrification and denitrification processes. Therefore, the nitrate removal efficiency will decrease in winter.

During long-term operation, the drain field can be clogged by biofilm. When the drain field is clogged, the soil becomes anaerobic, which is the optimal condition for the denitrification process. However, a clogged drain field cannot handle the design hydraulic loading, potentially resulting in an overflow and unpleasant odors. Therefore, replacement or resting the drain field for a year is recommended.

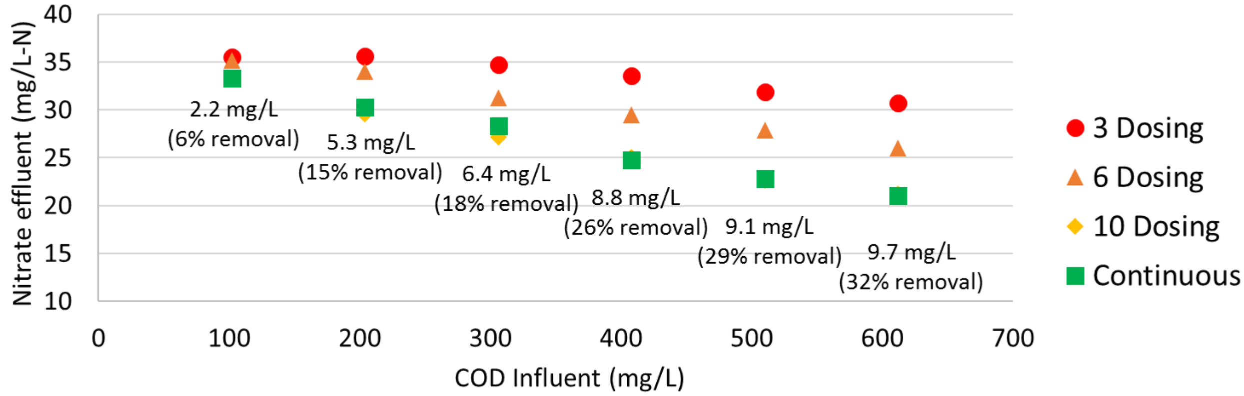


Figure 9. Impact of dosing frequency and COD concentration on nitrate effluent level estimation at 60.96 cm (2 ft) depth of sandy loam soil

Table 16. Relative difference from control for nitrate simulation (3 dosing frequency with 102 mg/L of COD)

Influent COD Concentration (mg/L)	**Relative difference from control (mg/L)**			
	3 Doses	**6 Doses**	**10 Doses**	**Continuous**
102	0	-0.3	-2.3	-2.2
204	0.1	-1.5	-5.9	-5.2
306	-0.8	-4.2	-8.3	-7.2
408	-1.9	-6	-10.5	-10.7
510	-3.6	-7.6	-12.9	-12.7
612	-4.8	-9.5	-14.3	-14.5

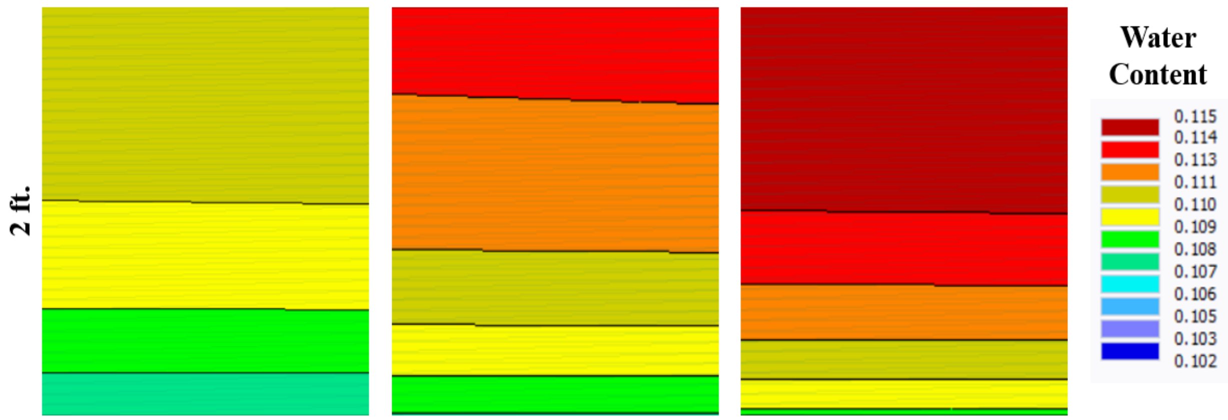


Figure 10. Simulation of the impact of volumetric water content in soil by different dosing frequencies at 150th day; 3 dosing (left), 6 dosing (middle), and 10 dosing (right)

3.4. Conclusion

In conclusion, a domestic wastewater land application system is effective in treating COD, BOD_5, TP, and ammonia but not nitrate. In comparison to a conventional wastewater treatment system, domestic wastewater land application can save energy for the treatment, consequently reducing GHG emissions. Besides these benefits, lower capital and operation cost, less maintenance requirement, and no chemicals requirements are other benefits.

HYDRUS CW2D was successfully calibrated and validated using measured volumetric water content from a laboratory experiment. Most of the COD treatment in domestic wastewater land application system occurs within 15.24 cm (1 ft) depth of sandy loam soil. Two feet depth of soil for domestic wastewater land application system is ideal with the consideration of a safety factor. The model simulation shows the potential nitrate removal by increasing both dosing frequency and influent COD concentration.

Future studies are warranted. The model should be verified in a long-term study. In addition, the model should be calibrated and validated using different types of soil, weather condition, and hydraulic and organic loadings. Once the model is calibrated and validated, multiple scenario to determine the best operation strategies should be performed. The results should be provided as an index, which is an integrated approach. The index can be beneficial for onsite wastewater engineers/designers and regulators to determine the depth operation strategies.

Chapter 4. Food processing wastewater land application

This chapter provides background information on the food processing wastewater irrigation demonstration site. Monitoring strategies to prevent and minimize nitrate and metal leaching into the groundwater are first discussed followed by the evaluation of site condition and the benefits of food processing wastewater land application such as freshwater saving, nitrogen and phosphorus reuse, energy savings, and GHG reduction associated with the energy saving. In addition, the potential use of HYDRUS CW2D is discussed.

4.1. Introduction

In the United States, 1 trillion gallons of wastewater are produced annually from food processing industry (Aryal 2015). Food processing wastewater characteristics vary depending on the facility, technology, and type of food being processed. Typically, food processing wastewater includes organic carbon, nutrients, suspended solids, descaling chemicals, food additives, salts, and equipment cleaners (Safferman et al. 2007). The characteristics of food processing wastewater are summarized in Table 17.

Table 17. Characteristic of food processing wastewater

Type of processor	COD (mg/L)	BOD_5 (mg/L)	TP (mg/L-P)	TN (mg/L-N)	Nitrate (mg/L-N)	Reference
Milk and dairy products	1,025	4,841	154	663		(Christian 2010)
Meat	1,684	863	328	2,744		
Slaughterhouse	1,000-6,000	1,000-4,000	80-120	250-700		(Tritt and Schuchardt 1992)
Milk	2,833	1,216	77	70		(Konieczny et al. 2005)
Meat	2,392	646	13	80		
Fish	3,017	914	43	181		
Confectionery	530-2,620					(Di Berardino et al. 2000)
Starch	6,222					(Deng et al. 2003)
Poultry	364-1,219				2.9-13.5	(Pierson and Pavlostathis 2000

Table 17. Characteristics of food processing wastewater (cont'd)

Type of processor	COD (mg/L)	BOD_5 (mg/L)	TP (mg/L-P)	TN (mg/L-N)	Nitrate (mg/L-N)	Reference
Fish	326-1,432	3,500		117		(Chowdhury et al. 2010)
Olive oil	220-400					(Wang, Huang, and Yuan 2005)
Slaughterhouse	2,870					(Sayed et al. 1988)
Potato	5,978	12,489	1,277	308	0.22	(Dornbush, Rollag, and Trygstad 1976)
Pear	3,050	2,040				(Esvelt 1970)
Peach	2,150	1,810				
Apple	1,400-1,520	950-1,230				
Wine		300-30,000		1-225		(CVRWQCB 2005)
Apple	9,000					(Van Ginkel et al. 2005)
Potato	21,000					
Confectionery	600-20,000					

This wastewater typically contains nutrients and water, valuable resources for crop production. Producing a valuable crop commodity has the benefits of reducing the use of fresh water, commercial fertilizers, and energy by eliminating the need for a traditional wastewater treatment facility and, consequently, reduces GHG emission. The calculation was conducted using the data from 2016 at the project's long-term food processing wastewater land application site as a demonstration to show the potential benefits of food processing wastewater land application.

The average BOD_5 concentration of the food processing wastewater and groundwater from the sampling wells were 680 mg/L and 2 mg/L, respectively. The total BOD_5 removal was 459,427 kg (1,012,864 lb). To remove 0.45 kg of BOD at a traditional activated sludge wastewater treatment facility with a flow less than 3.78 million liters/day (1 million gallons/day), 4.1 kWh of energy is required (NYSERDA 2007). Therefore, 4,200,000 kWh electricity was saved in 2016. This results in a GHG reduction of 3,126 metric tons, which is equivalent to the GHG emission from 669 passenger vehicles driven for one year with assumptions of 22 miles/gallon and 11,443 miles/year driven (USEPA 2016).

Nitrogen and phosphorus are essential for crop production. Applied nitrogen loading at the demonstration site was estimated at 41.7 g-N/m^2 (371.9 lb-N/acre) in 2016. The amount of nitrogen required for the crop yield was estimated at 22.4 g-N/m^2 (200 lb-N/acre). This indicates that the excessive nitrogen is applied on the land. The higher than required nitrogen loading is of concern, especially when the crops are not actively growing. The applied nitrogen was adequate for crop production and the yield was expected. In addition to not having to purchase commercial fertilizers, savings results from minimizing fuel use for tractors to apply fertilize and trucks to transport fertilizers, which result in GHG reduction.

Although there are many benefits, improper operation can result in nitrate leaching or metal mobilization into groundwater (Dong et al. 2017a; Julien and Safferman 2015; Redding 2012). The USEPA provides water quality standards. The standards include a maximum contamination levels, which are regulated, and a secondary maximum contamination levels, which are not regulated but of concern. The main concerns in the study were the concentrations of nitrate, arsenic, manganese, and iron in the groundwater. Maximum allowable level of contamination for nitrate is 10 mg/L-N (USEPA 2017d). Secondary maximum contaminant level for arsenic, iron, and manganese are 0.01 mg/L, 0.30 mg/L, and 0.05 mg/L, respectively (USEPA 2017e).

The monitoring of the demonstration site has been ongoing for 8 years. Monitoring strategies include three parts; tracking hydraulic and organic loadings, using real-time soil sensor clusters to monitor soil conditions, and groundwater monitoring for verification that impact are not occuring. In addition, visual observations and selected soil sampling were conducted to qualitatively assess site conditions. In particular, areas that appeared to be less optimal for irrigating were delineated to evaluate the cause. Soil characteristics that were analyzed included texture, compaction, infiltration, uniformity, and localized water condition.

Current design criteria are based on limited evidence and have not been fully developed for food processing wastewater. As Table 17 shows, the characteristics of food processing wastewater vary depending on the type of plant and processes. The COD concentration ranges from 220 to 20,000 mg/L. Because of the diverse nature of the wastewater, it is challenging to develop design criteria. HYDRUS CW2D modeling may be a valuable design tool to simulate multiple operation strategies and predict the treatment performance, including carbon degradation, nitrification, and denitrification. The model result can provide operational strategies to maximize the treatment while minimizing environmental impacts.

4.2. Materials and methods

Unlike domestic wastewater, there is not a lot of literature on the application of food processing wastewater and that available is not consistent. A detailed summary of the literature found that the reported acceptable hydraulic loading ranged from 2.53 to 14.97 liter/m^2/day (2,700 to 16,000 gal/acre/day) and the organic loading ranged from 4.48 to 201.75 g/m^2/day (40 to 1,800 lb BOD/acre/day) (Mokma 2006). This study also found that there were no clear scientific basis for any of these values (Mokma 2006).

Consequently, this research first reports on the methodology used to demonstrate the feasibility of a food processing wastewater irrigation system and then examines the benefits compared to a traditional wastewater treatment plant. Design approaches using HYDRUS CW2D modeling were also explored.

4.2.1. Monitoring of food processing wastewater land application

Comprehensive monitoring has been continuously ongoing for over 8 years at the demonstration wastewater irrigation facility. Daily hydraulic and organic loadings were tracked. Multiple soil sensor clusters are installed to measure the soil's volumetric water content, oxygen level, and temperature. Groundwater quality was monitored quarterly. Characteristics of the soil such as its texture, compaction, infiltration, and localized high water condition were also monitored.

4.2.1.1. Background of demonstration site

Detail information of the demonstration site is described in below, which was extracted from the Water Environmental Federations Technical, Exhibition and Conference (WEFTEC) proceedings (Dong et al. 2017).

The overview map of the food processing wastewater land application demonstration site is shown in Figure 11. The size of the demonstration wastewater land application site is 675,825 m^2 (167 acres). The type of food processing plant at the demonstration site is canning. Corn and alfalfa are grown on the wastewater irrigation site and used as animal feed. The wastewater produced by the food processing plant varies in quantity and characteristics depending on processing at the production plant, technology, type of food, and facility. Before land application, the wastewater is screened and then flows into an aerated equalization tank (Figure 12). In this study, the food processing wastewater land application system refers to the sequential treatment of screening, equalization tank, and land application. Because the wastewater is applied to the surface using a center pivot irrigation system, localized runoff is collected by a Hickenbottom pipe (Fairfield, IA) (Figure 13), flows into the storage tank, and is then applied on secondary irrigation sites using solid set distributors. This ensures that no water is transported off site by surface movement. The soil types at this the site vary and includes loamy sand, sandy loam, and sand, depending on the location and depth. Average groundwater depth is 9.1 m (30 ft).

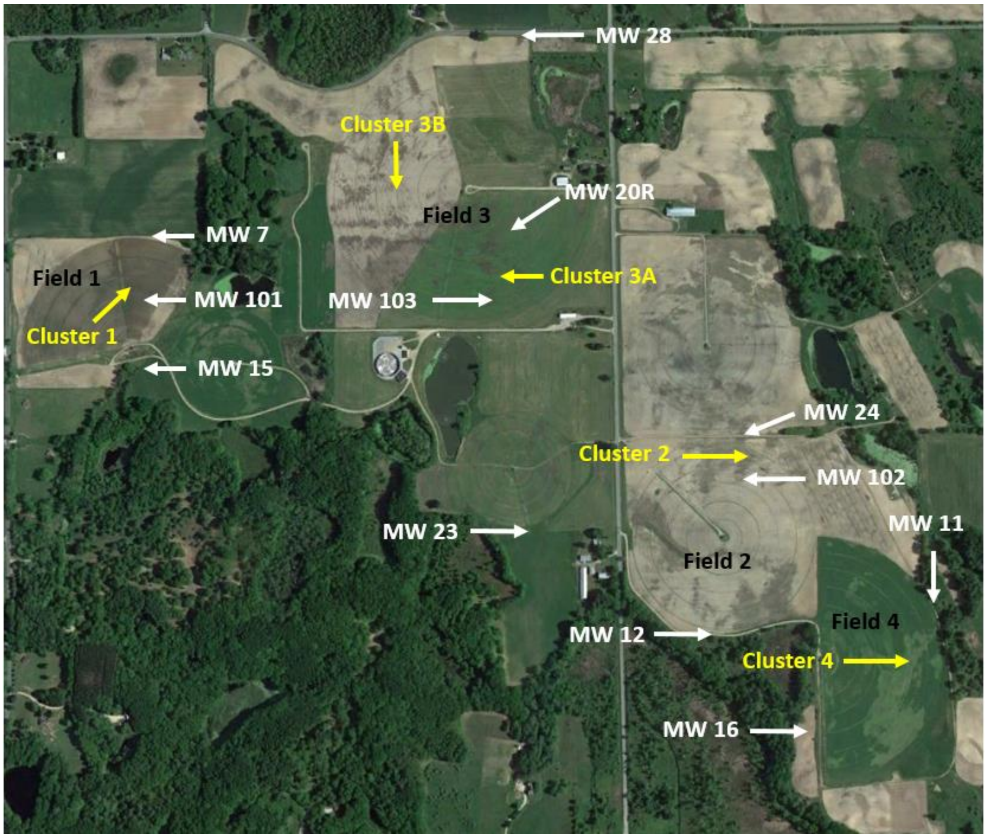


Figure 11. Overview map of demonstration site

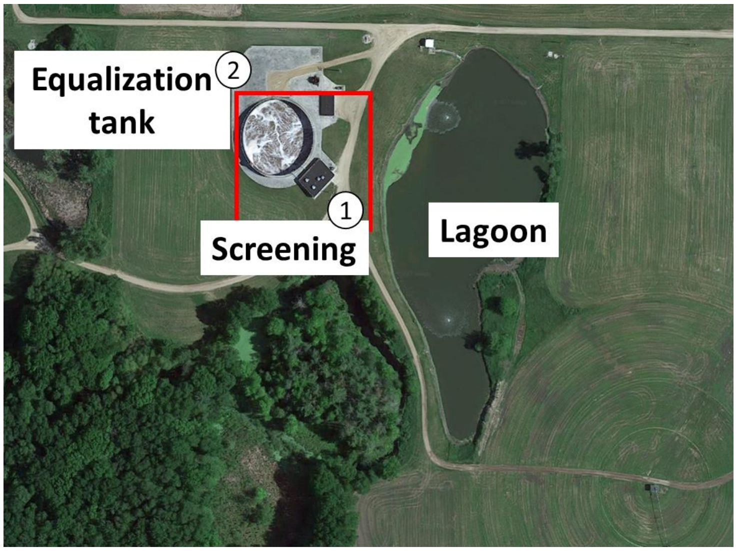


Figure 12. Wastewater flow of the demonstration site

Figure 13. Hickenbottom pipe for inducing and collecting run off

4.2.1.2. Monitoring strategies

Monitoring strategies included tracking hydraulic and organic loadings, observing soil conditions using soil sensor clusters, and testing groundwater quality. Each are discussed in the subsequent paragraphs.

4.2.1.2.1. Hydraulic and organic loadings

Hydraulic loading is tracked daily at the demonstration site. The influent wastewater is characterized biweekly. Organic loading was calculated by multiplying the hydraulic loading by the BOD_5 concentration. Table 18 show the average, maximum, and minimum characteristics of wastewater at the demonstration site.

Table 18. Characteristics of wastewater at the demonstration site

Parameters	**Average (mg/L)**	**Maximum (mg/L)**	**Minimum (mg/L)**
pH (in S.U.)	7.25	7.7	6.3
COD	1,156	2,900	405
BOD_5	651	1,480	153
Nitrogen, total kjeldahl	40.1	64.4	17.3
Ammonia-nitrogen	3.21	6.4	1.4
Nitrate-nitrogen	<0.1	3.3	0.4
Nitrite-nitrogen	0.4	0.4	0.4
Phosphorus, total (as P)	8.59	23.4	3.28
Sodium, total	59.1	416	28.6
Calcium, total	87	87	87
Iron, total	1.3	2.75	0.3
Magnesium, total	35.7	35.7	35.7
Manganese, total	0.1	0.18	0.06
Potassium, total	534.7	717	33.5
Chloride	342.1	526	196

4.2.1.2.2. Soil sensor cluster

A remote monitoring system consisting of five soil sensor clusters were used. The locations are shown in Figure 11. Figure 14 shows an overview of the soil sensor cluster. Each consist of a yagi antenna, surge protector, solar panel, 12v battery, CR 1000 datalogger, RF 401 radio, 3 soil response thermistor reference oxygen sensors, and 3 volumetric water content sensors. All of the parts were manufactured by and purchased from Campbell Scientific (Logan, UT) except for the soil response thermistor reference oxygen sensors, which are from Apogee instrument (Logan, UT)). Figure 15 shows the composition of the devices in the weather resistant enclosure. In order to communicate with the data loggers, each soil sensor cluster is fitted with a RF401A, 900MHz radio that transmits up to one mile with an omnidirectional antenna or up to 10 miles with a higher gain directional antenna. Regular maintenance was conducted twice a year, including examing sensor wire connections, moisture build up in the antenna connector, antenna direction, and solids accumulation on the surface of solar panel. Moisture in the antenna connector was found to interfere with the signal strength resulting in communication disruptions and occasional failures to the soil sensor cluster. The battery in each cluster was routinely replaced every 2 - 3 years to avoid catastrophic failures.

Figure 14. Overview of sensor cluster at the demonstration site

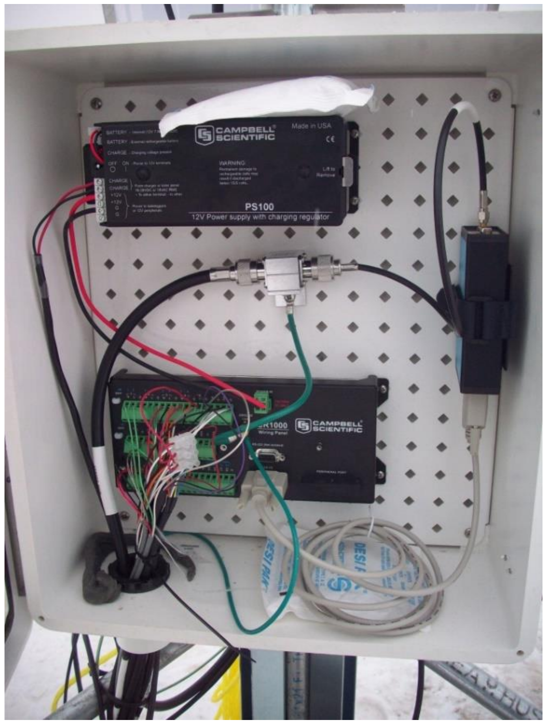


Figure 15. Composition of soil sensor cluster

The sensors were installed at depths of 30.48, 60.96, and 91.44 cm (1, 2, and 3 ft), as shown in Figure 16. Measurements were taken every 5 minutes and average daily values recorded. From these measurements, it can be determined if the soil is aerobic or anaerobic, which provides information on the potential for nitrate and heavy metal mobilization.

Figure 16. Sensors installed at depths

SO-110 soil response thermistor reference oxygen sensor were manufactured by Apogee (Logan, UT), designed for continuous gaseous oxygen measurement in air, soil/porous media, sealed chambers, and in-line tubing. The sensors consists of galvanic cell sensing element (electrochemical cell) Teflon membrane, and reference temperature sensor (Apogee Instruments 2016). The sensors measure from 0 to 100% O_2, and have a standard response time of 60 seconds.

The CS 616 water content reflectometer measures the volumetric water content from 0% to saturation in a soil using two 30 cm (11.8 in) stainless steel rods. The variability between probes is ±0.5% volumetric moisture content in dry soil and ±1.5% volumetric moisture content in typical saturated soil (Campbell Scientific 2014).

LoggerNet software, was developed by Campbell Scientific (Logan, UT) and was used for programing, communication, and data retrieval between the data logger and the computer. Each day LoggerNet downloads measurement data from all sensors and saves it in a CSV file format. A CSI Web Server, which was also developed by Campbell scientific (Logan, UT), allows users to view the saved data via the web browser.

4.3.1.2.3. Groundwater monitoring

Strategically positioned monitoring wells allow for the quarterly collection of groundwater samples for carbon, metal, and nutrient testing. In the specific study area, a total of 16 monitoring wells (MWs) (Figure 17) are upstream and downstream of each field. Locations of the MWs are shown in Figure 11. Domestic wells downstream of the site are also monitored to verify that the demonstration site does not contaminate neighbor's wells.

