Coupled process modeling for energy generation from municipal solid waste degradation: Laboratory-scale and field-scale simulations

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ABSTRACT

Modern regulated landfills are designed to protect the environment by containing and isolating municipal solid waste (MSW) from the environment. Instead of treating MSW as a hazard to be contained, next generation landfills, here termed Sustainable Energy Reactor Facilities (SERFs) are envisioned to be designed and operated with a main focus on energy generation (through anaerobic biodegradation processes) and sustainability. Towards this vision, a coupled hydraulic-biochemical-mechanical model that is based on the HBM model, is implemented in large laboratory-scale and field-scale studies to simulate the degradation process of MSW. The model captures the consumption of biodegradable organic fraction in the waste by microorganisms, eventually leading to biogas generation and changing solid waste and leachate characteristics. The model was first tested against 0.04 m³ laboratory experiments to assess its ability to predict the observed behavior and derive values for the various model parameters. Subsequently, the model is implemented on a 37m³ field lysimeter at Deer Track Park Landfill. The presented data indicates that the model has the capacity to be implemented in field scale and generate geospatially variable estimates of methane (i.e., energy) yield.

Keywords: municipal solid waste, biodegradation, modeling, coupled processes, sustainable energy

1 INTRODUCTION

1.1 The unsustainability of modern landfill regulations

The design philosophy of modern landfills in the United States (US) and worldwide, focuses on containment of municipal solid waste (MSW) by isolating the waste mass from the environment with base containment and capping systems. The subtitle D regulations of the Resource Conservation and Recovery Act in the US for MSW landfills were a major step forward for environmental protection and addressed the immediate major concern to protect the groundwater and surface water from uncontrolled waste disposal that was extensive at that time. Based on these regulations, sanitary landfills have been operated as “dry-tombs”, which means that efforts are made to minimize water percolation in the waste mass. However, these landfill practices, which are still used to manage 53% of the generated MSW in the US to date (USEPA, 2020), are inherently unsustainable. Although without increased moisture content, waste degradation rate is dramatically lowered and the risk of a leakage and groundwater contamination is reduced, the inhibition of waste degradation rate means that the waste remains a hazard to the environment for practically indefinitely, or at least until moisture in the waste mass increases. Still, the waste mass is isolated by engineered systems that have a finite design life. The durability of composite containment systems has high variability (Rowe, 2020), and as we head towards the fourth decade of operation of these containment systems, there are signs that “indefinite” containment system longevity is not likely or viable (Khanbilvardi et al., 1995; Bonaparte et al., 2002; Moo-Young et al., 2004; Rowe, 2005; Laner et al., 2012; Rowe, 2020). Concurrently, due to the construction and operation costs of modern landfills, the sizes of landfills have increased significantly, and thus, the long-term risks of various types of failures and leakages are increasing.

In addition to long-term integrity concerns, under sub-optimal degradation conditions (due to low moisture content), the rate of biogas generation remains low, and not suitable for energy generation. With the exception of regions with high precipitation where water percolation in the waste mass is sufficient and gas generation is enhanced, energy generation from biogas
collection is not common. Presently in the US, landfill gas (LFG) is mostly flared, vented, or leaked to the atmosphere, contributing to greenhouse gas (GHG) emissions. LFG consists primarily of methane (~50%) and carbon dioxide (~50%), both GHGs. As a result, MSW landfills are the third-largest source of anthropogenic methane emissions in the US, releasing an estimated 111 million metric tons of carbon dioxide ($\text{CO}_2$) equivalent (MMTCO$_2e$) to the atmosphere in 2018 alone (USEPA, 2020). Currently, out of 1,754 landfills that receive the majority of MSW produced in the US, only 558 landfills collect methane that is generated under sub-optimal conditions; the majority of landfills (67%) do not. Out of the total volume of methane generated in the US (~17,500 million m$^3$), less than one-third (~5,000 million m$^3$) is collected for energy annually. In summary, current practices result in lost opportunities for energy generation, while burdening the environment with GHGs. Progress has been made with bioreactor landfills, where moisture is introduced and leachate is recirculated in the waste mass to enhance waste degradation and stabilization (Reinhart and Townsend, 1998; Pohland, 1975; Sharma and Reddy, 2004). However, bioreactor operations have not grown in engineering practice in recent years due to a number of reasons including regulatory restrictions and the fact that many operations and technologies in bioreactor landfills mimic those of Subtitle D landfills that are not suited for maximizing energy recovery. Increased gas emissions, increased odors, physical instability of waste mass due to increased moisture and density, clogging of leachate collection and recirculation systems, surface seeps, and underground waste smoldering are among concerns affecting their operation.