Figure 17. MW 103 at the demonstration site

4.2.1.3. Site evaluation

In order to evaluate the long-term site function, visual observation, soil characteristics, uniformity, and localized high water table conditions were conducted.

4.2.1.3.1. Visual observation

Quarterly to biannual visual observations and selected soil sampling were conducted to qualitatively evaluate site conditions. In particular, areas that appeared to be less optimal for irrigating were evaluated to determine the cause. Particular attention was focused on non-optimal areas as to identify potential causes. These areas are defined by standing surface water and low crop growth. Figure 19 shows examples of optimal and non-optimal areas. Delineation of non-optimal areas were conducted using a Juno 3B GPS Handheld (Figure 18), manufactured by Trimble Inc. (Sunnyvale, CA). The Juno 3B GPS Handheld is also a computer that can integrated a GPS and digital camera.

Figure 18. Juno 3B GPS Handheld

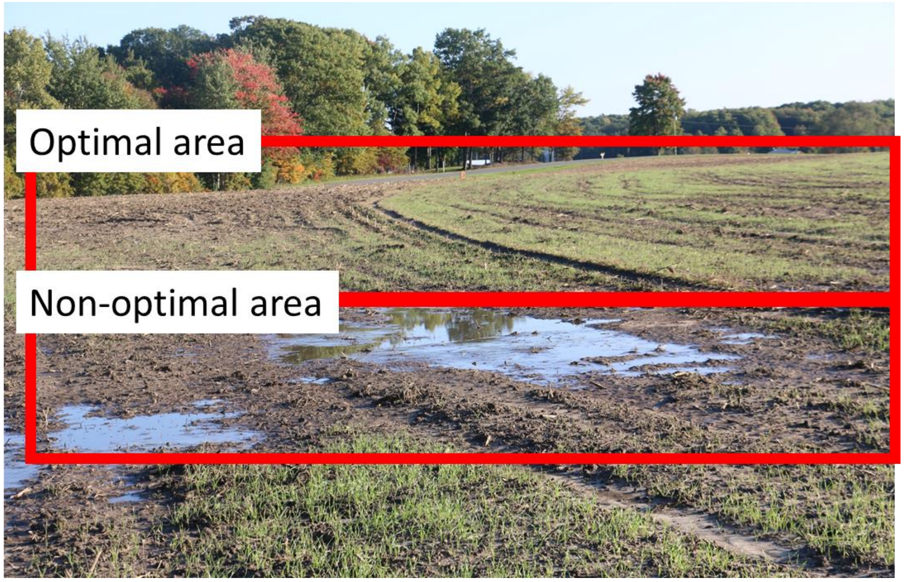


Figure 19. Optimal and non-optimal areas

4.2.1.3.2. Soil texture

Soil samples were collected at the surface, 30.48, 60.96, and 91.44 cm (1, 2, and 3 ft) depth of soil in optimal and non-optimal areas of study Fields 1, 2, and 3. Soil texture analysis was conducted via Michigan State University Soil and Plant Nutrient Laboratory (East Lansing, MI).

4.2.1.3.3. Soil compaction

Soil compaction analysis was conducted using a soil compaction tester, manufactured by AgraTronix (Figure 20) (Streetsboro, OH). This tester can measure up to 60.96 cm (24 in) depths and provide testing ranges such as green 0-1,378 kpa (0-200 psi) for good growing condition, yellow 1,378-2,068 kpa (200-300 psi) for fair growing condition, and red 2,068 kpa (300 psi) and above for poor growing condition.

Figure 20. Soil compaction meter by AgraTronix

4.2.1.3.4. Infiltration

Soil infiltration test was conducted using an ASTM 3385 double ring infiltrometer (Figure 21), manufactured by Turf-tec International (Tallahassee, FL). The 15.24 cm (6 in) ring was driven into the ground up to the 7.62 cm (3 in) mark by using a block of wood and mallet. Water was added up to 2.54 cm (1 in) , 444 mL, into the outer ring, then the inner ring. Once filled, the rings were covered by plastic wrap. The time to infiltrate 2.54 cm (1 in) of water was recorded. The outer ring helps the water in the inner ring flow down, not disperse to the side. Measurements were conducted on optimal and non-optimal areas for comparison. Statistical analysis was conducted using a t-test to compare the infiltration rate in optimal and non-optimal areas. The test was conducted using Rstudio software (Boston, MA) and the code is described in Appendix B.

Figure 21. Infiltrometer by Turf-tec International (Tallashassee, FL)

4.2.1.3.5. Uniformity

The central pivot irrigation nozzles can clog from suspended solids in the wastewater and freeze during cold weather. This can restrict the flow leading to non-uniform irrigation. Uniformity is critical as it is hypothesized that metal mobilization can result and impact ground water from small disjointed locations within the field, especially those that have compacted soil or localized high water table condition. Further, uniform application of wastewater is critical to plant health. Consequently, irrigation uniformity testing was conducted on fields 1 and 3.

Uniformity was conducted following the ANSI/ASAE S436.1 standard. The procedure includes placing 946 ml (32 oz) disposable soda cups at a 6.1 m (20 ft) distance apart in a straight line outward from the pivot elbow. Time was recorded from the point when water touches the cup from the pivot elbow to the point when the water stops hitting the same cup. The water level in each of the cup is measured and recorded.

The system uniformity coefficient is a numeric determination of the overall performance of even distribution in an irrigation system. Typically, the coefficient of 85 or higher is considered well distribution. The system uniformity coefficient of 80 below requires an adjustment to the sprinkler system (Kelley 2014).

4.2.1.3.6. Localized high water table condition

In order to observe localized high water table, a geoprobe (Figure 22) and auger were used. This equipment allowed for observations down to approximately 182.88 cm (6 ft).

Figure 22. Geoprobe

4.2.3. HYDRUS CW2D modeling

The procedure for model calibration and validation were performed following the method discussed in chapter 3.2.2.4. Minimum, average and maximum of COD concentration at the demonstration site were 405 mg/L, 1,156 mg/L, and 2,900 mg/L, respectively. Similar to the domestic wastewater modeling calibration process, two strengths were considered, Domestic/Food WW and Food WW from laboratory experiment. Average COD concentrations in Domestic/Food WW and Food WW were 661.5 mg/L and 2900 mg/L, respectively. Calibration using the Food WW condition was attempted but could not be completed due to the limited resource. Further research is needed. Therefore, the model was calibrated and validated using the Domestic/Food WW condition. Once the model was calibrated and validated, numerical goodness of fit was conducted to evaluate the model's performance. The procedure is described in Chapter 3.2.2.4.2.

4.2.3.1. Scenarios

In a food processing wastewater land application system, hydraulic and organic loadings are generally fixed by the food processing production facility, thus dosing frequency is the only operational parameter to maximize the treatment while protecting the environment. Different dosing frequency, including 3, 6, and 10 doses, were simulated. Multiple strength of hydraulic and organic loadings, 1, 2, and 3 times of strength, were also considered because wastewater composition and volumes from food processor is highly variable. Effluent concentrations of COD, ammonia, and nitrate were observed to understand the impact of dosing frequency and hydraulic and organic loadings on carbon degradation, nitrification, and denitrification.

Table 19 shows the time variable boundary conditions for multiple strengths of hydraulic and organic loadings for scenario simulation. Time variable boundary condition multiple dosing frequency were previously described in Table 7. Each loading flux was calculated by multiplying the original flux by the respective intensity of loading.

Table 19. Time variable boundary condition for multiple loading strengths

Time (day)	**1x loading flux (cm/day)**	**2x loading flux (cm/day)**	**3x loading flux (cm/day)**
0.31215	0	0	0
0.31250	1019.52	2039.04	3058.56
0.43020	0	0	0
0.43055	1019.52	2039.04	3058.56
0.53437	0	0	0
0.53472	1019.52	2039.04	3058.56
1	0	0	0

4.3. Result and discussion

Results and discussion for monitoring, benefits, and modeling of food processing wastewater land application are presented in the following sections.

4.3.1. Monitoring of food processing wastewater land application

The evaluation of monitoring strategies and site conditions at the long-term demonstration site are discussed.

4.3.1.1. Monitoring strategies

Monitoring strategies include tracking hydraulic and organic loadings, monitoring the soil condition via soil sensor clusters, and testing ground water quality.

4.3.1.1.1. Hydraulic and organic loading

Figure 23 shows the annual hydraulic loading in million gallons and organic loading in lbs of BOD_5/acre/day. A past concern at the demonstration site was metal mobilization resulting from the anaerobic conditions associated with excessive microbial growth due to the high concentration of the applied carbon. As a solution to the problem, implemented before MSU's involvement, hydraulic loading was lowered through a reduction in water use within the food processing plant. The decrease of hydraulic loading decreased metal mobilization at the site.

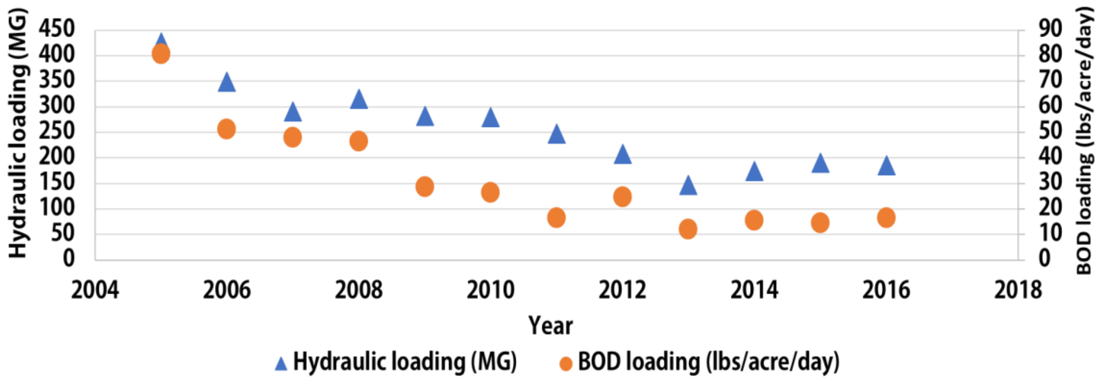


Figure 23. Hydraulic and organic loading at the demonstration site

4.3.1.1.2. Soil sensor cluster

To ensure that the soil remains aerobic to prevent metal mobilization, five soil sensor clusters were installed and have been monitoring the soil's volumetric water content, oxygen content, and groundwater quality for over 8 years. Figure 24 shows daily volumetric water content from soil sensor Cluster 1. Daily volumetric water shows that soils never reached saturation level (~30%), except briefly in early 2015. This indicates that the upper levels of the soil are generally aerobic, preventing the growth of metal reducing microorganisms.

Figure 25 shows the daily oxygen content for soil sensor Cluster 1. Daily oxygen level in all depths confirms that aerobic condition is maintained throughout the years. The generally aerobic upper level of soil, indicated by the dissolved oxygen concentration, prevents metal leaching but also limits denitrification. The oxygen sensors showed similar patterns as the moisture sensors and the greater expense and maintenance did not add significant value in having this more direct environmental measurement.

Figure 26 shows the temperatures at different depths in Field 1. It can be observed that the soil never freezes, even though typical soil outside of the irrigation zone is frozen to depths greater than 107 cm (42 in) during the winter. This indicates that the microbial population remains active year round.

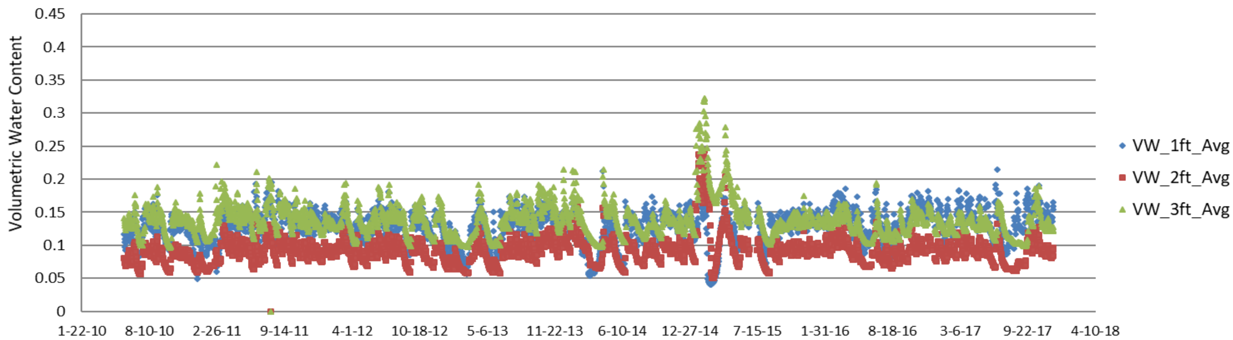


Figure 24. Daily volumetric water content at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 1

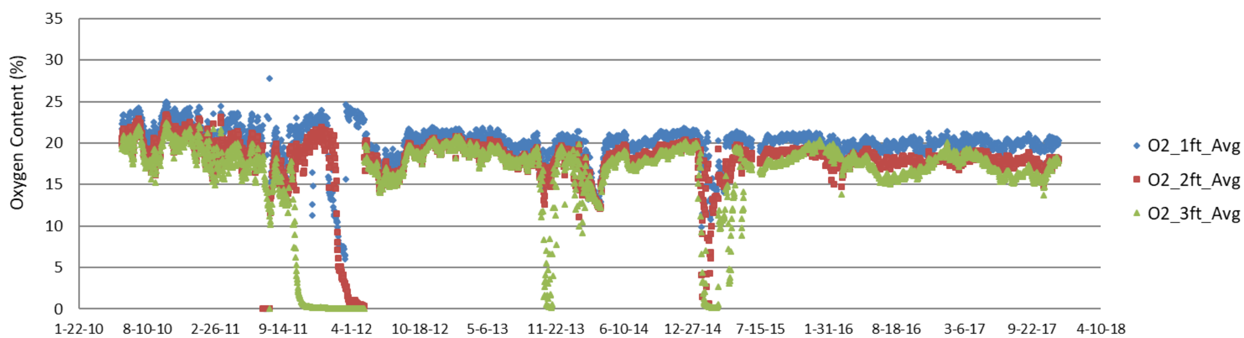


Figure 25. Daily oxygen level at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 1

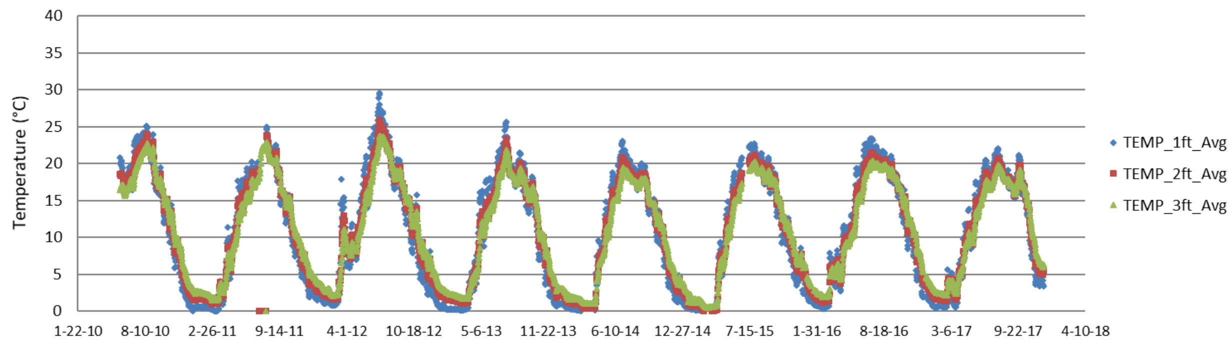


Figure 26. Daily temperature at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 1

Figure 27 shows daily volumetric water content for soil sensor Cluster 2. The daily volumetric water content was higher than Cluster 1. Figure 28 shows the daily oxygen content for soil sensor Cluster 2. The oxygen sensors in this cluster also malfunctioned at September in 2011 and November 2013. Regardless of depth, oxygen levels fluctuated and were consistently lower in comparison to the daily oxygen level data for Cluster 1. Soils with a higher volumetric water content and lower soil oxygen level are more likely to cause metal mobilization but less likely to encourage denitrification. Figure 29 shows the temperatures at different depths in Field 2. Similar to the results obtained in soil sensor Cluster 1, the soil temperature never falls below freezing. This indicates that microbial population within Field 2 remains active year round.

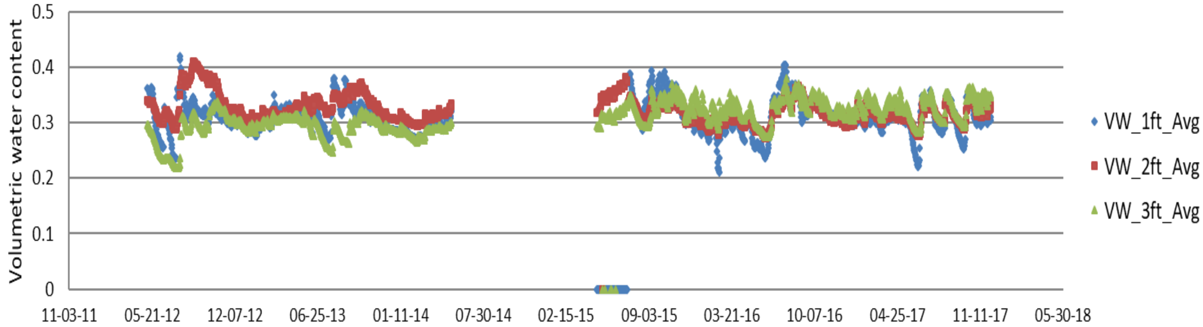


Figure 27. Daily volumetric water content at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths ror Cluster 2

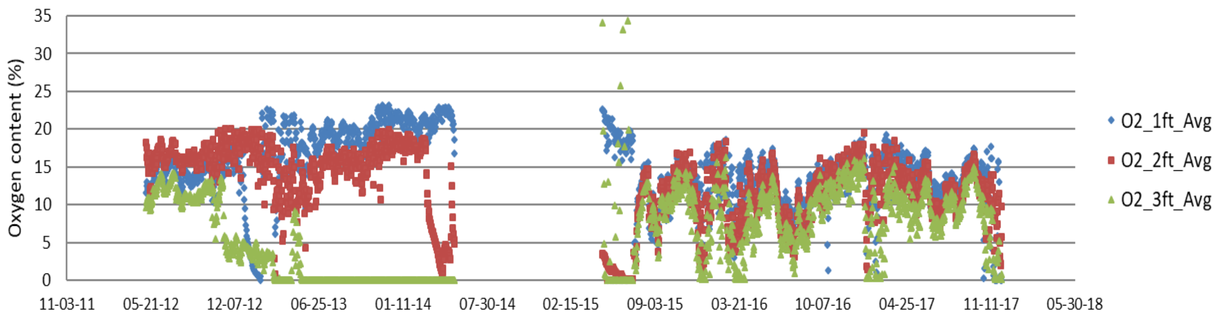


Figure 28. Daily oxygen level at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 2

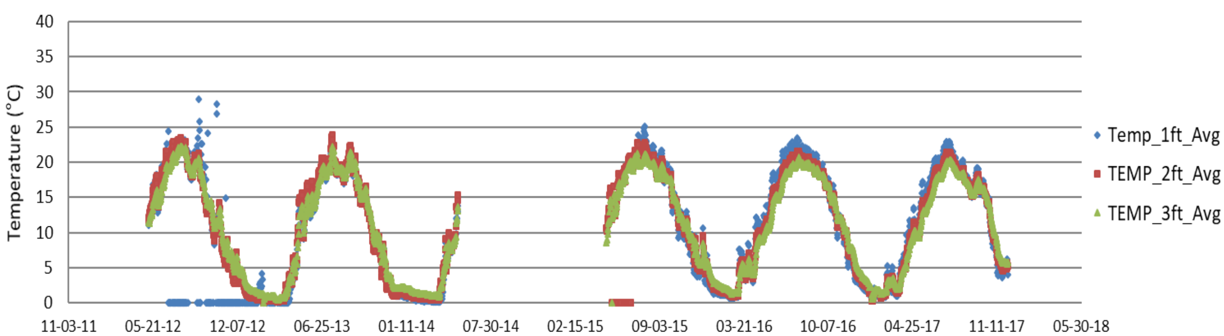


Figure 29. Daily temperature at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 2

Two soil sensor clusters are installed in Field 3. Soil sensor Cluster 3A is located on optimal condition of Field 3. Figure 30, 31, and 32 show the daily volumetric water content, oxygen content, and temperatures, respectively..

Figure 30 shows daily volumetric water content from soil sensor Cluster 3A. Daily volumetric water shows that soils reached saturation level (~30%) during a few time periods. This indicates that the upper levels of the soil are generally aerobic, preventing the growth of metal reducing microorganisms. Figure 31 shows the daily oxygen content from soil sensor Cluster 3A. Daily oxygen level in 30.48 (1 ft) depth shows aerobic condition. Interestingly, at a depth of 91.44 cm (3 ft), levels fluctuated and did not show the same trend as the volumetric water content. The oxygen sensor at a depth of 91.44 cm (3ft.) depth malfunctioned. The oxygen sensor also measure temperature. Figure 32 shows temperature at different depths in Field 3. It can be observed that the soil never freezes. This indicates that the microbial population remains active year round.

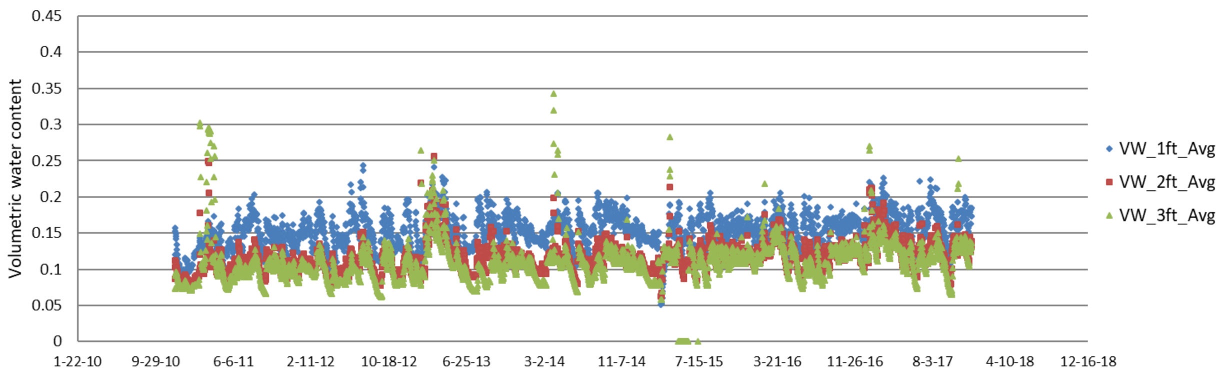


Figure 30. Daily volumetric water content at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 3A

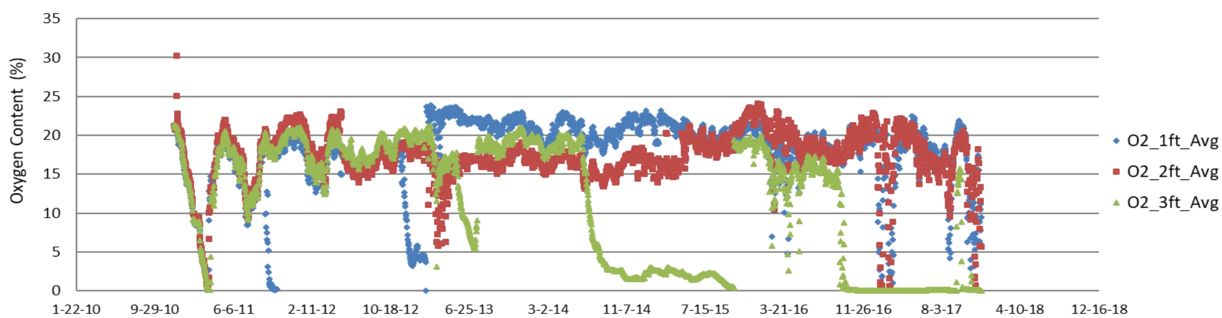


Figure 31. Daily oxygen level at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 3A

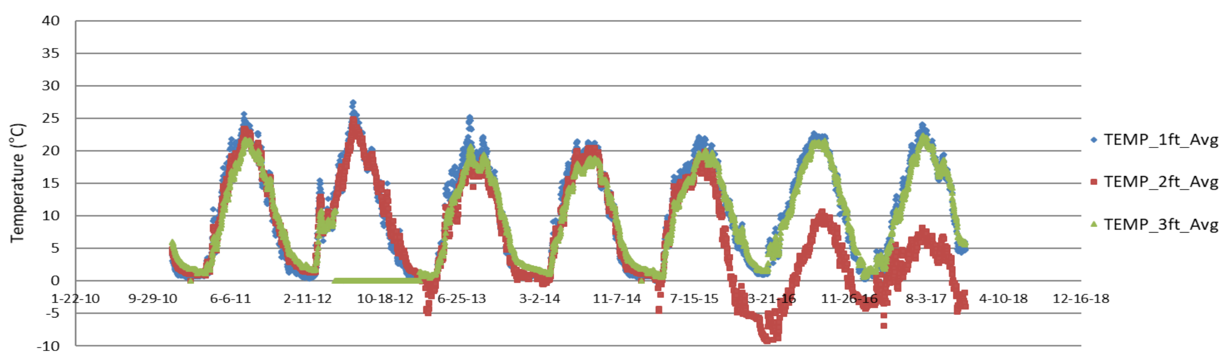


Figure 32. Daily temperature at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 3A

Cluster 3B was placed in the non-optimal area of Field 3. Figure 33 shows the daily volumetric water content. Interestingly, high levels of volumetric water contents were observed at all depths. One possible cause is the soil type, which was loamy sand at 30.48 and 61.96 cm (12 and 24 in) and sandy loam, at 76.2 cm (30 in). Because sandy loam has less porosity than loamy sand, the water flow might be restricted. Restriction of water flow results in a high volumetric water content. High levels of volumetric water indicates overall anaerobic conditions in the upper levels of soil, which could result in metal mobilization. Groundwater analysis in MW 103 (within Field 3) confirmed that manganese was present. However, anaerobic conditions are more conducive for denitrification.

Figure 34 shows the daily oxygen content from soil Cluster 3B. The oxygen content is generally lower, which indicates possible denitrification, but may promote metal mobilization. Temperatures at different depths for Cluster 3B are shown in Figure 35. Consistent temperatures above freezing indicate that the microbial population remains active year-round.

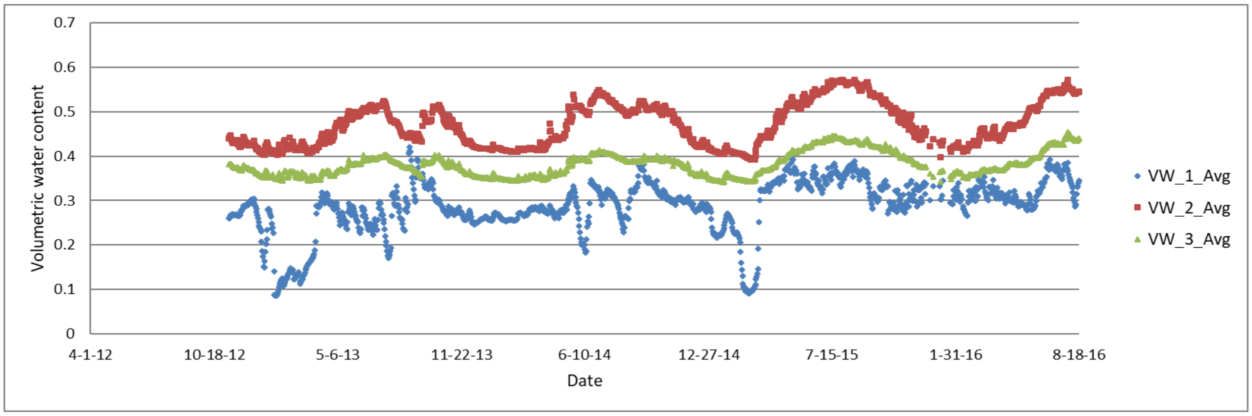


Figure 33. Daily volumetric water content at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 3B

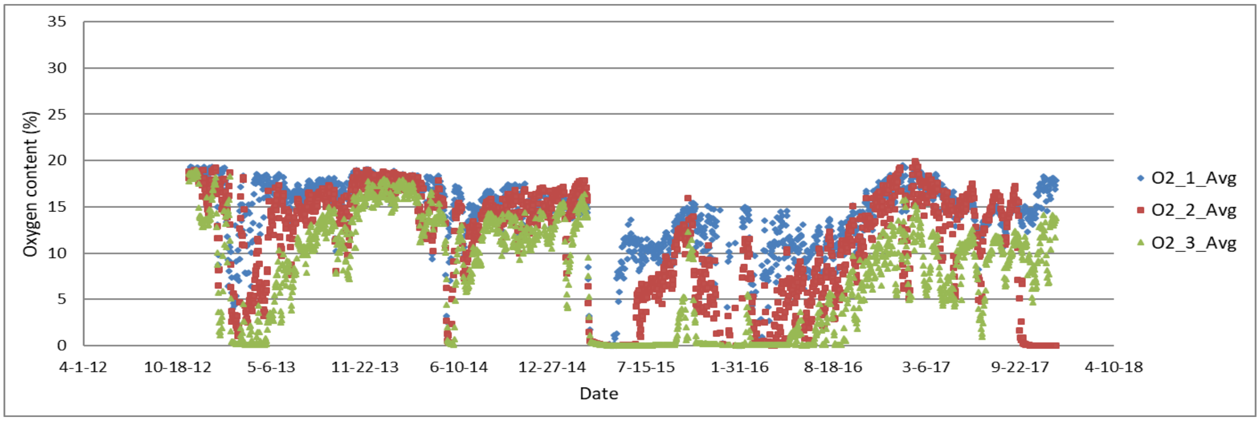


Figure 34. Daily oxygen level at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 3B

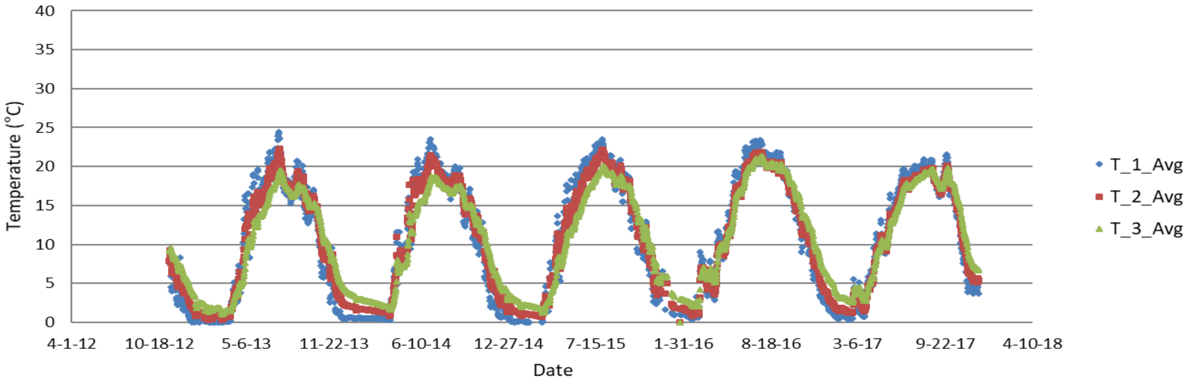


Figure 35. Daily temperature at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 3B

Figure 36 shows daily volumetric water content for soil Cluster 4. The volumetric water content at a depth of 30.48 cm (1 ft) were generally lower than the volumetric water content at depths of 61.96 and 91.44 cm (2 and 3 ft). The oxygen content (Figure 37) at a depth of 30.48 cm (1 ft) remained aerobic throughout the years, however, oxygen levels at 61.96 and 91.44 cm (2 and 3 ft) fluctuated. Fluctuation could have been caused by high localized water conditions. When the soil was excavated for the installation of the sensors, standing water and heavy wet soil were observed at a depth of 121.92 cm (4 ft), however, results from Chapter 4 indicate that treatment likely occurred before the water reached that depth. Figure 38 shows the temperatures at 30.48, 61.96, and 91.44 cm (1, 2, and 3 feet) depths from Cluster 4. Given that the soil never freezes, the microbial population remains active year round.

In summary, sensor readings at the demonstration site indicated a substantial range of soil moisture conditions and oxygen levels, indicating environments that were aerobic to anaerobic. The sensors were also very responsive. Moisture sensors proved to be good indicators if the soil was aerobic or anaerobic, based on a comparison to the oxygen sensors. In addition, the moisture sensors are less complex resulting in better reliability, less maintenance, and significantly less cost. Additionally, temperatures at the lower depths did not drop below freezing indicating that the microbiological environment was at least somewhat active all year. However, as temperature decreases, the activity of microorganisms decrease.

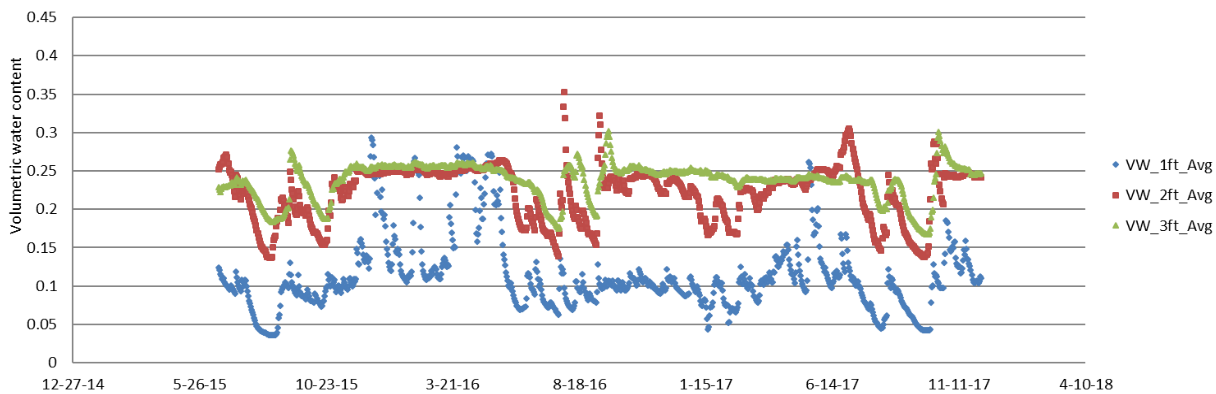


Figure 36. Daily volumetric water content at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 4

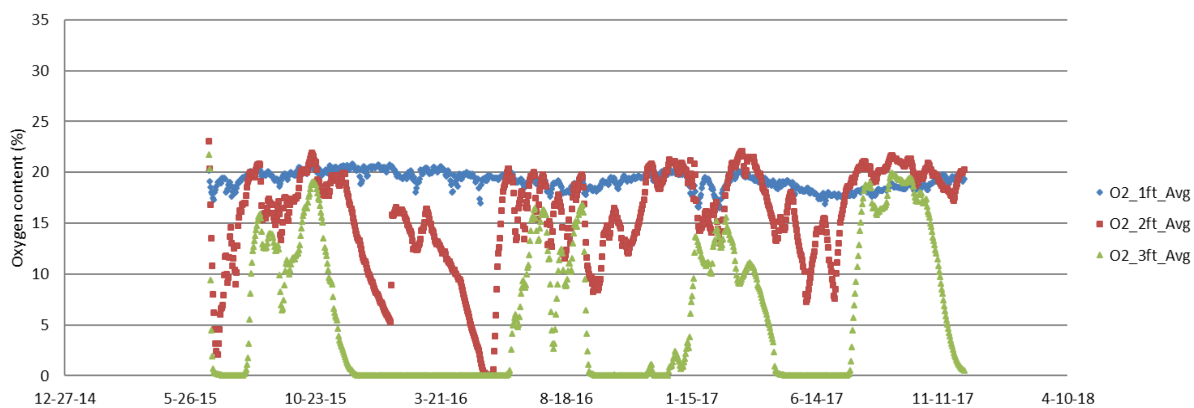


Figure 37. Daily oxygen level at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 4

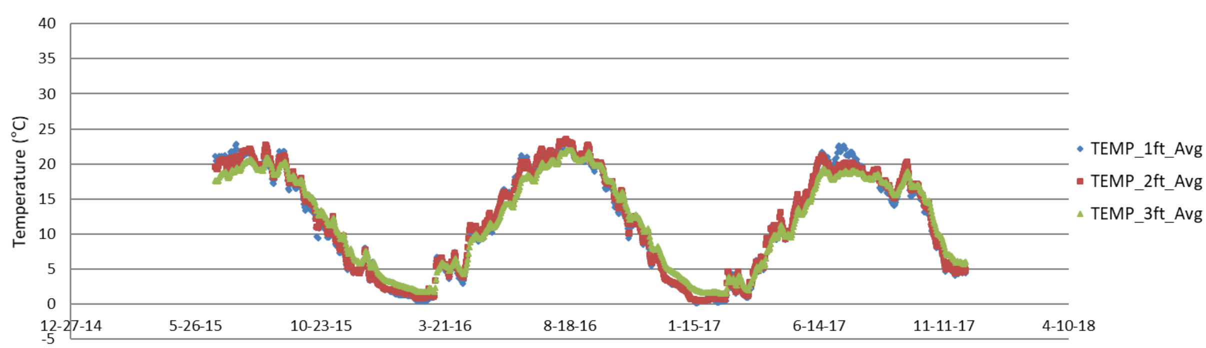


Figure 38. Daily temperature at 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths for Cluster 4

4.3.1.1.3. Groundwater monitoring

Figure 39 shows the estimated groundwater flow at the demonstration site. In general, the groundwater flows from the north to south.

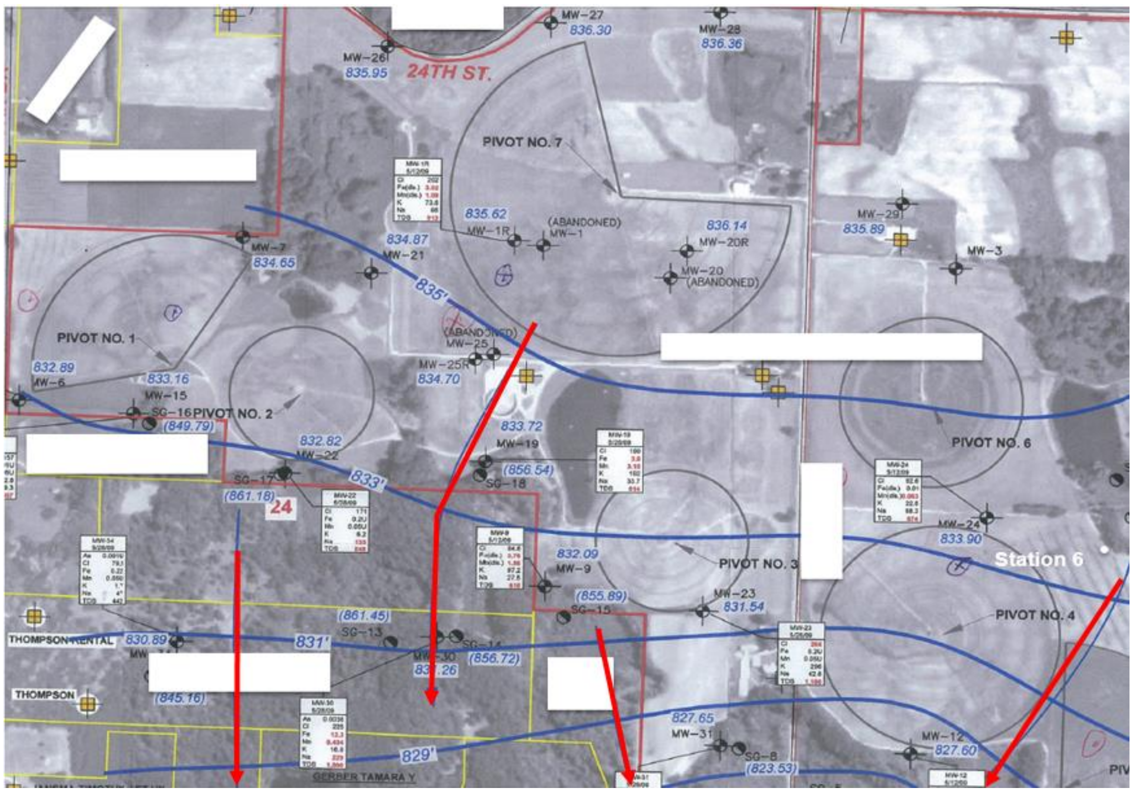


Figure 39. Estimated groundwater flow at the demonstration site

The demonstration site is located in an area with heavy agricultural activity, which may be one cause of the relatively high nitrate concentrations in the groundwater throughout the area. Figure 41, Figure 42, and Figure 43 shows the groundwater quality for MW 7 (upstream of Field 1), MW 101 (within Field 1), and MW 15 (downstream of Field 1), respectively. Monitoring well locations for Field 1 are shown in Figure 40. The nitrate concentration in MW 15 was generally equal to or less than 10 mg/L-N of nitrate. Interestingly, this indicates a decrease of nitrate as it progresses across the wastewater irrigation site. Metal mobilization into groundwater was not detected in Field 1.

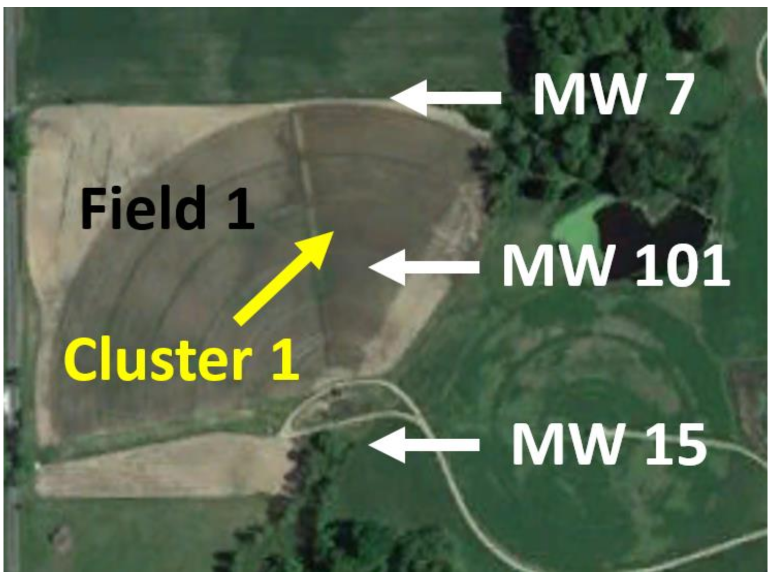


Figure 40. MW locations for Field 1

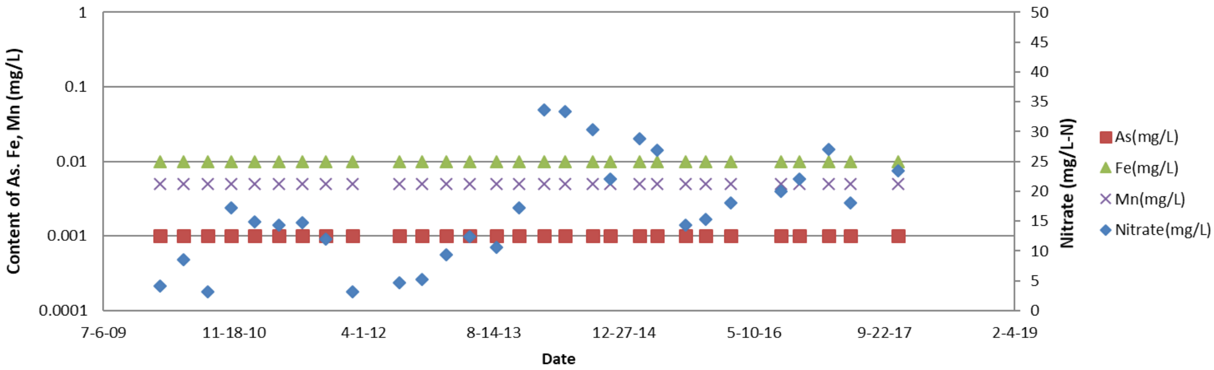


Figure 41. Groundwater quality for MW 7, upstream of Field 1

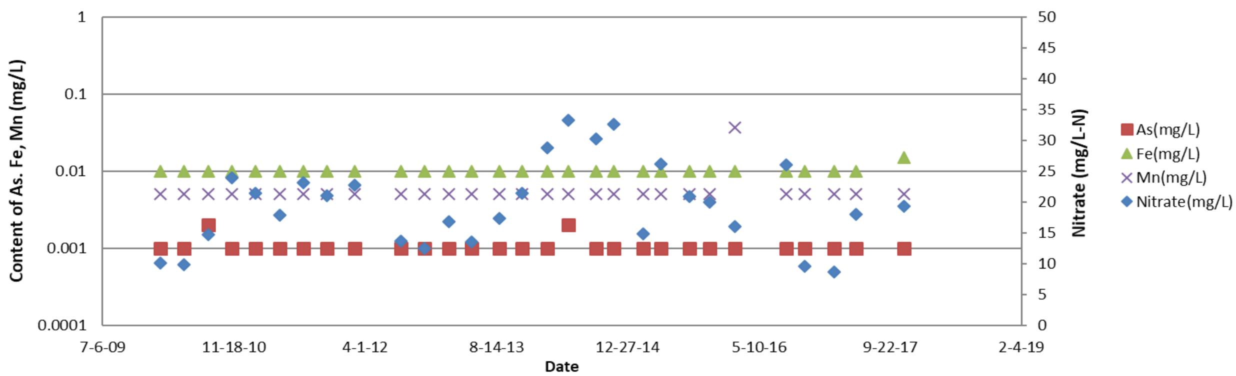


Figure 42. Groundwater quality for MW 101, within Field 1

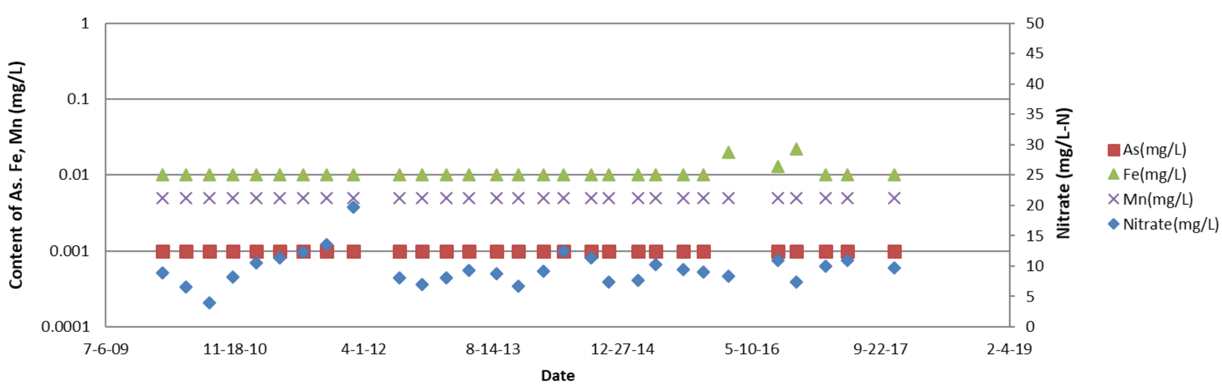


Figure 43. Groundwater quality for MW 15, downstream of Field 1

For Field 2, the groundwater quality of MW 24, (upstream of Field 2), MW 102 (within Field 2), and MW 12 (downstream of Field 2) are shown in Figure 45, Figure 46, and Figure 47, respectively. Monitoring well locations for Field 2 are shown in Figure 44. Relatively high nitrate concentrations were detected in MW 24 and MW 102. The nitrate concentrations in MW 12 generally maintained equal to or less than 10 mg/L-N of nitrate. This again indicates that nitrate levels were reduced across the site. Relatively high manganese concentrations were initially detected in MW 102 (within Field 2), but gradually decreased with time. This decrease correlates to the reduction of hydraulic loading. In general, low concentrations of arsenic, manganese, and iron were found in MW 12 (downstream of Field 2).