### 1.2 Shifting towards energy recovery and sustainability

New approaches that manage MSW as a resource that is well-understood and can be effectively harvested are needed. These approaches deviate from current regulatory frameworks that treat MSW as a hazard to be contained and require only a basic understanding of the waste material. It is now well understood that MSW has a significant energy potential (Staley and Barlaz, 2009; Krause, 2016). Under appropriate biochemical conditions and moisture, anaerobic processes can become truly efficient, generating significant amount of methane, which represents a sustainable source for power and heat generation (Barlaz et al., 2010; Benson et al., 2007; Pohland and Harper, 1986). Instead of operating landfills as a repository for biogenic carbon that is slowly biodegraded to methane and remains largely untapped as a nation-wide energy source, we propose that next generation waste management facilities are designed primarily as energy generation facilities that we refer to as Sustainable Energy Reactor Facilities (SERFS). SERFS represent next-generation landfill facilities and are designed and operated with two explicit objectives: (a) energy generation, and (b) sustainable environmental stewardship. An explicit focus of energy generation from solid waste at a large-scale is the new element in the operation of SERFs, in contrast to anaerobic digesters, for example, that are too expensive to build and operate at scales equivalent to landfills. SERFs take advantage of the fact that the majority of MSW landfilled is still organic waste (such as food, paper, wood, yard trimmings) that are also not easily (or at all) recycled and are biodegradable under anaerobic conditions. Such approaches will be more sustainable than conventional landfilling, but also more sustainable than composting, which still has high operation costs, strict waste input requirements, low treatment capacity, and no energy output.

To achieve this transformation of landfills from containment facilities to SERFs, a philosophical change from modern MSW management practices, which remain largely empirical, is needed. Synthesizing available data, Fei et al. (2016) demonstrated that the overall degradation rate of MSW reduces significantly as the amount of MSW that undergoes degradation increases. This reduction in degradation efficiency with size of waste mass processed is not unavoidable but reflects the inability of current practices to monitor and control full-scale operations for energy optimization. For example, in current operations, rarely are spatially distributed field measurements conducted to allow for an understanding of the state of degradation at different locations of the landfill, while the ability to control leachate recirculation is limited. New sensing techniques and technological advances however can make this possible.

To achieve this shift towards efficient energy generation, two main requirements need to be satisfied:

(a) The development of a holistic model that couples the interdependent processes of MSW undergoing degradation across scales and can yield realistic predictions of energy generation (i.e., methane yield); and

(b) An ability to monitor and control the energy generation process based on field monitoring and operations that are paired with the holistic coupled model.

Research in our group as well as others, has been ongoing for the last decade towards these requirements, including fundamental understanding of the processes involved (Fei and Zekkos, 2018; Olivier and Gourc, 2007, Bareither et al., 2012), automation of monitoring, (Bateman et al., 2016, Champagne et al., 2020) and coupled process modeling. In this paper, we focus on coupled process modeling that:

a) captures the interdependencies of the biochemical, physical, hydraulic and mechanical characteristics of MSW during degradation;
b) can be calibrated against recent large-scale well-instrumented laboratory experimental data;

c) predicts macroscopic, observed behaviors; and

d) can be updatable using pilot-scale/field-scale measurements.