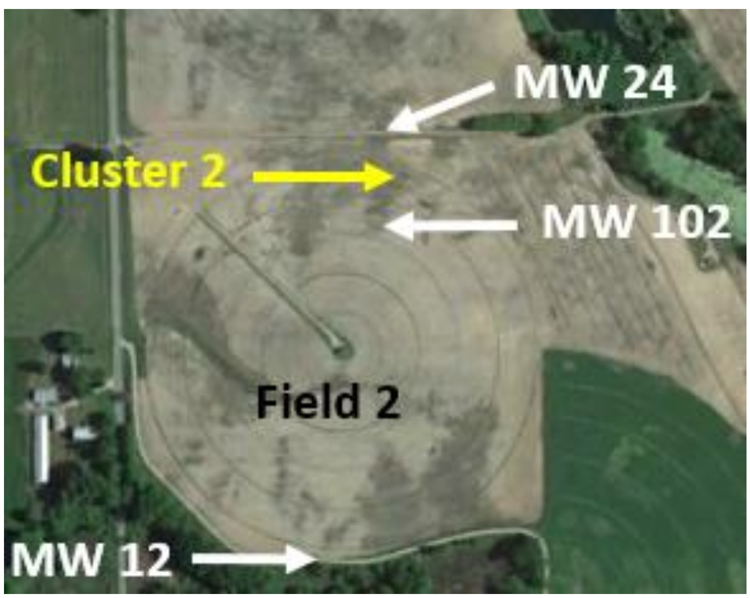


Figure 44. MW locations for Field 2

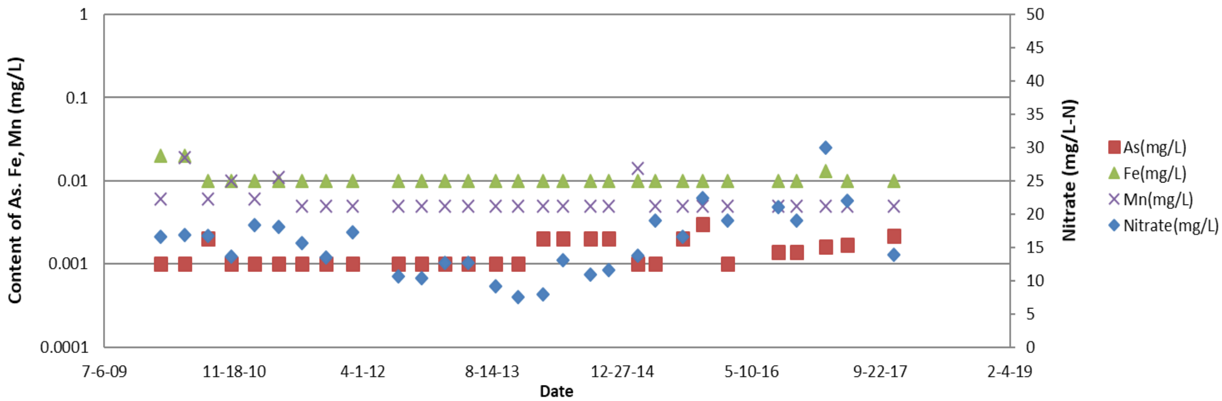


Figure 45. Groundwater quality for MW 24 (upstream of Field 2)

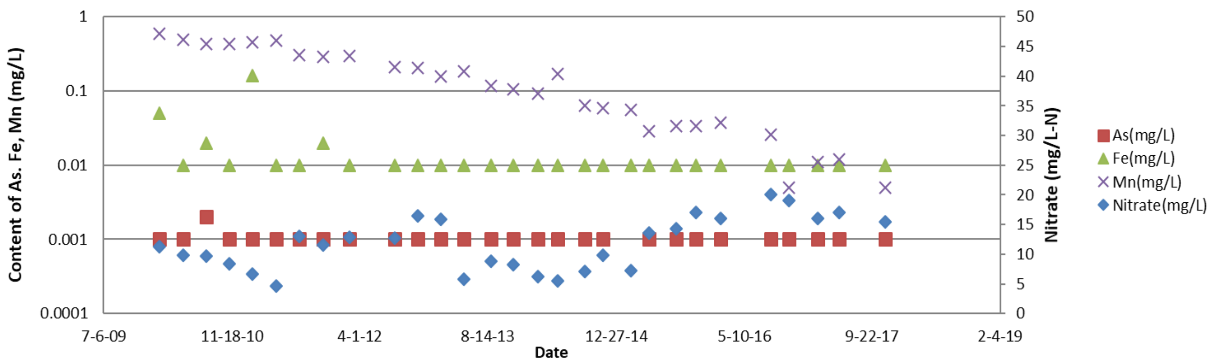


Figure 46. Groundwater quality for MW 102 (within Field 2)

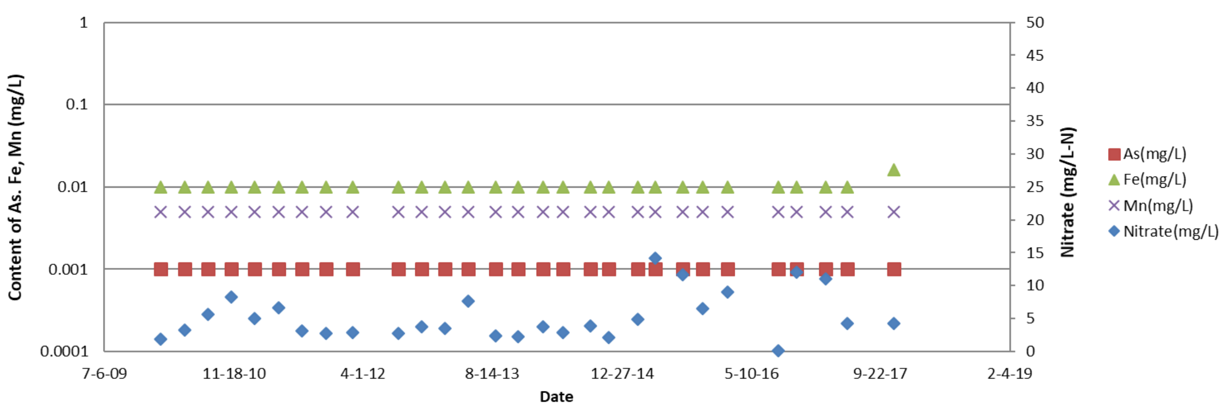


Figure 47. Groundwater quality for MW 12 (downstream of Field 2)

In Field 3, the groundwater quality of MW 28 (upstream of Field 3), MW 20R (within Field 3), MW 103 (within Field 3), and MW 23 (downstream of Field 3) are shown in Figure 49, Figure 50, Figure 51, and Figure 52, respectively. Monitoring well locations for Field 3 are shown in Figure 48. Relatively low metals and equal to or less than 10 mg/L-N of nitrate were detected in MW 28. Relatively high nitrate concentrations were detected in MW 20R. Possibilities include leaching from the field or the neighbor's farming practice. Figure 54 highlights the area owned by the neighbor. Initially high iron concentration in MW 103 was detected but the concentration decreased over time. MW 103 had an average manganese concentration of 1.5 mg/L. Manganese was not detected in MW 23, located downstream of Field 3 (Figure 52). This MW also had a relatively high nitrate concentration.

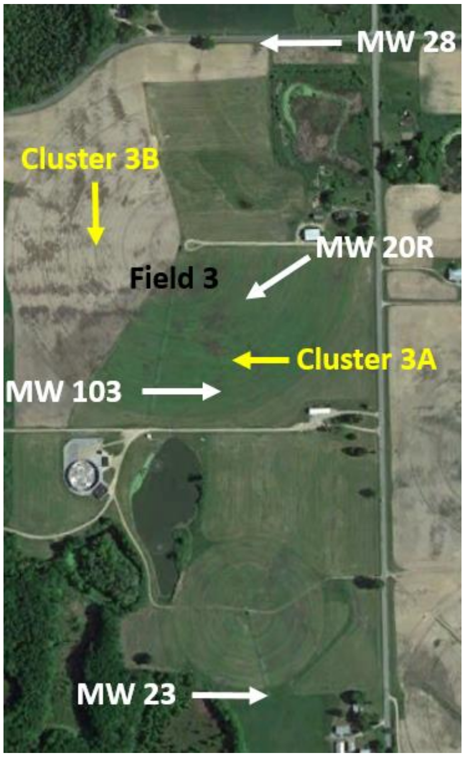


Figure 48. MW location in Field 3

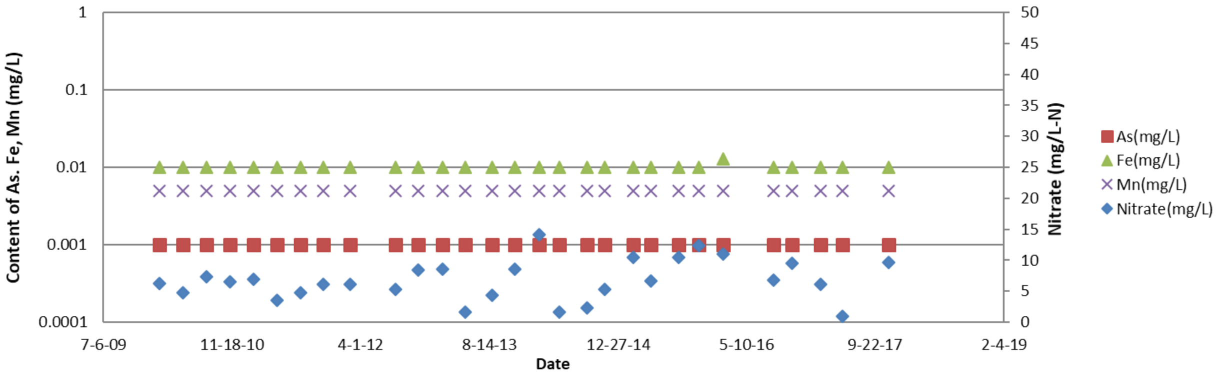


Figure 49. Groundwater quality for MW 28 (upstream of Field 3)

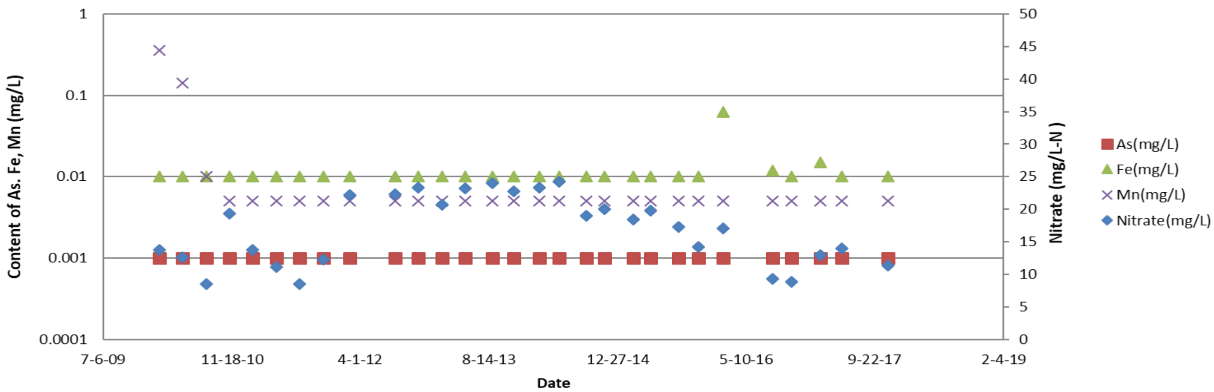


Figure 50. Groundwater quality for MW 20R (within of Field 3)

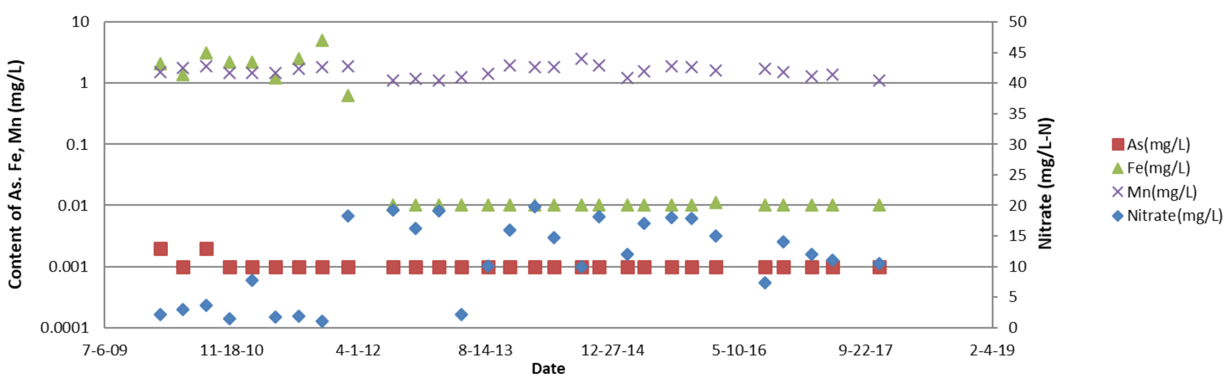


Figure 51. Groundwater quality for MW 103 (within of Field 3)

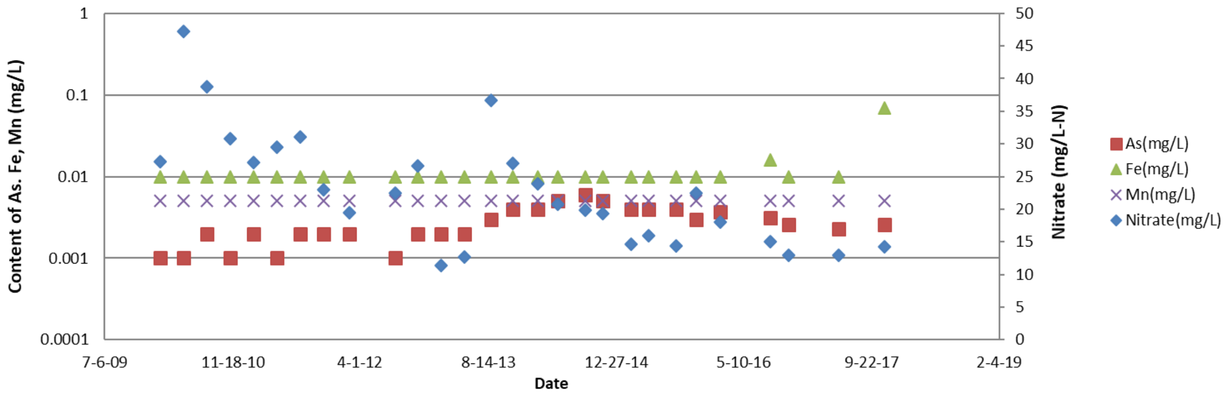


Figure 52. Groundwater quality for MW 23 (downstream of Field 3)

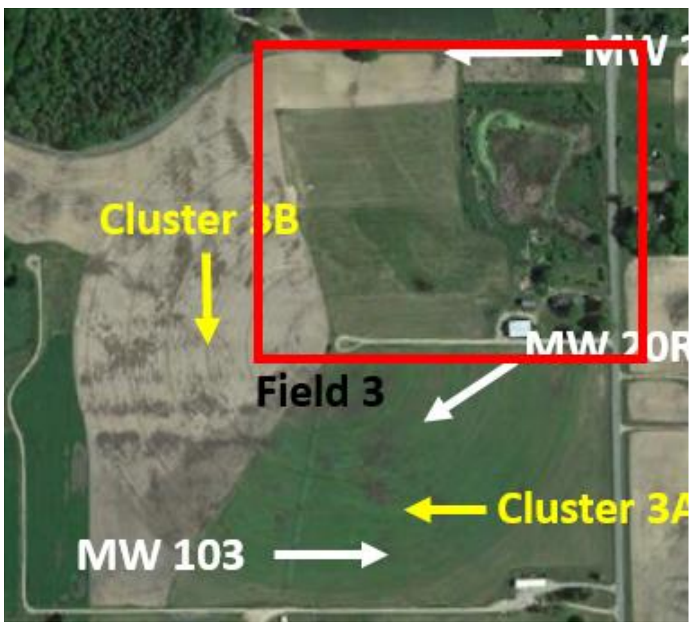


Figure 53. Area, owned by neighbor

In Field 4, the groundwater quality of MW 11 (upstream of Field 4) and MW 16 (downstream of Field 4) are shown in Figure 55 and Figure 56, respectively. Monitoring well locations for Field 4 are shown in Figure 54. Relatively high nitrate concentrations were detected in MW 11. In contrast, nitrate concentration in MW 16 were generally equal to or less than 10 mg/L-N of nitrate. Once again, the concentration of nitrate decreased across the site. The average manganese concentration was0.058 mg/L in MW 16, which is slightly higher than USEPA secondary drinking water standard.

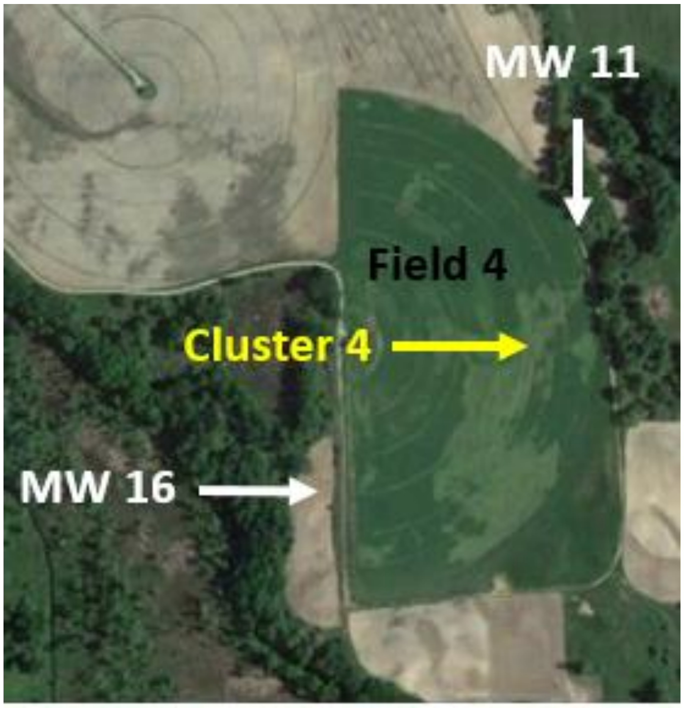


Figure 54. MW locations for Field 4

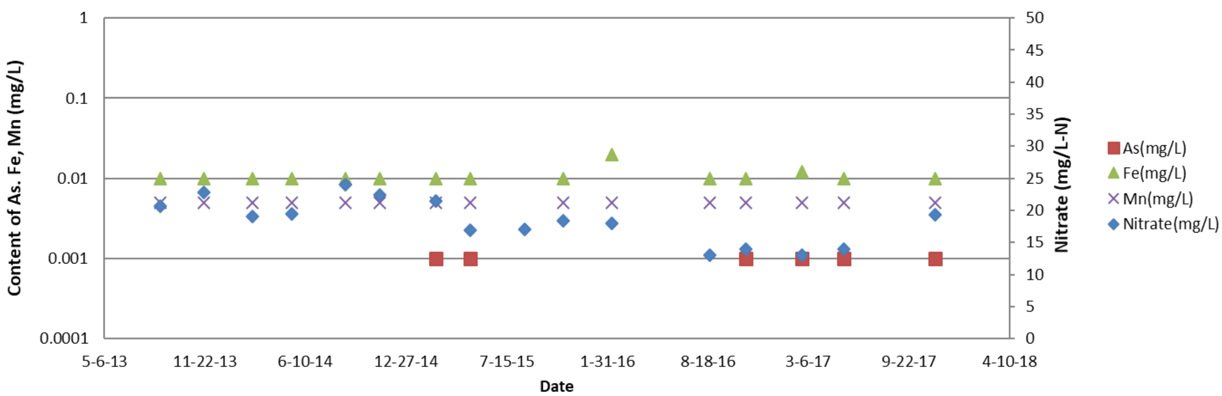


Figure 55. Groundwater quality for MW 11 (upstream of Field 4)

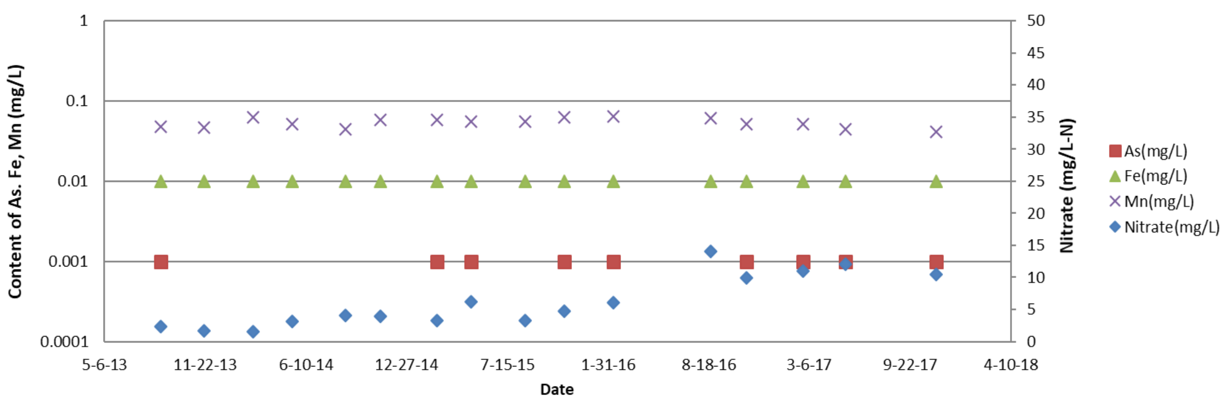


Figure 56. Groundwater quality for MW 16 (downstream of Field 4)

Nitrate monitoring has proven to be difficult to understand because of the complex site hydrogeology. Interesting to note that, in 2017, the average nitrate concentration in upstream wells was 10.4 mg/L, while downstream wells was 8.8 mg/L. Significant nitrate from area farms is clearly impacting the groundwater coming onto the demonstration site. Further study on understanding of nitrogen process at the demonstration site is needed while considering plant uptake. Pretreatment for nitrogen removal should also be considered. A preliminary study indicates that a substantial amount of nitrogen and carbon can be removed by pretreating the wastewater using coagulation/flocculation (data is not shown).

In summary, monitoring hydraulic and organic loadings, groundwater quality, and soil conditions using sensors has proven to be a safe and effective method to minimize environmental impacts from the land application of food processing wastewater. Further, this data has also proved to be important for making operational decisions.

4.3.1.2. Site evaluation

Although monitoring can provide a general overview of site conditions, additional evaluations were performed to delineate non-optimal areas based on the presence of standing water and poor crop growth. To understand the causes, soil analyses, uniformity, and localized high water condition were monitored at several of these locations.

4.3.1.2.3. Visual observation

Non-optimal areas were delineated for Fields 1, 2, and 3 and are shown in Figure 57, Figure 58, and Figure 59, respectively, as delineated in red Figure 60 show a photograph of a non-optimal area. Some of the non-optimal areas were possibly due to low elevations.

Figure 57. Delineated (red) areas on Field 1

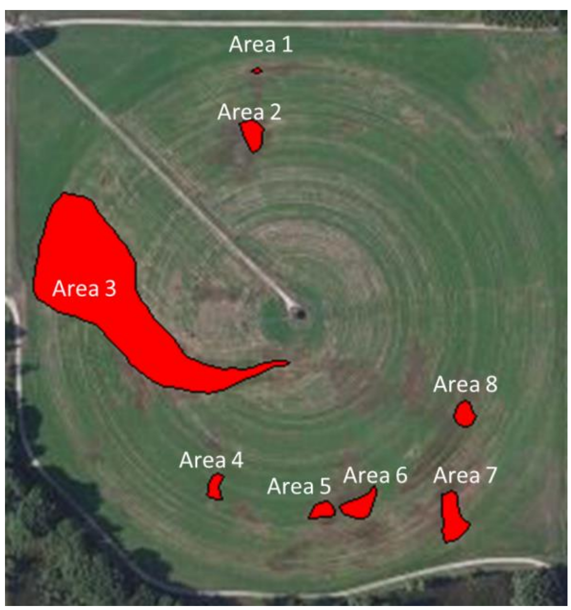


Figure 58. Delineated (red) areas on Field 2

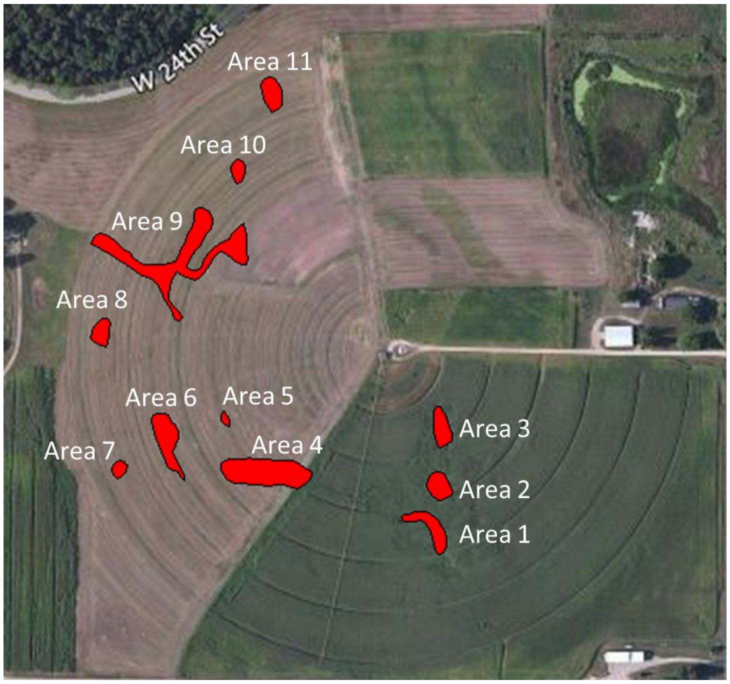


Figure 59. Delineated (red) areas on Field 3

Figure 60. Non-optimal areas

4.3.1.2.2. Soil texture

Soil sampling was conducted at the surface, 30.48, 61.96, and 91.44 cm (1, 2, and 3 ft) depths. Soil sample could not be collected in Field 2 at 91.44 cm (3 ft) depth because of the high level of compaction. Results show that soil textures differ in each field but there was no significant difference between optimal and non-optimal areas. Consequently, soil texture does not explain the cause of non-optimal areas. Table 20 shows the texture for non-optimal and optimal areas.

Table 20. Texture for non-optimal and optimal areas

Depth	Field 1		Field 2		Field 3	
	Non-optimal	Optimal	Non-optimal	Optimal	Non-optimal	Optimal
Top	Loamy Sand	Loamy Sand	Sandy Loam	Sandy Loam	Loamy Sand	Loamy Sand
30.48 cm (1 ft)	Loamy Sand	Loamy Sand	Sandy Loam	Sandy Loam	Loamy Sand	Loamy Sand
61.96 cm (2 ft)	Sand	Sand	*Sandy Loam	**Sandy Loam	Loamy Sand	Loamy Sand
91.44 cm (3 ft)	Sand	Sand	N/A	N/A	***Sandy Loam	Sand

* 50.8 cm (20 in) deep
** 71.12 cm (28 in) deep
*** 76.2 cm (30 in) deep
N/A: Not available

4.3.1.2.3. Soil compaction

Soil compaction was measured on both optimal and non-optimal areas up to a 61.96 cm (2 ft) depth. Figures 61, Figure 62, and Figure 63 show the locations of these measurements and Tables 21, 22, and 23 show soil compact result for Field 1, 2, and 3, respectively. A, B, and C designations in each area represent location that were randomly chosen. Overall, there was no clear difference between optimal areas and non-optimal in regard to soil compaction.

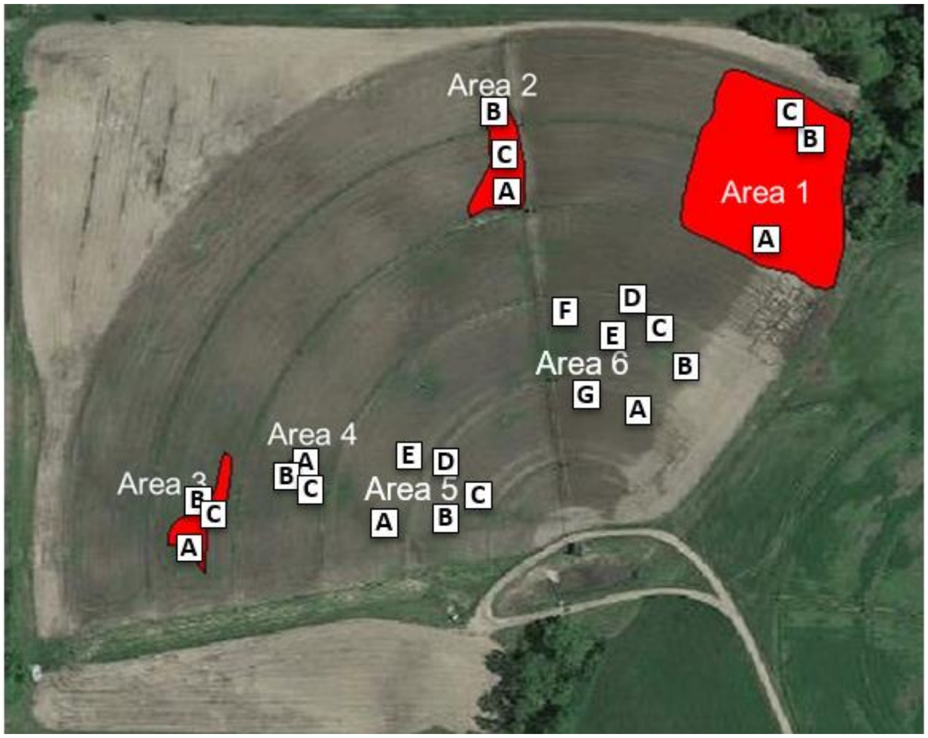


Figure 61. Locations of soil compaction analysis in Field 1

Table 21. Soil compaction result using soil compaction meter in Field 1

Depth	Non-optimal areas												Optimal areas											
	Area 1			Area 2			Area 3			Area 4			Area 5					Area 6						
	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	D	E	A	B	C	D	E	F	G
15.24 cm (6 in)	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G
30.48 cm (12 in)	R	R	R	Y	Y	R	R	R	G	R	R	R	R	Y	Y	R	R	R	Y	R	Y	Y	R	R
45.72 cm (18 in)	R	Y	Y	R	R	R	R	R	R	R	Y	R	R	R	Y	R	R	R	R	R	R	R	R	R
60.96 cm (24 in)	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R

(Green = 0-1,378 kpa (0-200 psi), Yellow = 1,378-2,068 kpa (200-300 psi), Red = > 2,068 kpa (300 psi))

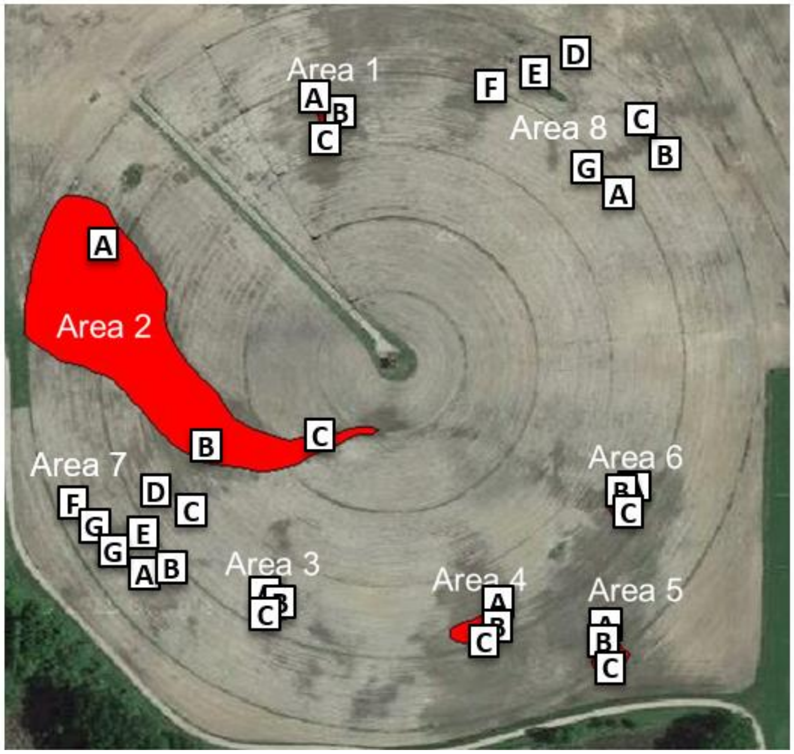


Figure 62. Locations of soil compaction analysis in Field 2

Table 22. Soil compact result using soil compaction meter in Field 2

Depth	Non-optimal areas																		Optimal areas												
	Area 1			Area 2			Area 3			Area 4			Area 5			Area 6			Area 7						Area 8						
	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	D	E	F	A	B	C	D	E	F	G
15.24 cm (6 in)	G	R	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	G	Y	G	G	G	Y	G	G	G
30.48 cm (12 in)	R	Y	R	R	Y	R	R	R	R	R	R	R	G	R	R	R	G	Y	R	Y	Y	R	R	R	Y	Y	R	Y	Y	R	R
45.72 cm (18 in)	Y	R	R	R	R	R	R	R	R	R	R	R	R	R	Y	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
60.96 cm (24 in)	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R

(Green = 0-1,378 kpa (0-200 psi), Yellow = 1,378-2,068 kpa (200-300 psi), Red = > 2,068 kpa (300 psi))

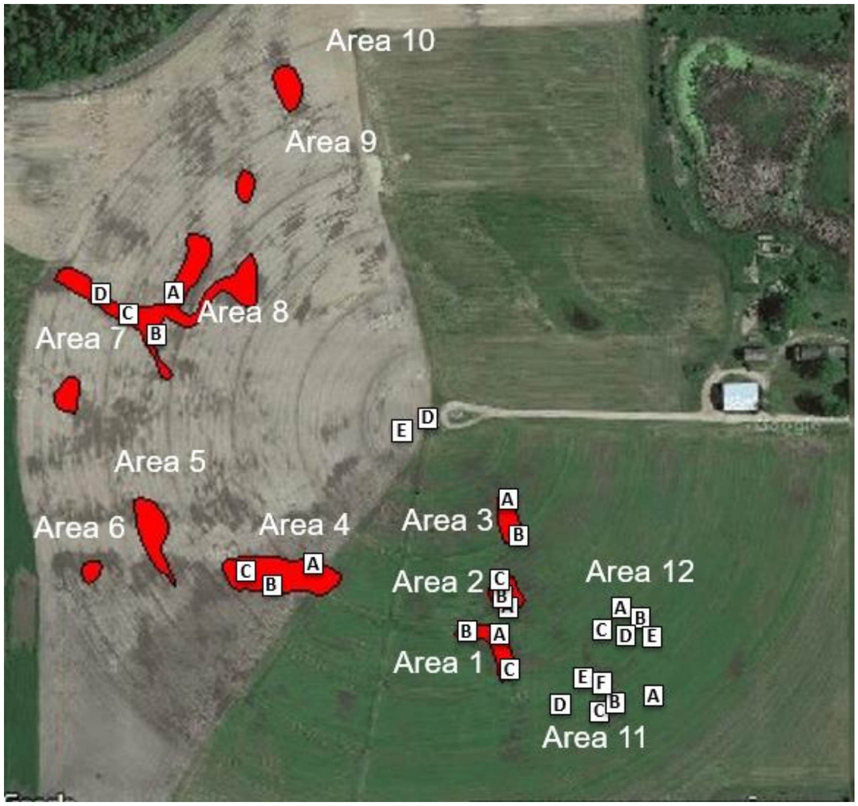


Figure 63. Locations of soil compaction analysis in Field 3

Table 23. Soil compact result using soil compaction meter in Field 3

Depth	**Non-optimal areas**															**Optimal areas**										
	Area 1			**Area 2**			**Area 3**		**Area 4**			**Area 8**				**Area 11**						**Area 12**				
	A	**B**	**C**	**A**	**B**	**C**	**A**	**B**	**A**	**B**	**C**	**A**	**B**	**C**	**D**	**A**	**B**	**C**	**D**	**E**	**F**	**A**	**B**	**C**	**D**	**E**
15.24 cm (6 in)	R	Y	G	G	G	Y	G	G	Y	R	G	Y	G	G	Y	G	G	G	G	G	G	G	G	G	Y	G
30.48 cm (12 in)	R	R	R	R	R	R	Y	R	R	R	R	R	R	R	R	R	R	Y	Y	Y	Y	Y	R	R	R	R
45.72 cm (18 in)	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
60.96 cm (24 in)	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R

(Green = 0-1,378 kpa (0-200 psi), Yellow = 1,378-2,068 kpa (200-300 psi), Red = > 2,068 kpa (300 psi))

4.3.1.2.4. Infiltration

Infiltration analysis was conducted in locations randomly selected in optimal and non-optimal areas on Field 1 (Table 24). Statistically, there was no different between optimal area and non-optimal area (p-value 0.6116, α=0.05).

Table 24. Infiltration analysis in Field 1.

Location	**Infiltration rate (in/sec)**	
	Optimal area	**Non-optimal area**
1	524	664
2	330	450
3	534	720
4	414	456
5	534	450
6	762	328
Average	516.3	511.3
Standard deviation	133.1	136.0

4.2.1.2.5. Uniformity

The water level in each of the cup distributed throughout the field to measure uniformity was measured and recorded. Figures 64 and 65 show the results for Fields 1 and 3. The average volume for Field 1 and 3 were 69 mL and 42 mL, respectively, indicating great variability. The impacts of uneven uniformity can be significant, such as poor plant health and metal mobilization from small disjointed locations. The system uniformity coefficient is a numeric measure of the overall performance of an irrigation system (Kelley 2014). The system uniformity for Fields 1 and 3 were 61 and 54, respectively. A system uniformity coefficient of 85 is considered a uniform distribution. Below a coefficient of 80 requires the sprinkler heads to be adjusted. Further study to address uneven distribution of irrigation system is needed. If suspended solids clog the nozzle, pretreatment may be required to reduce the suspended solids.

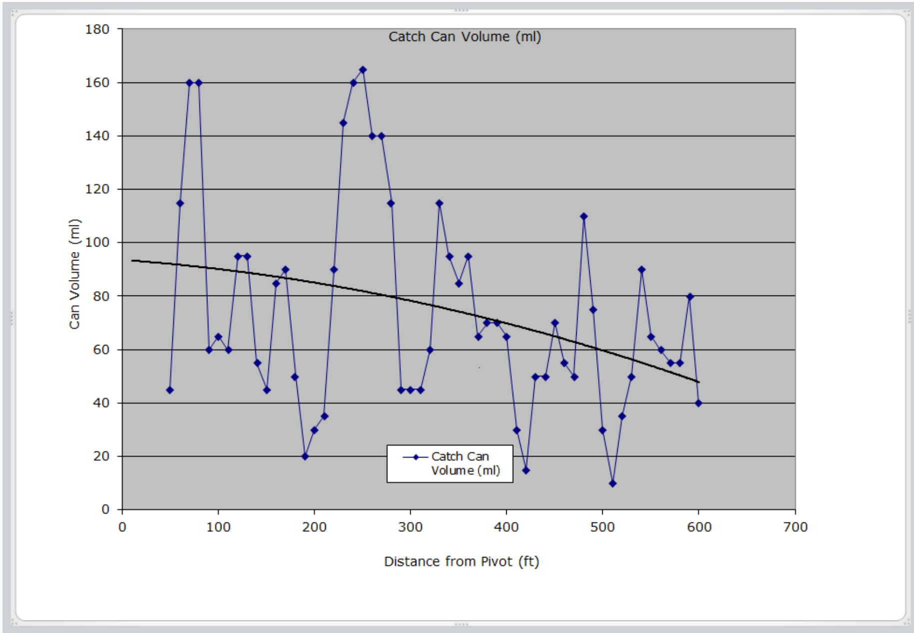


Figure 64. Catch Can Volume for Field 1

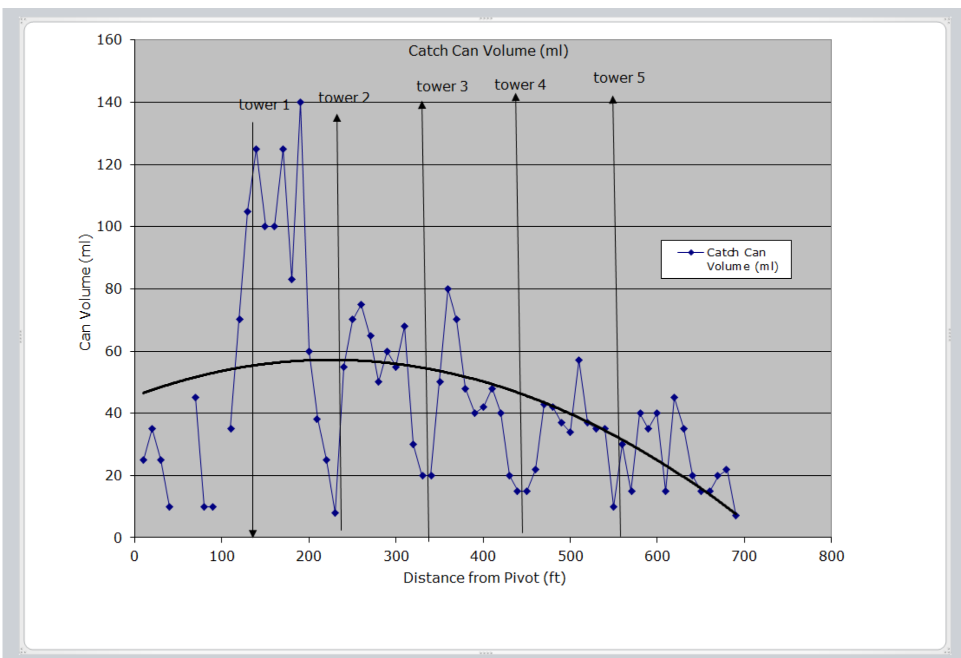


Figure 65. Catch Can Volume for Field 3

4.3.1.2.6. Localized high water condition

Observation of localized high water condition was conducted for Fields 2 and 3 (Figure 66 and Figure 67, respectively). Figure 68 shows localized high water condition found in Location 1 in Field 3. Table 25 shows the results of the depth of the high localized water condition and the total depth the geoprobe drove in the soil. The soils at Areas 1, 3, and 5 of Field 2 were compacted, especially in Location 1. The geoprobe could only proceed 50.8 cm (20 in) into the soil for that location. Localized high water conditions in Field 2 were observed depths of 121.92 cm (48 in), 142.24 cm (56 in), and 157.48 cm (62 in) for location 2, 3, and 4, respectively.

Two locations were selected in the Field 3 for testing. In both locations, free water at the bottom of the bore hole was found. This indicates that the localized high water tables at Areas 3 and 4 were both less than 121.92 cm (4 ft). Because locations 2, 3, and 4 in Field 2 and Areas 1 and 2 in Field 3 are within the non-optimal areas, localized high water condition may be a potential cause of non-optimal conditions.

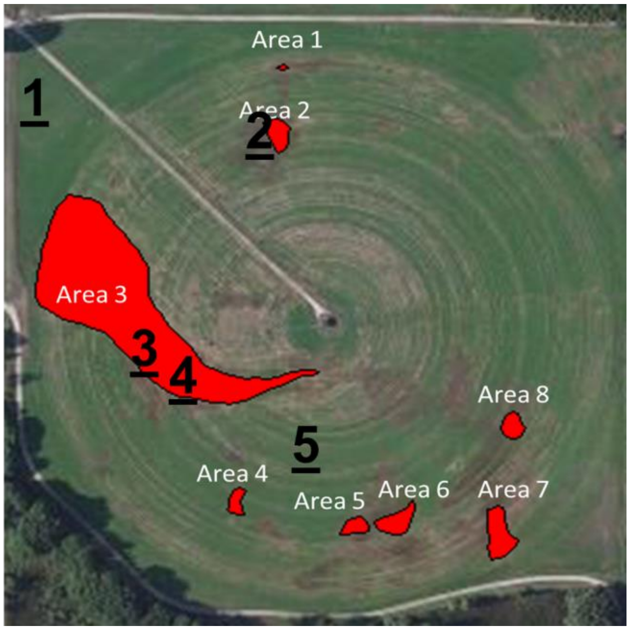


Figure 66. Locations for high localized water condition analysis in Field 2

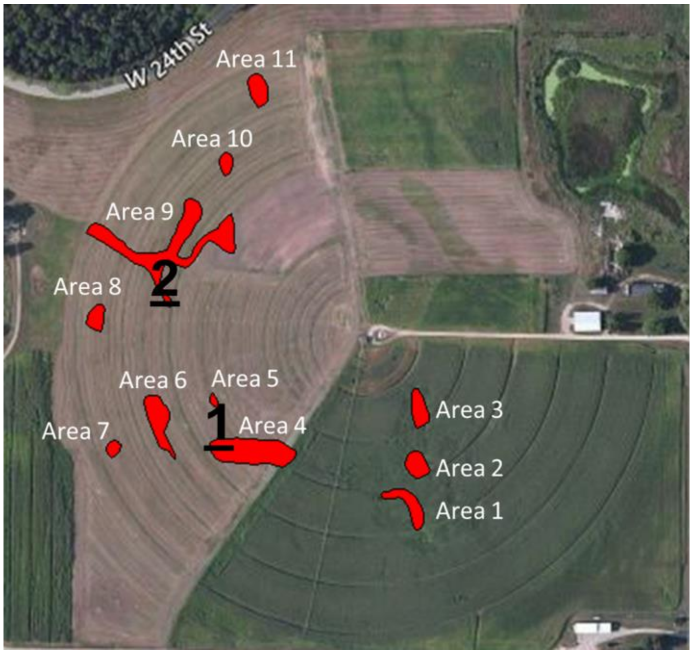


Figure 67. Locations for high localized water condition analysis in Field 3

Figure 68. High localized water condition found in location 1 on Field 3

Table 25. High localized water condition analysis for Fields 2 and 3

Field	**Location**	**High localized water condition depth**	**Total depth**
2	1	N/A	50.8 cm (20 in)
	2	< 121.9 cm (48 in)	121.9 cm (48 in)
	3	< 142.2 cm (56 in)	142.2 cm (56 in)
	4	< 157.4 cm (62 in)	157.4 cm (62 in)
	5	N/A	121.9 cm (48 in)
3	1	< 121.9 cm (48 in)	121.9 cm (48 in)
	2	< 121.9 cm (48 in)	121.9 cm (48 in)

N/A – not available

In summary, non-optimal locations were identified at the demonstration site and analyses were conducted to understand the cause. Site characteristic for non-optimal conditions were evaluated, including soil texture, compaction, infiltration, non-uniformity irrigation, and localized high water condition. Interestingly, soil texture, compaction, and infiltration rate did not appear to be the primary cause. The soil at the demonstration site are generally compacted, likely from driving heavy equipment on the wet soil. Drying the fields is desirable to prevent compaction but resting the field is a challenge because of the fixed hydraulic loading coming from the food processing plant. Precipitation may also delay the drying process. Low areas in the field combined with localized high water conditions may be problematic, although more research is required. Uniformity testing shows that the water is unevenly distributed, which is typical for center pivot irrigation systems, especially under the circumstances at the demonstration site. The most likely cause was excessive moisture content as many of the non-optimal locations were in lower areas within the field and contained localized high water conditions.