2 A COUPLED MODEL FOR DEGRADATION PROCESSES

During biodegradation, the process of anaerobic conversion of organic waste in primarily methane and carbon dioxide, affects the solids, liquids and gaseous phases of waste with time, due to interdependent biochemical-mechanical-hydraulic processes (Barlaz et al., 1998, McDougall 2007, Olivier and Gourc 2007). The process begins with breakdown of the complex organic molecules into simpler and soluble compounds, leading to mass loss and void creation in the waste matrix (Olivier and Gourc 2007). Consequently, physical properties, such as porosity and density, change, affecting the mechanical properties of MSW. The soluble compounds are then converted to biogas.

In recent years, several models have been developed that capture aspects of the inter-dependencies among degradation processes and the evolving physical, biochemical and mechanical characteristics of the waste (McDougall, 2007; Gawande et al., 2010; White and Beaven, 2013; Hubert et al., 2016; Hu et al., 2020; Kumar and Reddy, 2021; Li et al., 2021; Lu et al., 2021). Such models strengthen our ability to assess the state of waste in the field and optimize energy generation. However, a common challenge in all modeling efforts is the lack of large-size laboratory and field experiments that collect the comprehensive data that are needed to validate these models. In addition, for energy generation purposes, the ability of the models to be implemented in engineering practice is critical. This requires an understanding of the influence of the various model parameters on the observed results, as well as an ability to select reliable input model parameters based on waste characterization and time-dependent field monitoring data.

The model implemented in this study is built on the hydro-bio-mechanical (HBM) model originally proposed by McDougall (2007) with two main amendments. First, the original HBM formulation is modified to consider the presence of hemicellulose in the organic matter of MSW, in addition to cellulose considered in the original formulation. This is important as the modeled stoichiometry has implications on energy output. Second, the load induced mechanical (stress-strain) behavior of the original HBM model is modified to consider the complex nature of MSW. By leveraging a comprehensive set of large-size experimental data, the updated HBM model parameters are calibrated providing the opportunity to estimate reasonable ranges of values for the various model parameters, and, when appropriate, relate model parameters to waste composition and field operations.

2.1 Formulation and governing equations

Due to the spatial heterogeneity of MSW, changes in stress conditions, and variations in flows of liquids and gases within a waste body, a model that can consider spatial variability is key for energy extraction. For example, leachate recirculation in landfills is not uniform and preferential flow paths have been shown to affect accessibility to liquids, which is a critical component of energy generation (Huber et al., 2004; Rosqvist et al., 2005; Woodman et al., 2015; Zhang and Yuan, 2019). The developed coupled hydraulic-biochemical-mechanical model is extended to two dimensions (space and time) and implemented first to model the variation (in flow, vertical stress) within each laboratory specimen, and second to model larger-scale studies in the field to better understand the degradation process and guide decisions in optimizing it.

The coupled modeling framework is shown in Figure 1. It solves simultaneously a hydraulic model for unsaturated flow, a biodegradation model for the

![Fig. 1. Modeling framework for degradation of MSW in landfills as incorporated in this study.](image-url)
anaerobic degradation process and a mechanical model to account for the stress-strain response of MSW. The hydraulic model uses Richard’s equation (governed by Darcy’s law) for unsaturated porous media flow where the water retention behavior is governed by the Van Genuchten (1980) model. The biodegradation model is a two-phase anaerobic process incorporating hydrolysis of organic fraction, organic acid formation, and soluble substrate depletion. The mechanical model consists of load-induced compression behavior, biodegradation-induced settlements and time-dependent residual creep. The mathematical modeling framework is incorporated in the finite element-based multi-physics software COMSOL (version 5.4).

2.2 Component modules

Hydraulic module

The hydraulic model is an unsaturated flow model based on the Richard’s equation in which the main system variables are hydraulic pressure head and moisture content. The amount of moisture retained is related to the (negative) porewater pressure or suction through the van Genuchten’s expression. Moisture flow is assumed to comply with Darcy’s law. In unsaturated zones, the coefficient of permeability or hydraulic conductivity \(k(\theta)\) is dependent on the amount of moisture.

Biodegradation module

The biodegradation module of the model describes a two-stage anaerobic digester that accounts for hydrolysis of degradable matter in relation to moisture content, product inhibition, substrate digestibility and microbial controls. Degradable matter of MSW consists of cellulose and hemicellulose which account for 91% of the methane potential for typical MSW (Barlaz et al., 1989; Eleazer, 1997) and is commonly 63–78% cellulose (Angelidaki and Ahring, 2000) with hemicellulose content as high as 20% (Barlaz, 1997).