Runoff is kept on site so environment risks are eliminated but the practice of catching and redistributing the water increases maintenance and expenses. An alternative is the use of variable irrigation, to reduce water loading in non-optimal areas. Cover crops can tolerate high water contents. Soil amendments such as gypsum, and biochar may help on absorbing water and nutrients.

4.3.3. HYDRUS CW2D modeling

The result of model calibration, validation, and scenario simulation to observe impact of dosing frequency on the performance of treatment are provided.

4.3.3.1. Model calibration and validation

Model calibration was conducted by inverse modeling using volumetric water content measurement data. Figure 69 shows the comparison of measured and fitted HYDRUS volumetric water content values. The measured volumetric water content data of Domestic/Food WW condition in the first two dosing were used for calibration and the remaining measured data was used for model validation.

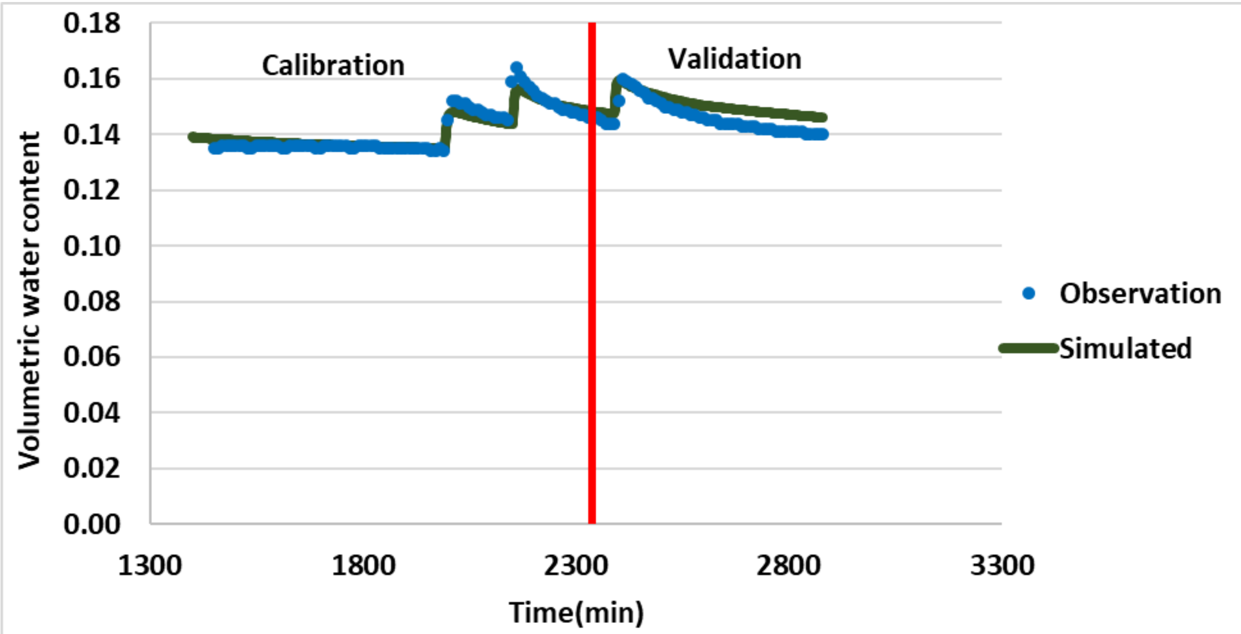


Figure 69. Fitted HYDRUS to measured value using volumetric water content from Domestic/Food WW

Calibrated and validated water flow model was evaluated by E, IA, and RMSE. As shown in Table 26, all the values were met according to the quality of model criteria, E > 0.5, IA > 0.8, and RMSE <0.014.

Table 26. Goodness of fit result for calibration and validation

Model	**Goodness of fit**		
	E	**IA**	**RMSE**
Calibration	0.9	0.96	0.002
Validation	0.61	0.88	0.002

Table 27 shows adjusted HYDRUS CW2D parameters for food processing wastewater land application modeling. Based on literature review, six parameters were considered for calibration process of the solute flow. After using the trial and error method, the maximum aerobic growth rate of XANs, Maximum denitrification rate of XH, and Fraction of CI in biomass lysis were adjusted.

Table 27. Adjusted HYDRUS CW2D parameters for food processing wastewater land application modeling

Parameter	**Description**	**Unit**	**Standard**	**Adjusted**
k_h	Hydrolysis rate	1/d	3	-
b_h	Lysis rate for XH	1/d	0.4	-
$b_{ANs/b}$	Lysis rate for XANs/b	1/d	0.15	-
μ_{ANs}	Maximum aerobic growth rate of XANs	1/d	0.9	0.45
μ_{dn}	Maximum denitrification rate of XH	1/d	4.8	4
$f_{BM,CI}$	Fraction of CI in biomass lysis	1/d	0.02	0.01

Table 28 shows the simulated effluent concentrations before and after the fitting process. The table contains averages and standard deviations of measured influent, effluent, and simulated values using standard and adjusted parameters from the calibration process. The measured values from day 108 to 170 (62 days) in Domestic/Food WW were used.

Table 28. Simulated effluent concentrations before and after the fitting process for calibration

Condition	Inf/Eff	Type of value	HYDRUS CW2D Parameters	COD (mg/L)	Ammonia (mg/L)	Nitrate (mg/L)
Domestic/Food WW	Influent	Measured Avg. (Std.)		619.0 (142.3)	28.9 (3.2)	1.0 (0.6)
	Effluent	Measured Avg.(Std.)		48.1 (7.2)	0.5 (0.2)	29.4 (4.6)
		Simulated	Standard	59	0.6	24.6
		Simulated	Adjusted	50.3	0.25	29.6

Table 29 shows the relative differences between measured and simulated values in the calibration process. The COD, ammonia, and nitrate simulated values using standard parameters in Domestic/Food WW condition are different than the measured values. Thus, parameters such as maximum aerobic growth rate of XANs, maximum denitrification rate of XH, and fraction of CI in biomass lysis were adjusted. After adjustments were made, the relative difference of COD, ammonia, and nitrate in Domestic/Food WW decreased from -18.47% to -4.37%, from 67% to 0%, and from 19.51% to -0.68%, respectively.

Table 29. Relative difference between measured and simulated values for calibration

Condition	HYDRUS CW2D	COD	Ammonia	Nitrate
Domestic/ Food WW	Standard	-18.47%	67%	19.51%
	Adjusted	-4.37%	0%	-0.68%

Table 30 the simulated effluent concentrations before and after the fitting process for validation. The table contains average and standard deviations of measured influent, effluent, and simulated values using standard and adjusted parameters in the validation process. The measured values from day 171 – 225 (54 days) in Domestic/Food WW wastewater were used.

Table 30. Simulated effluent concentrations before and after the fitting process for validation

Condition	Inf/Eff	Type of value	COD (mg/L)	Ammonia (mg/L)	Nitrate (mg/L)
Domestic/Food WW	Influent	Measured Avg (Std)	746.5 (111.1)	28.0 (4.5)	1.1 (0.9)
	Effluent	Measured Avg (Std)	49.3 (13.1)	0.1 (0.1)	30.9 (3.3)
		Simulated	52.1	0.27	27.9

Table 31 show the relative differences between measured and simulated values in the validation process. Except ammonia, the largest relative difference between measured and simulated was 10.75%. Although the large relative difference is observed for ammonia, the difference between 0.1 and 0.27 is not significant with respect to field conditions.

Table 31. Relative difference between measured and simulated values for validation

Condition	HYDRUS CW2D	COD	Ammonia	Nitrate
Domestic/Food WW	Adjusted	-5.37%	-63.0%	10.75%

4.3.3.2. Scenario – Treatment performance enhancement

Potential nitrate leaching into groundwater is a concern for food processing wastewater land application systems. Monitoring results from the demonstrate site showed the difficulty in determining its fate for a complex, large site. HYDRUS CW2D was conducted to observe the fate of nitrate for current operating conditions and the potential impact of dosing frequency. Multiple strength of hydraulic and loadings along with dosing frequency were observed. A nitrate effluent concentration on the 150th day at 60.96 cm (2 ft) depth of soil was observed. As

Figure 70 shows, the COD effluent concentration did not significantly change after 120 days as indicated by the very low slope of -0.001. In Figure 71, the growth of heterotrophic bacteria also did not change after 120 days.

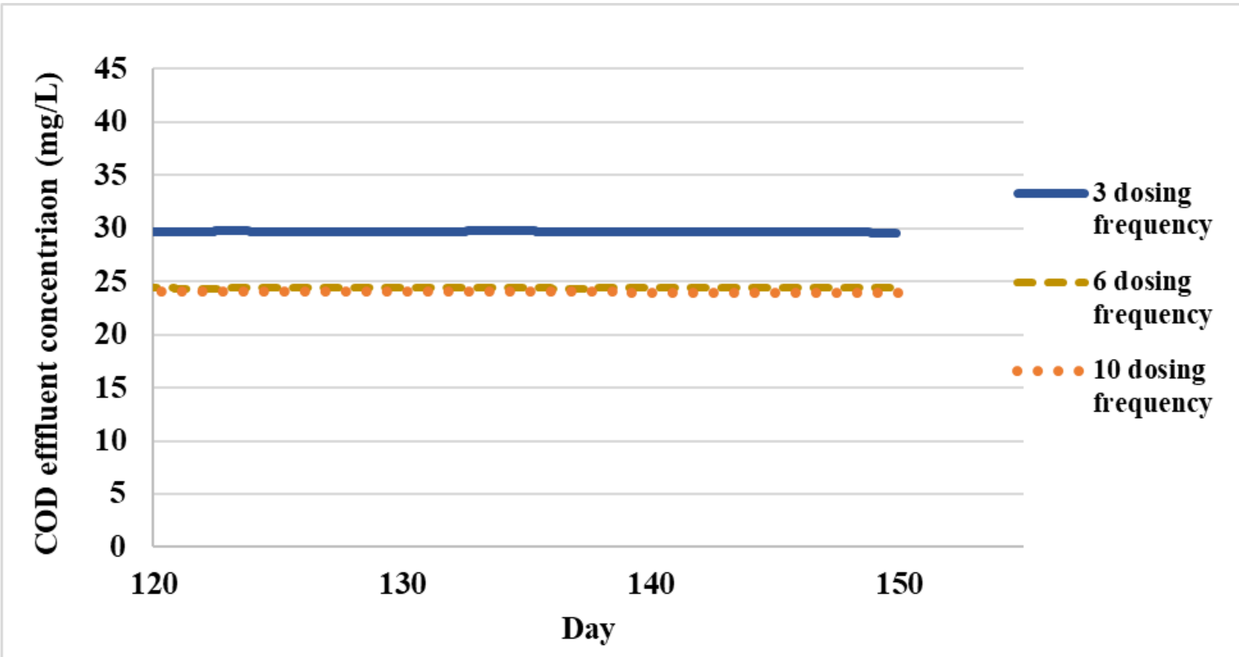


Figure 70. Simulated COD effluent concentrations with multiple strength of loadings

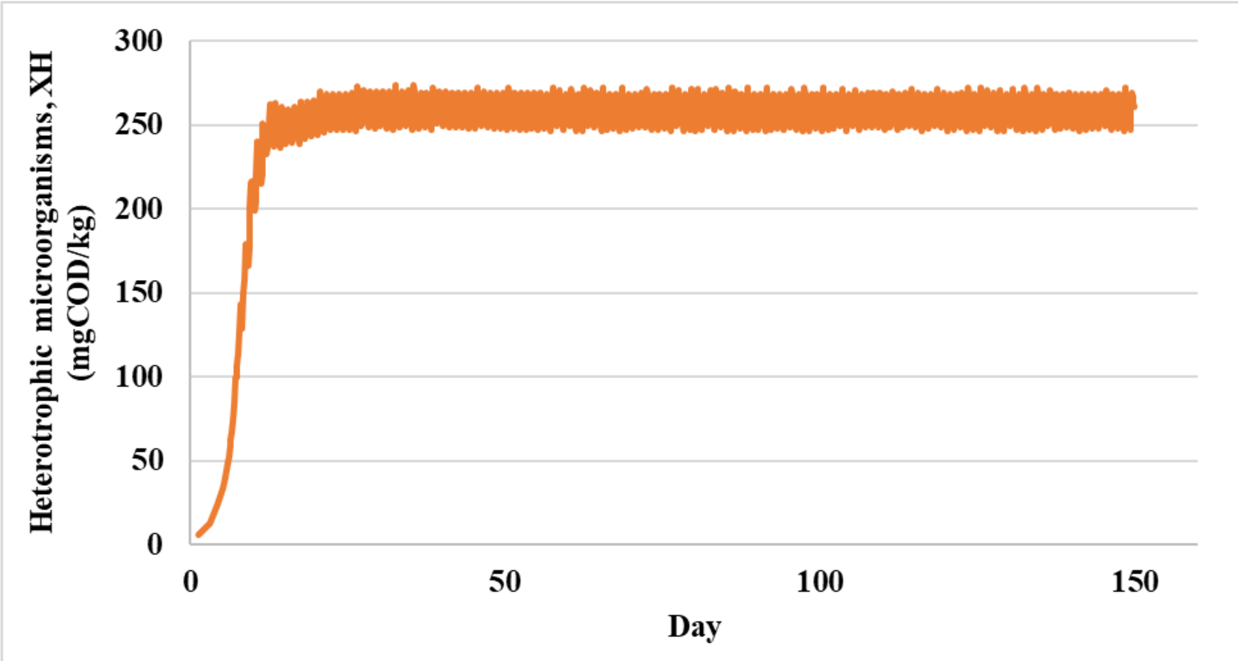


Figure 71. Steady state condition of heterotrophic microorganism (XH) growth

Figure 72 shows the relative differences from the control which is 1 dosing frequency with 619 mg/L of COD. Table 32 shows the relative difference from control for COD simulation (1 dosing frequency with 619 mg/L of COD). Although the strength of hydraulic and organic loadings increased, the effluent COD concentration at 60.96 cm (2 ft) depth of sandy loam did not change. Increasing dosing frequency was able to reduce COD effluent concentration by a maximum of 5.4 mg/L of COD. A possible explanation for a lower COD effluent concentration with higher dosing frequency is an increase in the retention time. High organic loading rate can cause the bioclogging, caused by excessive growth of microorganisms. This is also known as a biofilm. The growth of a biofilm will change of the hydraulic properties but this is not considered in the HYDRUS CW2D model and future study is needed. If the soil surface is clogged by biofilm, soil ponding or surface runoff will occur. Surface runoff can result in soil erosion or contamination of local surface water.

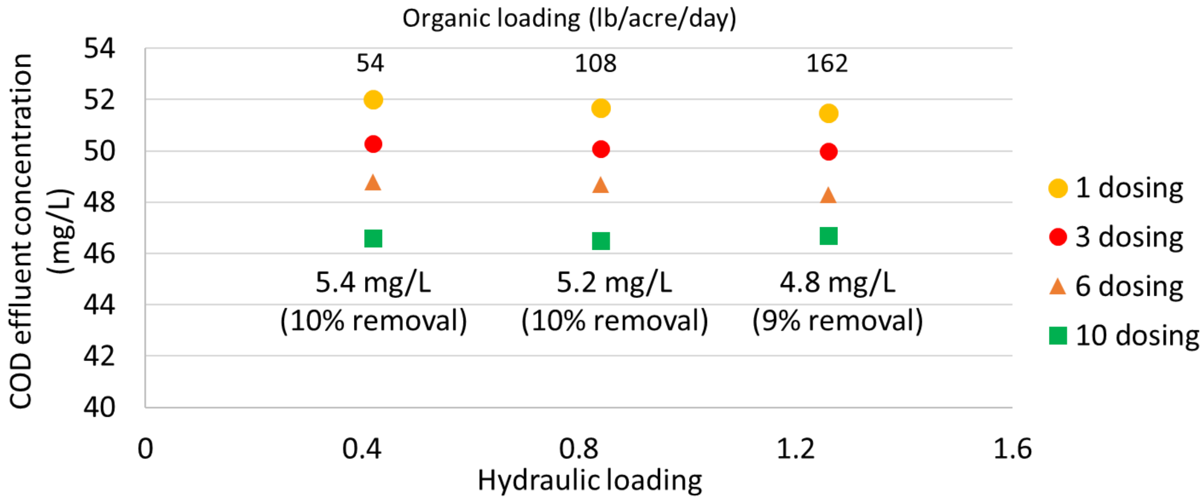


Figure 72. Impact of dosing frequency, and hydraulic and organic loadings on COD effluent concentration on the 150th day at 60.96 cm (2 ft) depth of sandy loam soil

Table 32. Relative difference from control for COD simulation (1 dosing frequency with 619 mg/L of COD)

Hydraulic Loading (in/day)	Relative difference from control (mg/L)			
	1 Dosing	3 Dosing	6 Dosing	10 Dosing
0.42	0	-1.7	-3.2	-5.4
0.84	-0.3	-1.9	-3.3	-5.5
1.26	-0.5	-2	-3.7	-5.3

Figure 73 shows the effect of the dosing frequency and the strength of hydraulic and organic loadings on nitrate effluent concentration. Table 33 shows the relative difference from the control (1 dosing frequency with 102 mg/L of COD).

As doing frequency and hydraulic and organic loadings increases, the relative difference also increases proportionally. The model simulation shows the potential nitrate removal by increasing doing frequency from 36.5 mg/L- NO_3-N to 23.9 mg/L-NO_3-N. When both dosing frequency and hydraulic and organic loading increased, the maximum reduction was from 36.5 mg/L- NO_3-N to 15.7 mg/L-NO_3-N. Higher dosing frequency may increase the moisture content of the soil, leading to a reduction in the oxygen content, providing more optimal conditions for denitrification.

The Michigan Department Environmental Quality accepted a value of 5.6 g-BOD/m^2/day (50 lb-BOD/acre/day) as a monthly average with monitoring groundwater and soil conditions. Figure 74 show that the nitrate removal efficiency for 6 and 10 dosing frequency with 6.0 g-BOD/m^2/day (54 lb-BOD/acre/day) are 33% and 35%, respectively. Thus, 6 or 10 dosing frequency are recommended to achieve at least a 33% of nitrate removal. A center pivot irrigation system is typically used for wastewater application. More frequent application requires more energy, which leads to higher operation costs. Since the removal efficiency for 6 dosing frequency and 10 dosing frequency are not significantly different, 6 dosing frequency is recommended for Michigan.

This model did not consider factors such as precipitation, weather conditions, plant uptake, growth of biofilm, and topography. If considered, the differences may be less or more significant. Consequently, further study is required.

Plant uptake can be significant in removing nutrients. When the crop is actively growing, nutrients in food processing wastewater are used by crops. Corn and alfalfa are grown at the food processing wastewater land application demonstration site. The yields were a combined average of 20 tons/acres and 6.7 tons/acre for corn and alfalfa, respectively, without adding additional commercial fertilizer. The yields were higher than the average corn yield in Michigan. While the crop is growing, increasing dosing frequency for nitrate removal is not needed.

When the temperature decreases, the microorganism's activity also decreases, which slows the denitrification process. During the winter, pretreatment technology and winter cover crop are recommended to minimize nitrate leaching into groundwater.

In Michigan, the permit for wastewater land application systems must be renewed every 5 years. This model can be beneficial to regulator and operator/manager at the food processing wastewater land application facility. Once the model is calibrated and validated with different types of soil, weather condition, and hydraulic and organic loadings, multiple scenarios for different operation approaches can be conducted and provide the recommendations as an index. The index allows to determine the best operation strategies for site and waste- specific condition.

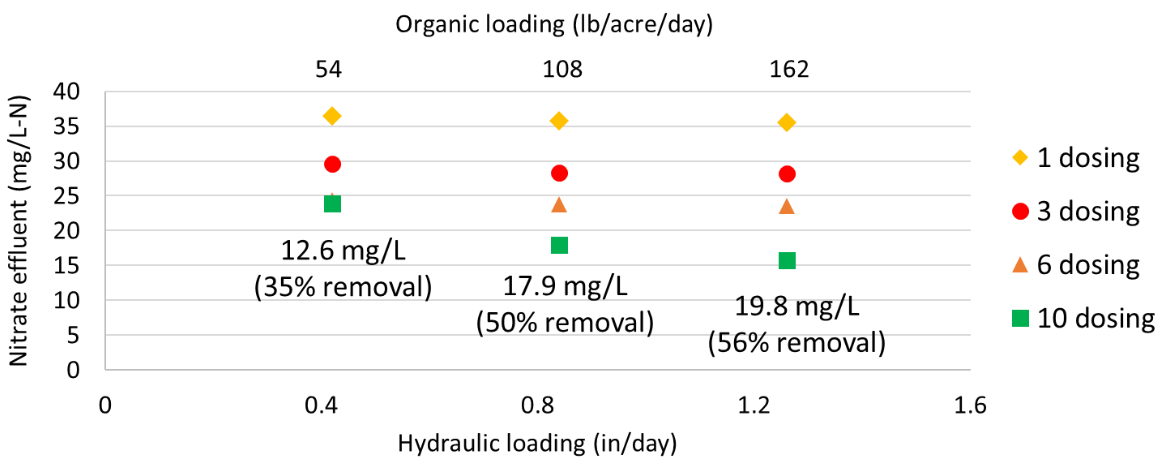


Figure 73. Effect of dosing frequency and the strength of hydraulic and organic loadings on nitrate effluent concentration

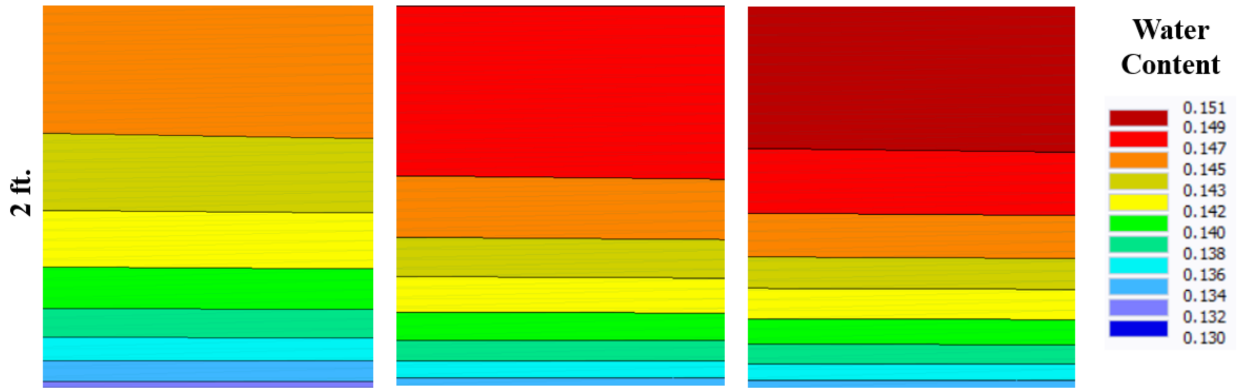


Figure 74. Simulation of the impact of volumetric water content in soil by different dosing frequencies at 150th day; 3 dosing (left), 6 dosing (middle), and 10 dosing (right)

Table 33. Relative difference from control for nitrate simulation (1 dosing frequency with 619 mg/L of COD)

Hydraulic Loading (in/day)	**Relative difference from control (mg/L)**			
	1 Doses	**3 Doses**	**6 Doses**	**10 Doses**
0.42	0	-6.9	-12.1	-12.6
0.84	-0.7	-8.2	-12.7	-18.6
1.26	-1	-8.3	-12.9	-20.8

4.4. Conclusion

In conclusion, the multiple monitoring practices assured the manufacturer and regulatory agency that environmental risk was minimized. Results also indicate that nitrate needs to be further investigated as the complexity of the site and likelihood of a substantial amount in the groundwater is actually entering the site makes it difficult to understand its fate. Regardless, this research illustrated a methodology for determining if the land application of high strength food processing waste is applicable for examining other unique sites and demonstrated benefits. Included are cost savings; the reduced use of freshwater; nitrogen and phosphorus reuse for beneficial crop production; and energy conservation and the resulting GHG reduction associated with the energy saving.