Volatile fatty acids (VFA) and methanogenic biomass (MB) concentrations are the main field variables. Solid degradable fraction (SDF) depletion is calculated for each time step and is controlled by the VFA and MB concentrations and moisture content. The combined growth and decay of VFA and MB in the biodegradation model are described in Equations 1a and 1b:

\[
D_c \frac{\partial^2 c}{\partial x^2} + D_t \frac{\partial^2 c}{\partial z^2} + \frac{q_x}{\theta} \frac{\partial c}{\partial x} - \frac{q_z}{\theta} \frac{\partial c}{\partial z} + [r_g - \eta_h] = \frac{\partial c}{\partial t} \tag{1a}
\]

\[
D_m \frac{\partial^2 m}{\partial x^2} + D_t \frac{\partial^2 m}{\partial z^2} + \frac{q_x}{\theta} \frac{\partial m}{\partial x} - \frac{q_z}{\theta} \frac{\partial m}{\partial z} + [r_g - \eta_k] = \frac{\partial m}{\partial t} \tag{1b}
\]

where \(D_c\) [\(\text{m}^2\cdot\text{s}^{-1}\)] is the VFA diffusion coefficient, \(q_x\) and \(q_z\) are the advective fluxes and \(D_m\) is the MB diffusion coefficient. The kinetics of the stoichiometry are controlled by growth and decay parameters for each of the biodegradation variables, \(c\) [\(\text{g.m}^{-3}\)] is the VFA concentration in aqueous medium; \(m\) [\(\text{g.m}^{-3}\)] is the methanogen biomass (MB) in aqueous medium. In general, reaction rate equations are expressed in \(\text{g.m}^{-3}\cdot\text{aqueous.day}^{-1}\). The two simultaneous ordinary differential equations are solved iteratively by updating system parameters until a consistent solution is obtained which agrees well with the experimental data.

The SDF is depleted in each timestep using Equation 2:

\[
S_{t+\Delta t} = S_t - \frac{A\times162 + (1-A)\times132}{60} \rho_{H,O} \Delta t \tag{2}
\]

where \(A\) is the fraction of cellulose, \(S_t\) is the solid degradable fraction remaining in timestep \(t\) and \(t+\Delta t\) is the next time step. In this study, biodegradation of both cellulose and hemicellulose is considered in the ratio of 80:20 (\(A=0.8\)). The expression in Equation 2 is derived from the overall stoichiometry, which indicates 60 g of acetic acid (representative VFA) is a result of the solubilization of 80% of 162 g of cellulose and 20% of 132 g of hemicellulose. The stoichiometry of the hydrolytic step also shows that 162 g of cellulose and 132 g of hemicellulose consumes 18 g of water; hence the effective volumetric moisture content is also depleted in each timestep according to Equation 3.

\[
d\theta = \frac{18}{(A\times162 + (1-A)\times132) \rho_{H,O}} dS \tag{3}
\]

where \(\rho_{H,O}\) is the density of water.

Hydrolysis, acidogenesis, and acetogenesis are the first three stages of the biodegradation, which represent the depletion of the organic content and its transformation into VFA. These latter intermediate products serve as substrates for MB. However, high VFA concentration has inhibitory effects on these reactions, which is also taken into account in the model through an inhibitor factor \(P\).

An enzymatic hydrolysis function accounts for the influence of the changing digestibility of the degradable fraction, product inhibition and moisture content on hydrolysis:

\[
r_g = \theta_c b \phi^* P \tag{5}
\]

where \(r_g\) denotes the rate of VFA accumulation [\(\text{g.VFA.m}^{-3}\cdot\text{aqueous.day}^{-1}\)], \(\theta_c\) is the effective moisture content; \(b\) [\(\text{g.m}^{-3}\cdot\text{aqueous.day}^{-1}\)] is the maximum VFA growth rate or maximum hydrolysis rate under the most favorable environmental conditions, which normally occurs at the early stage of hydrolysis reaction; \(\phi^* = 1 - [(S_0 - S)/S_0]^{n}\) is the relative digestibility decreasing with the solid degradable matter (S) depletion; \(S_0\) [\(\text{kg.m}^{-3}\)] is the initial solid degradable fraction and \(n\) is the
structural transformation parameter. Finally, \( P = \exp(-k_{VFA}(c)) \) is the inhibition factor accounting for the inhibitory effect of high VFA concentration, in which \( k_{VFA}[\text{g.m}^{-3}] \) is an inhibition constant.

The last stage of the biochemical reactions occurring in the MSW transforms the VFA generated in the previous stage to MB. The MB production rate \( r_j \) is calculated through Monod kinetics as shown in Equation 6, and the VFA consumption rate, \( r_h \), is directly linked to the MB accumulation through a substrate yield coefficient \( Y \) as shown in Equation 6.

\[
    r_j = \frac{k_0c}{(k_MC + c)^m} \quad (6)
\]

\[
    r_h = \frac{r_j}{Y} \quad (7)
\]

where \( k_0 [\text{day}^{-1}] \) is the maximum specific growth rate, \( k_MC [\text{g.m}^{-3} \ \text{aqueous}] \) is the half saturation constant. The MB decay \( r_h \) is given by:

\[
    r_h = k_2m \quad (8)
\]

where, \( k_2 [\text{day}^{-1}] \) is the methanogen death rate.

Methane and carbon dioxide productions are estimated from the stoichiometry of the digestion process. In other words, assuming the solid degradable fraction to consist of cellulose and hemicellulose, then 1 mole of cellulose results in 3 moles of CH\(_4\) and 3 moles of CO\(_2\) and 1 mole of hemicellulose results in 2.5 moles of CH\(_4\) and 2.5 moles of CO\(_2\).

**Mechanical model**

The mechanical model has three main components: (1) the load-induced volumetric compression and shear behavior of the waste, (2) the compression induced by decomposition of waste, and (3) the time-dependent creep compression behavior of the waste. The total settlement \( (\varepsilon_T) \) experienced by MSW in landfills is a combination of load-induced plastic strain \( (\varepsilon_p) \), biodegradation induced plastic strain \( (\varepsilon_B) \) and strain due to time-dependent creep \( (\varepsilon_C) \).

\[
    \varepsilon_T = \varepsilon_p + \varepsilon_B + \varepsilon_C \quad (9)
\]

The mechanical model for MSW is simulated using a plane strain formulation of modified Cam-clay model to determine the elastoplastic behavior of the MSW. The load induced model framework is a modification from the original HBM model, where a simple shear mode of shear deformation of MSW is considered, as opposed to a triaxial or direct shear failure. The load induced elasto-plastic behavior is not a focus on this paper.

The biodegradation-induced settlement is associated with loss of solids due to decomposition, denoted by \( dV_S \), as well as a change in void volume associated with decomposition, denoted by \( dV_r \). A constitutive relationship between decomposition of solid degradable fraction (i.e., a change in solid phase volume \( V_S \)), and the induced change in void volume \( V_r \) as proposed by McDougall and Pyrah (2004) and McDougall et al. (2013), is implemented through the introduction of \( \Lambda \), a decomposition (or degradation)-induced void change parameter defined as:

\[
    \Lambda = \frac{dV_r}{dV_S} \quad (10)
\]

The change in void ratio, \( de \), in the MSW matrix and the plastic strain due to biodegradation, \( e_B \) is:

\[
    e_B = \frac{de}{1 + e_0} = \frac{e - \Lambda}{1 + e_0} \cdot \frac{dV_S}{V_S} \quad (11)
\]

where \( e \) is the void ratio and \( e_0 \) is the initial void ratio.