Monitoring strategies including tracking hydraulic and organic loadings, measuring real-time soil condition by soil sensors, and testing groundwater quality showed the viability of using land application to treat food processing wastewater. This data, and in particular the sensor values, also helps make operation decision in regard to the best field to irrigate to at any one time.

Although the benefits and effectiveness, and means to evaluate for site-specific conditions, for the land application of food processing wastewater has been shown to be viable, the design basis is still very empirical. There is not a consensus in the literature on the most strategic operational strategies. Therefore, a finite element modeling approach was explored to determine its potential utility in predicting performance and aiding in design. HYDRUS CW2D model was successfully calibrated and validated using measured volumetric water content from the laboratory experiment previously described. The HYDRUS CW2D simulation shows the potential for enhanced nitrate removal by increasing dosing frequency. This results because increasing dosing frequency can raise the soil's water content in soil resulting in reduced oxygen levels, which can stimulate denitrification. However, this strategy may result in metal mobilization into groundwater. The optimal dosing frequency needs to be studied to mitigate nitrate leaching and metal mobilization into groundwater. Further, causing intermittent high and low soil moisture contents by manipulating dosing frequencies - one of the few practical controls possible in a land application system – may enable both denitrification to occur and prevent metal mobilization.

However, the limitations of HYDRUS CW2D were discussed in Chapter 4.2.3.3, including the failure to represent clogging. Clogging can result from suspended solids and excessive growth of biofilm resulting from the application of high strength food processing wastewater. Once the soil is clogged, standing water and higher moisture contents can result, which has a negative effect on crop growth. To prevent clogging, hydraulic and organic loadings may need to be reduced as well as the collection of runoff and its re-application onto the secondary irrigation land. Currently, HYDRUS CW2D does not consider causes of clogging. However, both particle transport and bacteria growth models are available in the model (Kildsgaard and Engesgaard 2001; Mackie and Bai 1993). This data can be potential used to adjust the soil's pore size (Radcliffe and Simunek 2010). Understanding the thickness, hydraulic conductivity, and development rate of the biofilm under different conditions, is needed to achieve this.

Recommendation for non-optimal areas are discussed below. Novel management approaches may provide optimal conditions for crop growth and maximize soil treatment capacity.

When comparing cover crops, the first step is to identify unique site-specific attributes. Good options for cover crops include, but are not limited to, triticale, annual ryegrass, and cereal rye. Further details are provided on each. Triticale is thought to be the best option for a wastewater land application site. Triticale is a cross between wheat and rye with the quality and yield of wheat and the disease and winter hardiness of rye. Its fibrous root system prevents erosion and builds the soil structure, reducing compaction. The dense stand helps dry out wet spring soil enabling the planting of the primary crop earlier in the season. Triticale can be used as feed for various classes of livestock including lactating dairy cows.

Another option is variable irrigation. Variable rate irrigation uses solenoid valves through the length of the pivot arm to turn off and on predetermined segments as it progresses around the field. These off and on patterns can be matched to the soil's assimilation capacity as determined by localized field conditions such as the soil characteristics, topography, and localized high water conditions. Although this approach may work well in warm seasons, it may be difficult to implement in winter as the valves may freeze and not work properly.

Field and soil modifications can also be considered. If the non-optimal section is caused by the topography of the field, such as a low location, fill can be added. However, this may be very expensive, depending on the extent of the depression. Soil amendments that sorb water and increase porosity are also possible. Such materials include gypsum, lime, and biochar. Gypsum is composed of calcium sulfate dehydrate and is used for improving soil health, specifically saline soils (Grubb et al. 2012). Lime, composed of calcium carbonate, increases the pH of acidic soil, improves soil infiltration rates, decreases soil compaction, increases rooting depth, and adsorbs phosphorus (Berglund 1996). Biochar can optimizes soil conditions to increase crop production by reducing the soil's bulk density and increasing its nutrient absorption, pH, water holding capacity, infiltration rate, and soil microbial activity (Anderson et al. 2011; Graber et al. 2010; Hussain et al. 2016; Liu et al. 2015; Schnell et al. 2012; Ventura et al. 2012). However, this option may also be expensive so the extent of required treatment must be carefully considered.

Chapter 5. Conclusion and recommendations

In this section, results are first summarized. Thereafter, insights and recommendations for further research are provided.

5.1. Summary

The objectives of this study were to demonstrate the effectiveness of domestic wastewater land application systems by examination of the literature, evaluate the effectiveness of food processing wastewater land application, compare the benefits of wastewater land application to conventional wastewater treatment systems, develop a simulation approach for the complex wastewater land application treatment system using HYDRUS CW2D, and analyze multiple scenarios using the calibrated model to correlate operational parameters. A summary of the results follow.

- Land application treatment systems for domestic wastewater are effective and can achieve an average removal efficiency of more than 60% for COD, BOD_5, TP, and ammonia. Denitrification to convert nitrate to nitrogen gas does not occur in domestic wastewater land application treatment systems, even with modified dosing frequencies and high application rates.
- Food processing wastewater land application was found to be effective and efficient at the demonstration site as found by monitoring hydraulic and organic loadings, real-time soil conditions by soil sensor clusters, and groundwater sample analyses. The methodology

developed and used at the demonstration site is applicable to other sites and can aid in making important operational decisions. Monitoring results are summarized below.

- Soil sensor clusters located under the center-pivot irrigation system over a long period of operation proved to be reliable and economical.
- Volumetric water content sensors demonstrated the same trends as oxygen sensors in regard to indicating if soil conditions were aerobic or anaerobic. The water content sensors require less maintenance and are more economical than oxygen sensors.
- Corn and alfalfa are grown at the food processing wastewater land application site. The yields were a combined average of 20 tons/acres and 6.7 tons/acre for corn and alfalfa, respectively, without adding additional commercial fertilizer. These yield were higher than the average corn yield in Michigan.
- Non-optimal areas at the demonstration site were delineated based on poor crop growth and standing surface water. Soil texture and soil compaction were not found to be the primary causes. Instead, low areas in the field combined with localized high water conditions appear to be the problem, although more research is needed.

- Wastewater land application systems, when compared to conventional wastewater treatment systems, provides benefits such as reducing usage of freshwater and energy saving that is required to treat the wastewater, consequently reducing GHG emission.
- HYDRUS CW2D was successfully calibrated and validated using measured volumetric water content data from laboratory experiments.
- Multiple scenarios analysis using the calibrated and validated model was conducted. Model simulation results are summarized below.

- Most of the COD removal in a domestic wastewater land application system occurs within a 15.24 cm (1 ft) depth for a sandy loam soil.
- Increasing the dosing frequency was effective in slightly reducing the COD effluent concentration. A possible explanation is the increase in retention time.
- At a typical influent COD concentration of domestic wastewater, nitrate-nitrogen removal could not be achieved by increasing the dosing frequency. Consequently, the hypothesis that increasing dosing frequency would increase the soil moisture content and/or increasing the amount of carbon that proceeding to a depth where the soil environment was anaerobic was not proven.
- At a high COD, such as in food processing wastewater, nitrate removal increased when both dosing frequency and hydraulic and organic loadings increased. The cause was hypothesized as above.

5.2. Recommendation

The followings are recommendations from this study.

- Design criteria for domestic wastewater land application system provides several options for soil types, including coarse sand, medium sand, sandy loam, and loamy sand. The soil types of the food processing wastewater land application demonstration site vary, and includes loamy sand, sandy loam, and sand, depending on location and depth. Calibrating the model using different soil types is suggested.
- Model parameters such as hydraulic loading, maximum aerobic growth rate of XANs, maximum denitrification rate of XH, and fraction of CI in biomass lysis were calibrated using laboratory experimental data. Calibrating the model using field data is recommended. Tile drainage and soil lysimeters are options to capture wastewater and nutrients transported through the soil column. Volumetric water content soil sensor is also recommended for calibrating water flow.
- Model simulation result shows that more frequent dosing and carbon are needed to promote denitrification for domestic wastewater land treatment systems. These conditions may stimulate the growth of a biofilm that may restrict flow, resulting in the premature life of drain field. The optimal dosing frequency and influent COD concentration to minimize biofilm growth should be studied.
- Metal mobilization is a concern at the demonstration food processing wastewater land application site. Modeling the metal mobilization using HYDRUS PHREEQC is recommended. HYDRUS 1D–PHREEQC can simulate the transport of multiple components and mixed equilibrium/kinetic biogeochemical reactions, including interactions with minerals, cation exchange reaction, and pH dependent cation exchanges (Šimůnek et al., 2013). Previous

studies have used HYDRUS to simulate metal mobilization in soil (Anwar & Thien, 2015; Dao et al., 2014; Nakamura et al., 2004; Wang et al., 2016). The HYDRUS PHREEQC modeling approach may help to understand the metal mobilization under different site and waste-specific condition.

- Modeling results for food processing wastewater land application show that increasing dosing frequency can stimulate the denitrification process. Higher dosing frequencies may increase the moisture content of the soil, leading to a reduction in the oxygen content, providing conditions for denitrification. However, reduced oxygen conditions in the soil may result in metal mobilization. The optimal dosing frequency needs to be studied to simultaneously mitigate nitrate leaching and metal mobilization into groundwater.
- In addition to domestic wastewater and food processing wastewater, research on other wastes that are commonly land applied such as winery and milking facility wastewater are suggested including monitoring, benefits, and modeling.
- Impacts of HYDRUS CW2D limitations should be determined. These limitations include the lack of consideration of heterogeneous mixture of soil, precipitation, evapotranspiration, topography, soil pH, plant uptake, macropores, and clogging by suspended particle and excessive growth of biofilm.
- Poplar tree growth in a land treatment system absorbs wastewater and nutrients. These trees have been used for phytoremediation for pollutants such as nitrate, atrazine, metals, organics, chlorinated solvent, and benzene (USEPA 2000). Root system of poplar tree can reach depth of 4.6 m (15 ft.). Aryal and Renhold (2015) have found that a poplar tree is effective in reducing the leaching of iron and manganese and can withstand the continuous saturation of soils condition, while maintaining high evapotranspiration rates (Aryal and Reinhold 2015). More

pilot studies are suggested on the effectiveness, efficiency, and benefits of employing poplar trees in a wastewater land application system are recommended.

Onsite wastewater treatment systems can be a source of pharmaceutical and personal care products (PPCP) that enters into groundwater. Sulfamethoxazole, carbamazepine, and nicotine were detected underlying septic field (Godfrey et al. 2007). Wastewater in Cape Cod (Barnstable County, Massachusetts) is mainly treated by onsite wastewater technologies and disposed through septic fields. Tetrachloroethylene (analgesic), acetaminophen (antibiotic), sulfamethoxazole, caffeine, carbamazepine, dehydronifedipine, diphenhydramine, and p-Xanthine were found in monitoring wells (Zimmerman 2005). Table 43 shows PPCPs found in before and after treatment using drain fields. More research on PPCP transport in soils is needed. Studies on treatment technologies to capture PPCP are also recommended. Previous studies reported a potential use of biochar to treat PPCPs. Yao et al. (2012) study reported that 2-14% of sulfamethoxazole was transported through biochar amended soil, whereas 60% of sulfamethoxazole leached through un amended soil (Yao et al. 2012). Chen et al. (2017) shows cabrbamazepine was effetely removed by biochar (Chen et al. 2017). Studies on the type of biochar and optimal mixture ratios are needed. A column study is recommended.

Table 34. Literature review on PPCP found in a domestic wastewater land application system

Reference	**PPCP**	**Before drain field (μg/L)**	**After drain field (μg/L)**
Swarz et al. (2006)	Caffeine	17 – 23	< 1.7
	Paraxanthine	55 – 65	< 1.7
Conn and Siegrist (2009)	EDTA (ethylenediaminetetraacetic acid)		2.4
	NP1EC (4-nonylphenolmonoethoxycarboxylate)		7.2
	NP (4-nonylphenol)		4.1
	Sulfamethoxazole		0.51
Matamoros et al. (2009)	Salicylic acid	16.4	0.66
	Ibuprofen	1.95	0.02
	OH-ibuprofen	3.45	0.28
	CA-ibuprofen	2.45	0.04
	Carbamazepine	4.5	
	Naproxen	0.09	
	Diclofenac	0.5	
	Ketoprofen	1.79	
	Caffeine	31.9	0.16
	Methyl-dihydrojamonate	8	0.04
	Hydrocinnamic acid	21.1	0.02
	Oxybenzone	3.35	
	Furosemide	4.65	