The deformation characteristics of waste are not only dependent on the load-induced stresses and biodegradation, but also on additional deformation that takes place under a sustained load, typically referred to as creep. The sequential placement of waste lifts in the landfill induces time intervals of constant stress conditions that result in time-dependent creep compression. Creep behavior is incorporated within the HBM model using the ‘equivalent time’ method (Yin and Graham, 1989) when visco-plastic strains develop. ‘Equivalent time’ allows the creep strain rate of an over-consolidated material and its hardening to be related to the normal consolidation line at all stages of loading. Creep strains at constant effective stress for incremental loading are modeled by Equation 12:

\[
    \frac{de_c}{dt_{eq}} = \chi \left( \frac{t_{eq}}{t_{eq} + t_{ref}} \right) \quad (12)
\]

where \( de_c \) is the change in void ratio due to creep, \( t_{eq} \) is the equivalent time, \( t_{ref} \) is the reference time to indicate when creep straining commences and \( \chi \) is the creep viscosity coefficient, which is considered a material parameter.

3 **MODEL CALIBRATION BASED ON LABORATORY DEGRADATION EXPERIMENTS**

The coupled model requires degradable fraction of MSW solids (\( S_d \)), degradable (\( \rho_d \)) and inert (\( \rho_i \)) phase densities of the waste specimen, and volumetric moisture content (\( \theta_E \)) as initial inputs, in addition to the as-placed characteristics of the waste, i.e., total mass and volume. These six input parameters describe the physical characteristics of the MSW material and can be calculated from the initial waste composition and experimental data. It is important to note that the initial amount of degradable solids in an MSW specimen directly influences the cumulative methane yield at the end of degradation and hence, estimating the amount of
The experimental data is described in detail by Fei and Zekkos (2018) and is shown in Fig. 2a. Briefly, a 30-cm in diameter and 60-cm in (initial) height laboratory column with volume of ~42 liter was used to carefully conduct degradation experiments. A range of waste compositions from different landfills in the US were tested and the experimental datasets involved continuous measurements related to the biochemical, physical and mechanical processes in the waste for testing durations between 2.5-4 yrs. Degradation testing involved the placement of material at a loose state, with temperature control and the recirculation of leachate three times a week with short-term (~15 min) submergence of the specimens followed by gravity drainage of the liquids. Leachate samples were analyzed for pH, soluble chemical oxygen demand (sCOD), and VFA concentrations. The specimen’s total weights at field capacity and when submerged were measured with time using weighing scales on which the set-ups were placed. The settlement in the specimen was measured continuously using a cable extension transducer positioned above the column. Biogas composition was measured by a gas chromatograph and biogas volume was measured by a mass flow meter, adjusted to standard temperature and pressure. The experiments were considered complete after no additional biogas generation was observed and further settlement was considered minimal. In this paper, for brevity, only results for a waste-rich (Los Reales Landfill in Arizona -AZ), soil-rich (Lamb Canyon Landfill in California -CA2), and intermediate (Austin Community Landfill in Texas -TX2) waste composition are presented. The initial dry waste composition, average dry moisture content, and average volatile solids (VS) content of the specimens are provided in Table 2. In addition, three parameters, introduced by Fei and Zekkos (2018) to quantify biodegradability of each specimen prior to degradation based on initial waste composition, which are percentage of biodegradable fraction ($B_0$), density of biodegradable fraction ($\gamma_{BI}$) and volatile solids of biodegradable fraction ($V_{SB, I}$) are used. Higher values of $B_0$, $\gamma_{BI}$ and $V_{SB, I}$ are indicative of higher biodegradability materials.

A two-dimensional finite element mesh geometry with triangular elements (maximum size 0.03 m) is created in COMSOL to represent the laboratory simulators (h = 0.6 m, d = 0.3 m) as shown in Figure 2b. The model domain is extended vertically to twice the height of the specimen to avoid boundary/edge effects on the simulation results. The lateral boundaries are impermeable. To model leachate inflow for 15 minutes into the column with a pumping rate of 350 ml/min, an equivalent mass flux of 0.0825 kg/m²/s is applied at the top boundary. A piecewise function is multiplied with the mass flux to simulate leachate recirculation periodically. The lower boundary is set up

degradable solids from the waste composition correctly is important. Datta et al. (2018) reported an overprediction in methane generation compared to experimental data, but subsequent work found that the initial degradable solids in the specimens were overestimated.

In addition, as shown in Table 1, the developed model has a total of 12 process modeling parameters, without considering the parameters needed for the hydraulic and mechanical components of the model. The large number of modeling parameters can be cumbersome for field implementation. However, for practical purposes, the range (or uncertainty) in some of these parameters is such that does not significantly affect the results. A major challenge in calibrating any of the available coupled models is the lack of comprehensive experimental data. In this study, a unique, comprehensive set of experimental test data on six large-size laboratory specimens with well-defined waste composition that was generated by Fei and Zekkos (2018) is leveraged and is used to derive calibrated model parameters. This was done already by Data et al. (2018) using a 1D formulation of the model, where the entire experimental column is considered as one element. In this study, the model is formulated in 2D with the goal to allow for subsequent modeling of more complex, spatially variable field conditions.

Table 1. Model parameters for coupled process model and estimated values based on laboratory and field implementation.

| Parameters                        | AZ  | CA2 | TX2 | Deer Track |
|-----------------------------------|-----|-----|-----|------------|
| Initial VFA concentration ($c_0$), g/m³ | 0   | 0   | 0   | 0          |
| Initial MB concentration ($m_0$), g/m³ | 450 | 80  | 30  | 1000       |
| Maximum hydrolysis rate ($b$), gVFA/m/d | 7000 | 700 | 2100 | 6000 |
| Product inhibition factor ($k_{P_{VFA}}$), m³/g | 8E-5 | 1.5E-3 | 2.4E-4 | 1E-5 |
| Structural transformation parameter ($n$) | 0.65 | 0.45 | 0.55 | 0.55 |
| Maximum specific growth rate for cellulose ($k_0$), day⁻¹ | 0.09 | 0.27 | 0.15 | 0.10 |
| Methanogen death rate ($k_d$), day⁻¹ | 0.005 | 0.008 | 0.005 | 0.005 |
| Half saturation constant ($k_{sAC}$), g/m³ | 4000 | 400 | 1000 | 5000 |
| Cell/substrate yield coefficient ($I$) | 0.3 | 0.3 | 0.3 | 0.2 |
| Decomposition induced void change parameter ($f$) | -0.9 | -0.4 | -0.7 | -0.3 |
| Creep viscosity coefficient ($\gamma$) | 0.15 | 0.04 | 0.08 | 0.12 |
| Reference time ($t_{ref}$), day | 150 | 150 | 200 | 900 |
as a mixed boundary condition (pervious layer) switching between no drainage and flow, when the drainage valve is closed and specimen is submerged for 15 minutes during every recirculation, and free drainage, when the waste is drained by gravity. To model no flow condition, the conductance of the bottom boundary is set to zero. To create free drainage condition, the conductance is set to a high number and hydraulic head is set to only elevation head because pressure head is zero (or atmospheric). The initial condition in the hydraulic model is set to the initial moisture content of the specimen, provided in terms of pressure head.

Table 2. Waste composition parameters for three specimens of variable waste composition.

| Parameters                  | AZ     | CA2    | TX2    |
|-----------------------------|--------|--------|--------|
| <20 mm particles (% dry)    | 67.9   | 68.3   | 79.1   |
| Paper (% dry)               | 21.4   | 4.6    | 10.7   |
| Soft plastic (% dry)        | 8.5    | 3.9    | 5.9    |
| Wood (% dry)                | 2.1    | 5.8    | 4.3    |
| Average wc (% dry)          | 27.5   | 24.6   | 34.6   |
| Average VS (% dry)          | 26.5   | 11.0   | 13.9   |
| Bo (% dry)                  | 30.6   | 10.3   | 16.4   |
| γB,I (kg/m³)                | 141    | 66     | 97     |
| VSBo (% dry)                | 26.4   | 7.7    | 12.2   |