APPENDICES

APPENDIX A: HYDRUS CW2D parameters

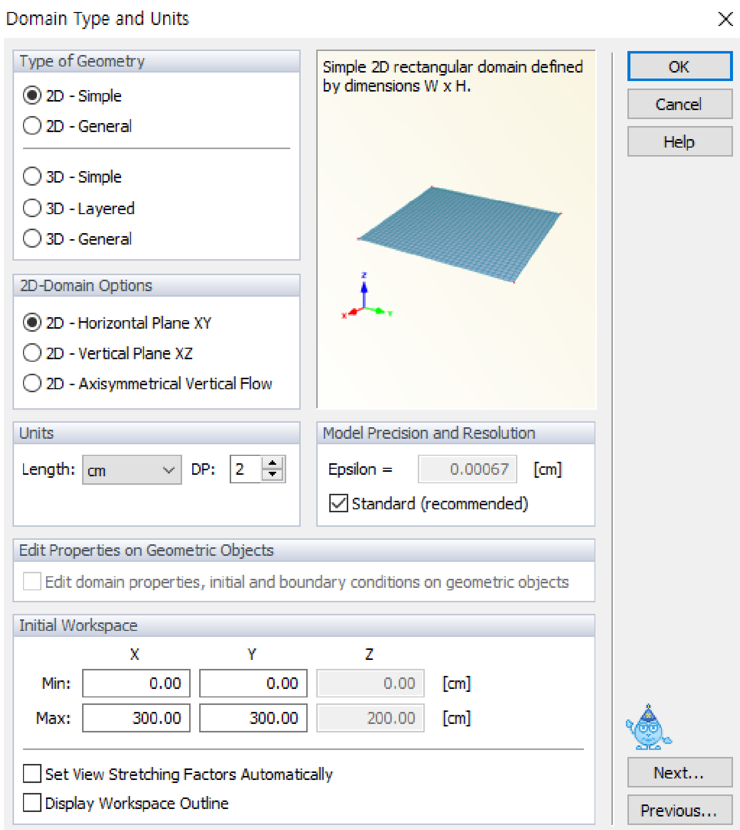


Figure 75. Domain type and units

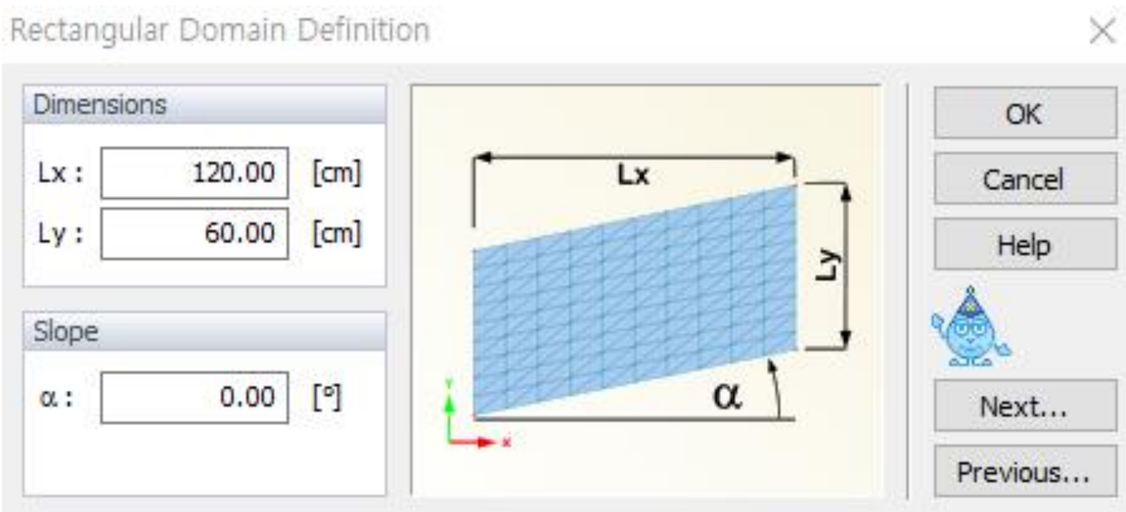


Figure 76. Rectangular domain definition

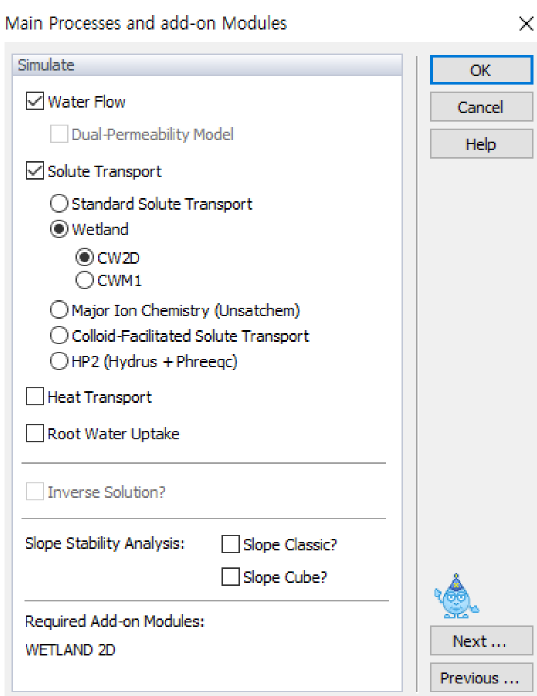


Figure 77. Main processes and add-on modules

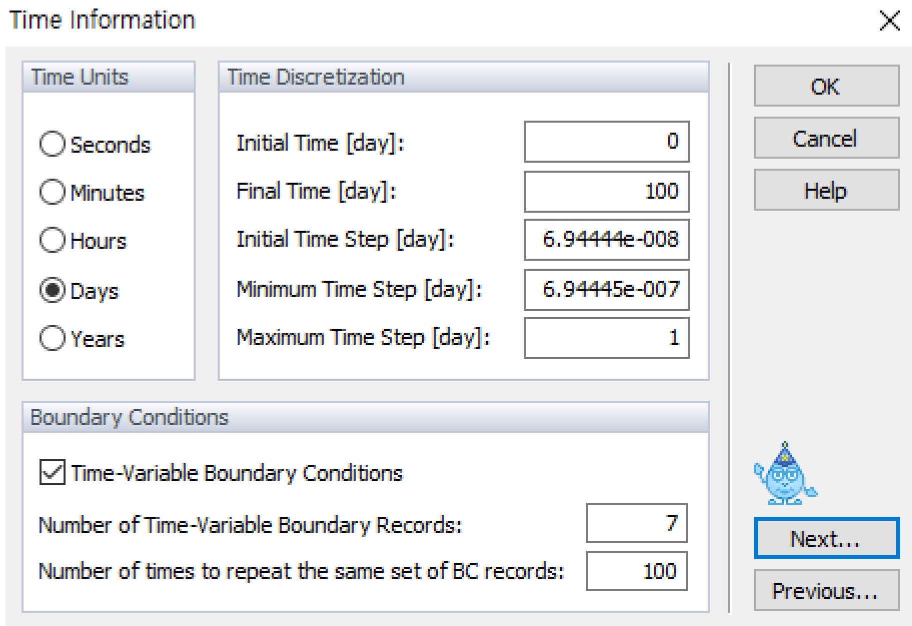


Figure 78. Time information

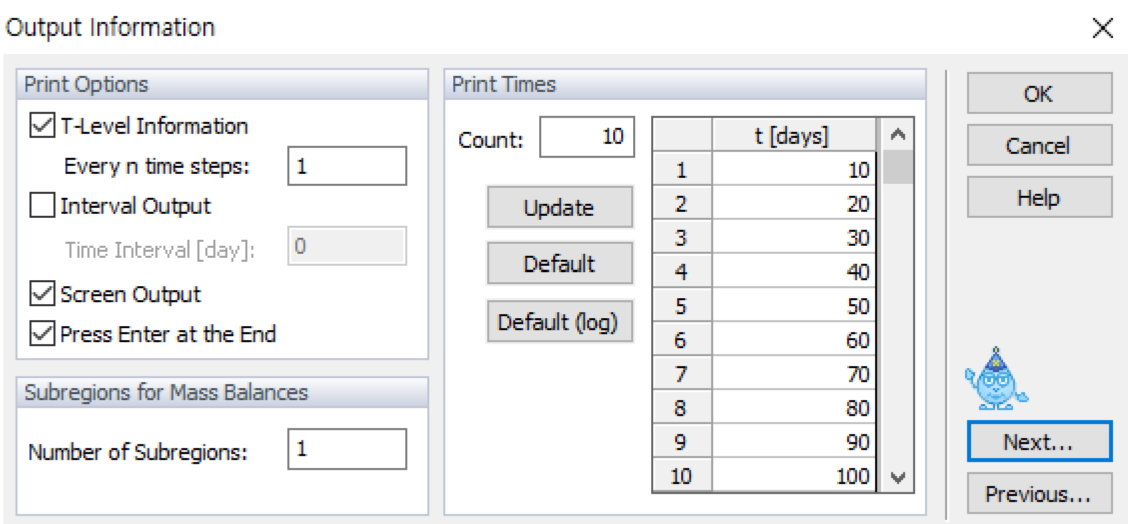


Figure 79. Output information

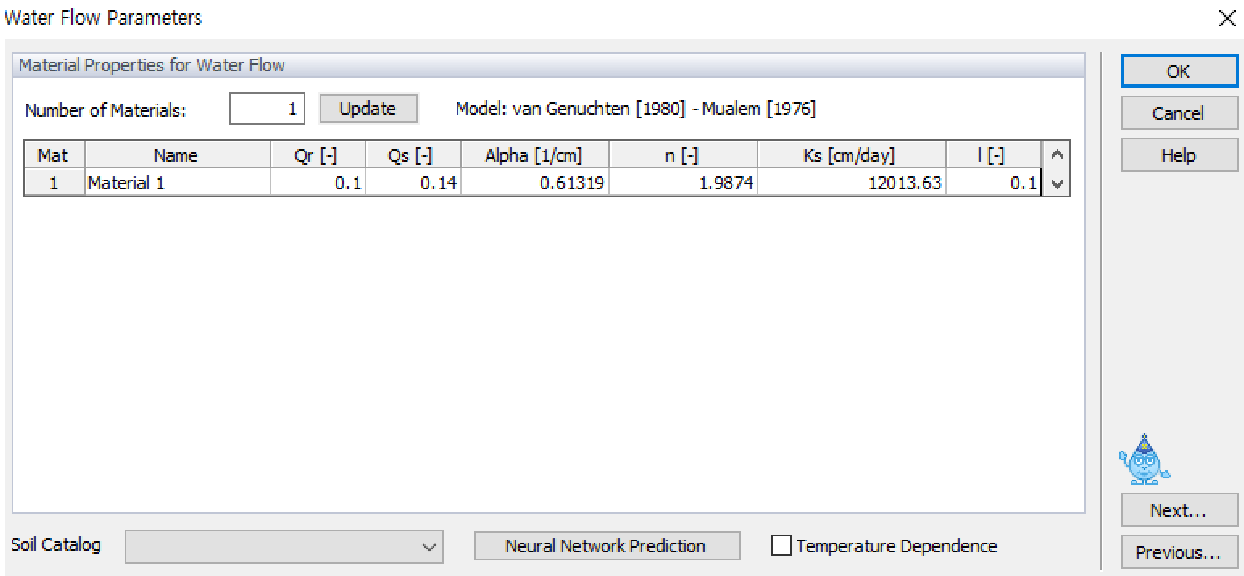


Figure 80. Water flow parameters

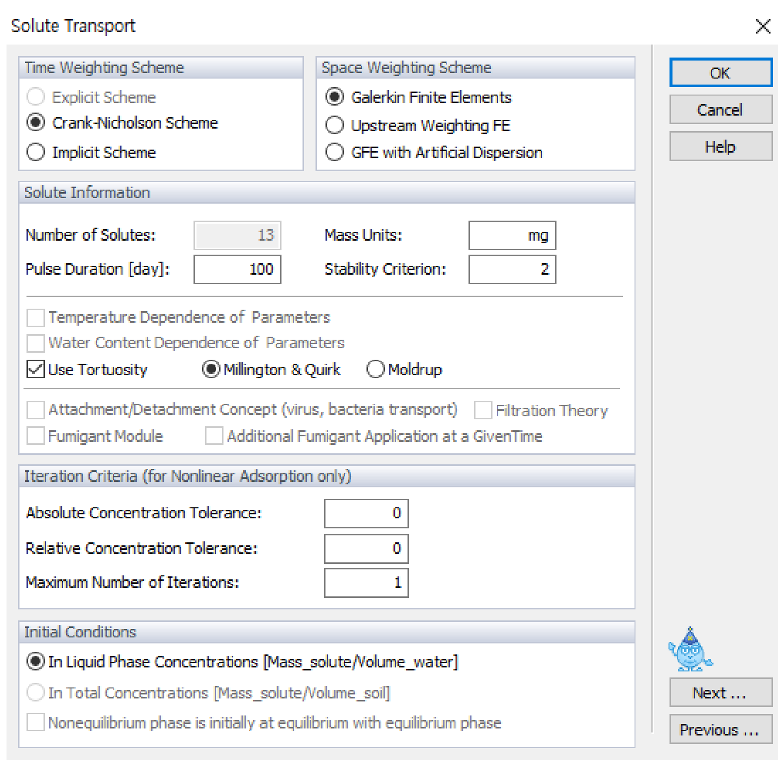


Figure 81. Solute transport

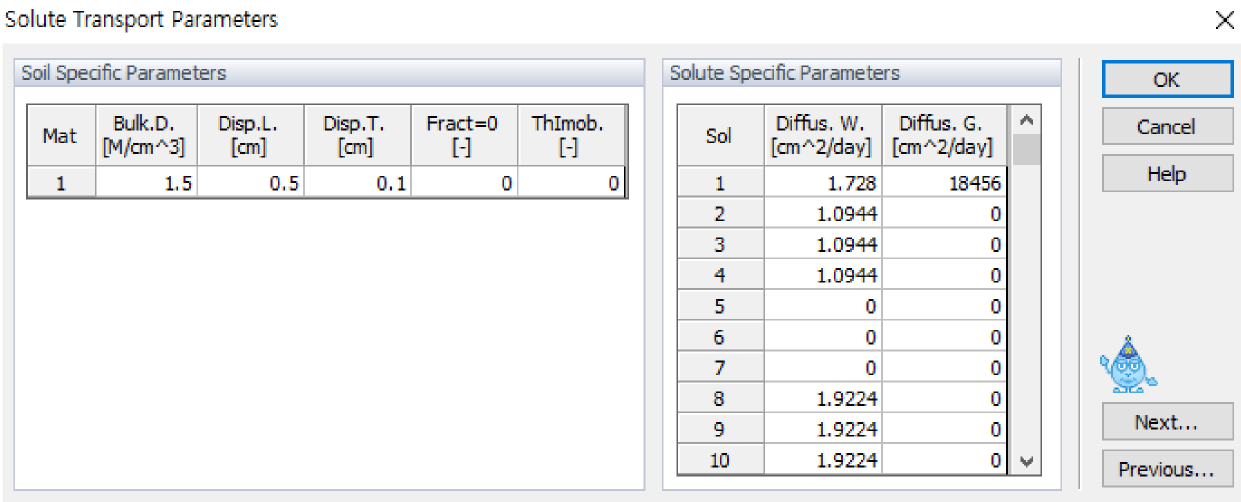


Figure 82. Solute transport parameters

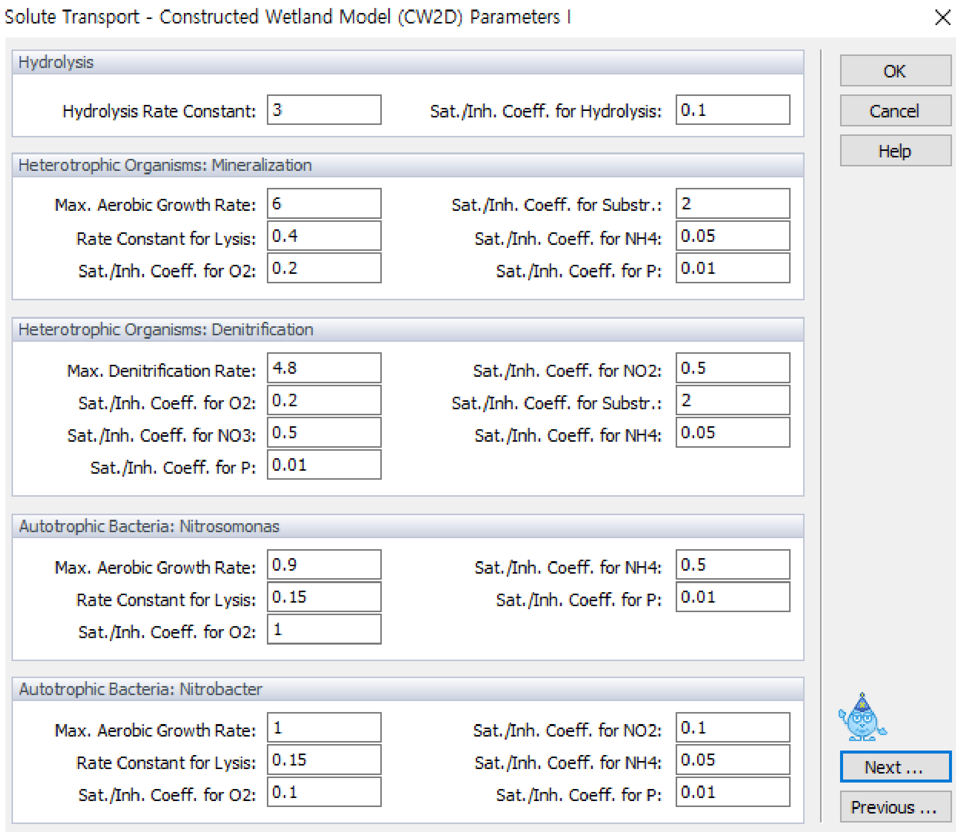


Figure 83. Solute transport – constructed wetland model parameter I (Default)

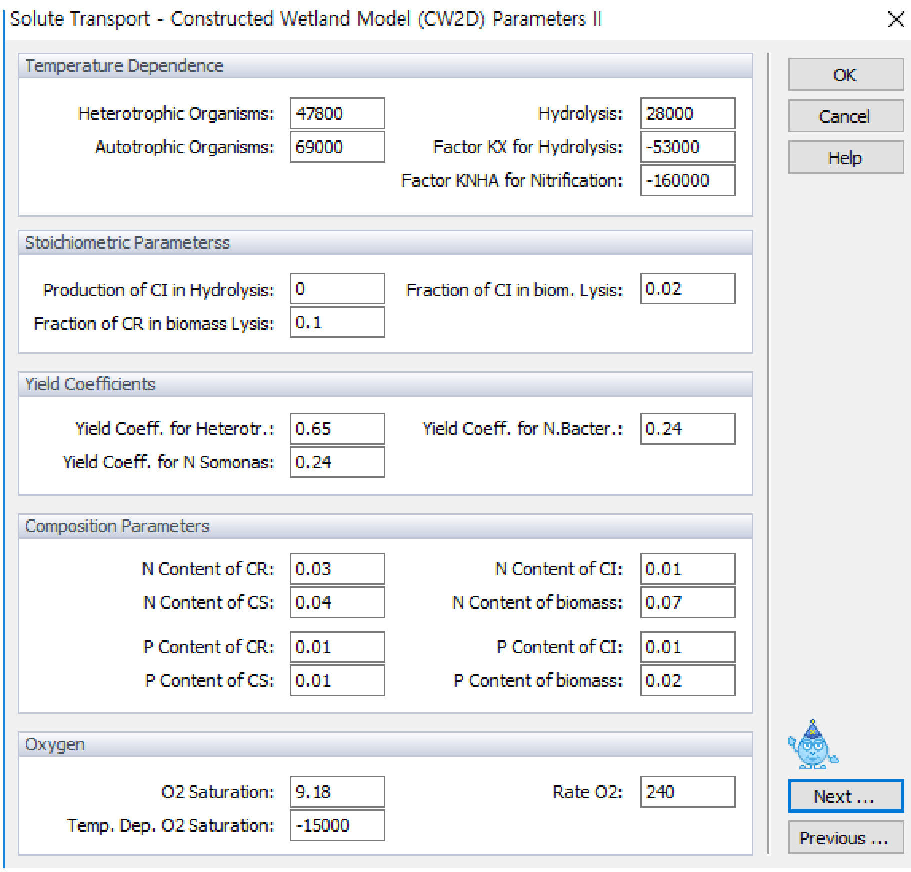


Figure 84. Solute transport - constructed wetland model (CW2D) parameters II (Default)

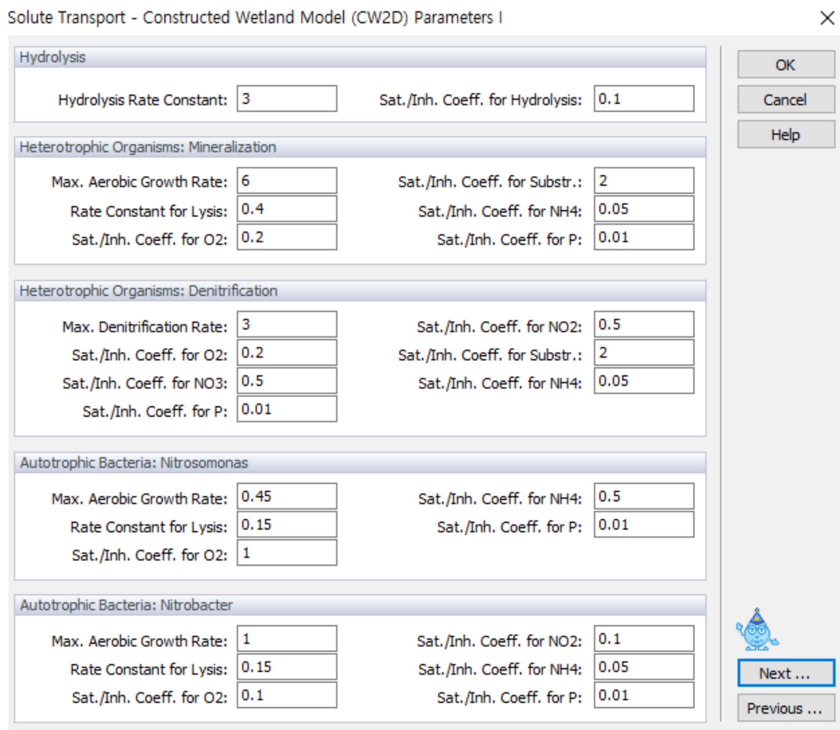


Figure 85. Solute transport – constructed wetland model parameter I (Adjusted)

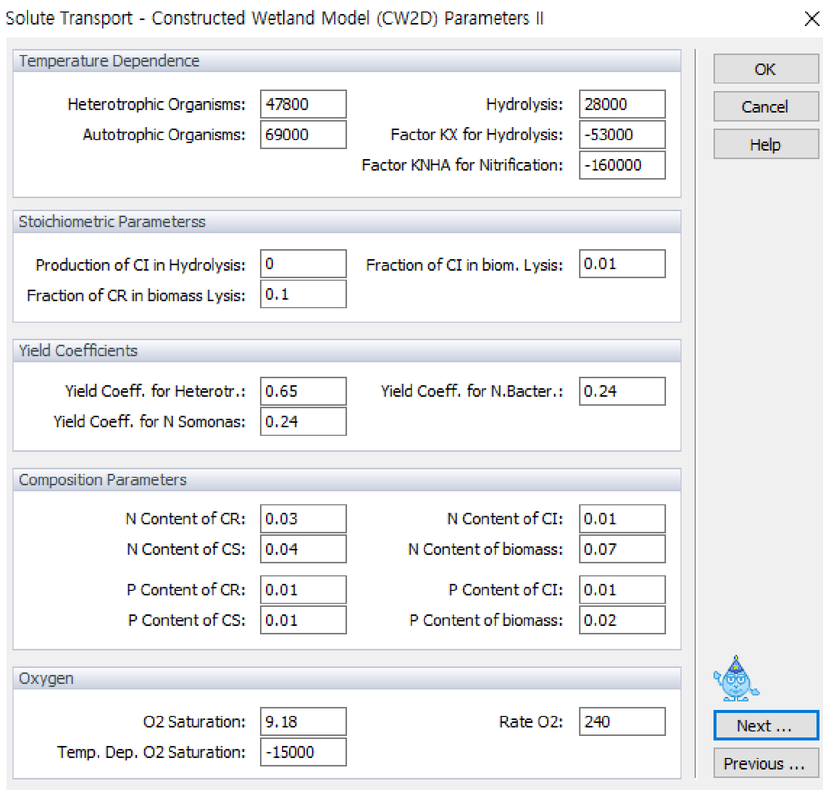


Figure 86. Solute transport – constructed wetland model parameter II (Adjusted)

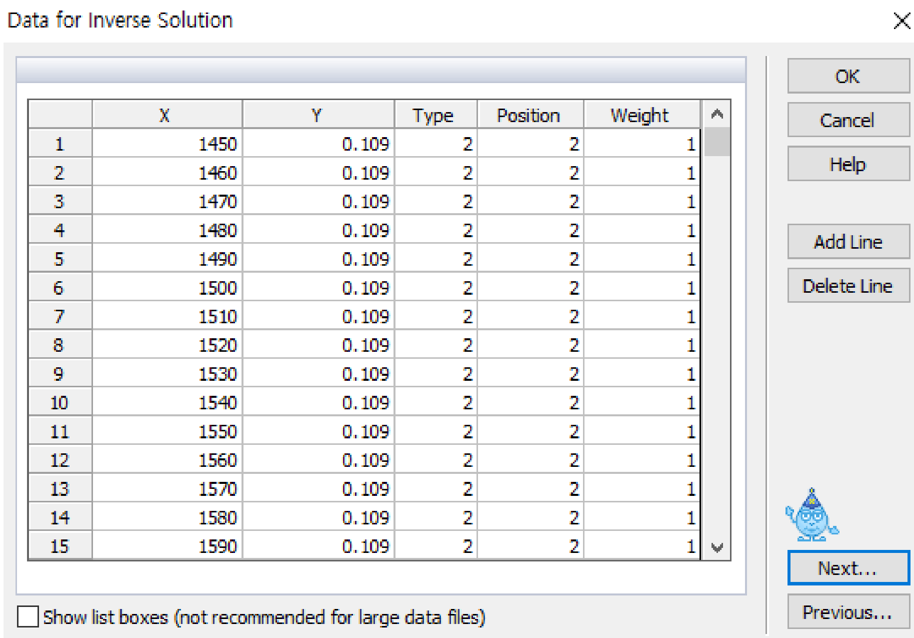


Figure 87. Data for inverse solution

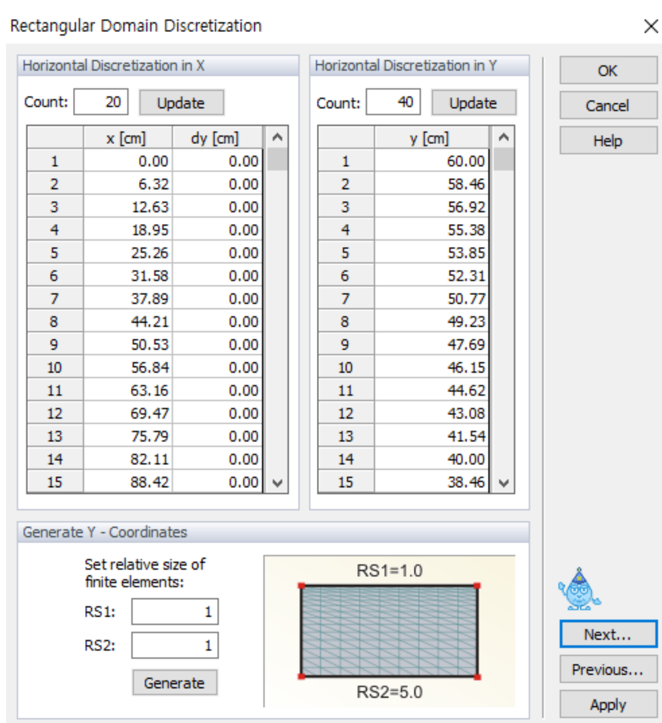


Figure 88. Rectangular domain discretization

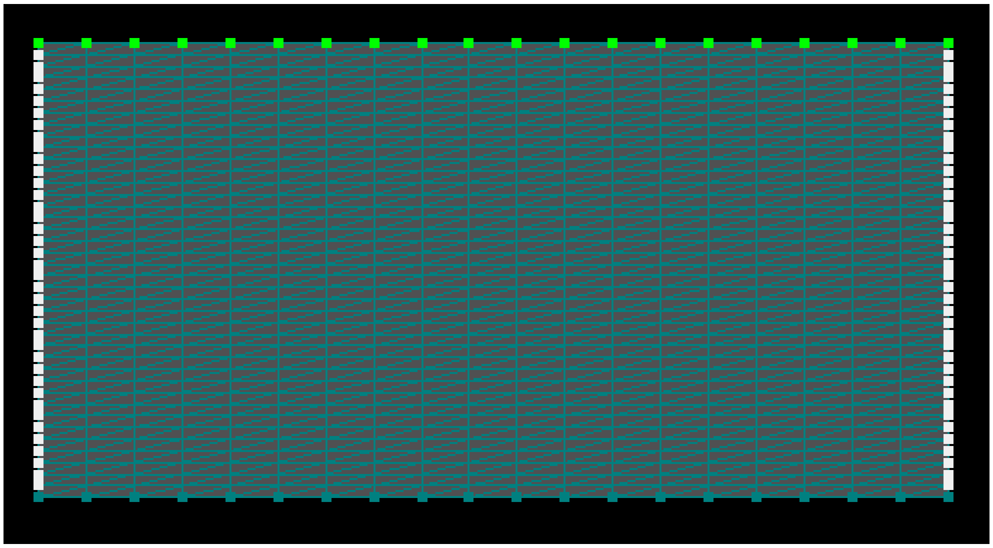


Figure 89. Water boundary condition

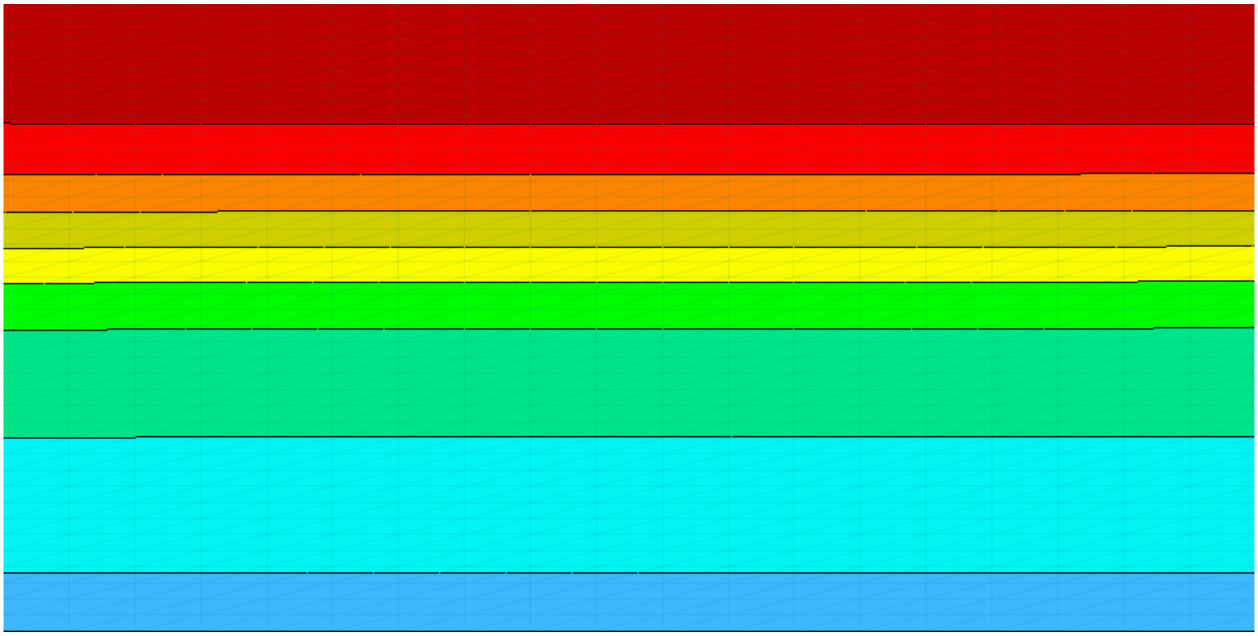


Figure 90. Graphic output from HYDRUS CW2D

APPENDIX B: R – Code (Goodness of fit)

The following R code addresses opening file and calculate modeling efficiency, index of agreement, and root mean squared error.

```
##R code for goodness of fit - water flow

#Setting the working directory
setwd("C:/Users/Dong/Desktop/validation_R/Raw data/Solute calibration data")

#Reading files for calibration
obs <- read.csv("Calibration_obs.csv", header = TRUE)
sim <- read.csv("Calibration_sim.csv", header = TRUE)

#Reading files for validation
obs <- read.csv("Validation_obs.csv", header = TRUE)
sim <- read.csv("Validation_sim.csv", header = TRUE)

#Setting each symbols
t_obs <- obs[,1]
y_obs <- obs[,2]

t_sim <- sim[,1]
y_sim <- sim[,2]


#Step function to organize y for x time scale
```

```r
ef_sim <- stepfun(x = t_sim, y = c(y_sim[1], y_sim), f = 0.5)

# Combine all datas
data <- cbind(t = t_obs, obs = y_obs, sim = ef_sim(t_obs))
head(data)

#Model efficiency
E <- 1 - sum((obs - sim)^2)/sum((obs - mean(obs))^2)

#Index of agreement
IA <- 1 - sum((obs - sim)^2)/sum(((abs(sim - mean(obs))) + abs(obs - mean(obs)))^2)

# Root mean squared error (RMSE)
RMSE <- sqrt(sum((obs - sim)^2)/length(time))
```

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REFERENCES

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