3.1 Model parameters

Saturated hydraulic conductivity ($k_{sat}$) is assumed to be constant ($\sim 10^{-2.5}$ cm/s) which is representative of the measurements from falling head tests conducted during the experiments by Fei (2016). In general, the measured $k_{sat}$ values ranged from $10^{-1} - 10^{-3}$ cm/s for all specimens. Saturated volumetric moisture content ($θ_s$) is considered equal to the porosity of the waste and residual moisture content is assumed to be 5%. In the absence of water retention curves for the waste specimens, Van Genuchten model parameters $α$, $n'$ and $l$ are suitably assumed from literature data (Beaven et al., 2008; White et al., 2015; Breitmeier et al., 2019). $n'$ is assumed to vary between 1.7 and 1.9 and $l$ is taken to be constant and equal to 0.5 for all specimens. Air entry suction parameter ($α$) is selected such that the field capacity moisture content rendered by the model is close to the laboratory measurements when the specimen is drained. Values of $α$ selected from literature range from 2.5-4.5 m⁻¹. As shown in the governing equations 1a and 1b, VFA and MB transport in the pore fluid is governed by diffusion, advection, growth and decay. It is reasonable to assume that the pore fluid velocity and the diffusion coefficient of the chemical species in the fluid are generally low enough that they do not affect the spatial distribution of the VFA and MB across the waste mass considerably (Kumar et al., 2020). The growth and decay behavior of VFA and MB is more dominant than the diffusion behavior (McDougall, 2007). Diffusion coefficient of VFAs in aqueous medium is proposed to be in the range of $10^{-6}$ cm²/s (Vavillin et al., 2003), is assumed in this study for both VFA and MB transport and is found to have negligible influence on the biochemical behavior. The advective transport is governed by leachate flow and is incorporated in the hydraulic model.

Fig. 2. (a) Experimental setup showing waste column for large-scale laboratory study by Fei and Zekkos (2018); (b) 2D model geometry showing FE mesh for waste column and initial and boundary conditions.
An illustration of the calibration process is shown in Figure 3. The six input parameters that describe the physical characteristics of the MSW were used. The process model parameters were adjusted \((m_0, b, k_{FEA}, k_0, k_{MC}, \Lambda\) and \(\chi)\) to assess whether the model has the capacity to match the laboratory experimental data. Then, the model parameter values that matched the experimental data the best for each MSW specimen are the final calibrated parameters and are listed in Table 1.

Example of the calibration results showing the coupled model behavior for the three MSW specimens is shown in Figures 4-6. The degradation process begins with the hydrolysis of organic fraction of MSW leading to VFA formation, which when depleted leads to the accumulation of methanogenic biomass and onset of methanogenesis. As a result, the organic solids are depleted and converted into biogas leading to a change in waste volume and settlement. The model is calibrated against VFA concentration, solids depletion, and settlement. Methane production (i.e., energy output) is an outcome of the model. Note that methanogenic biomass data were not measured during all experiments and hence, only model predictions of MB concentrations are shown.

Figures 4-6 (top left plots) show the evolution of VFA as a function of time. Initial VFA and MB concentrations are tuned to best match the experimental data. The VFA concentration starts from an initial value, reaches a peak and decays reaching a long-term residual value indicative of established methanogenesis. The AZ specimen produced the highest VFA (peaks at 13,500 g/m³), since more biodegradable material is available for hydrolysis and fermentation to take place, while CA2 produced the least (peaks at 450 g/m³). The maximum VFA concentrations are constrained by the product inhibition factor \((k_{FEA})\), whereas their subsequent decay is triggered by MB accumulation. The residual input parameters were therefore tuned by a combination of an increase in the maximum hydrolysis rate (to stimulate initial VFA accumulation), a reduction in the product inhibition factor (to allow higher peak
VFA concentration) and an increase in methanogen growth rate (to accelerate MB accumulation and thereby accelerate post-peak decay in VFA concentrations). Note that subsequent analyses demonstrated that the model predictions were not significantly affected by initial VFA concentrations. Therefore, for consistency, all specimens have been assigned an initial zero VFA concentration, assuming the specimens did not start degrading prior to the experiment.

Figures 4-6 (middle left plots) show predicted methanogenic biomass (MB) accumulation. MB concentrations start accumulating after VFA concentrations begin to decay. The initial MB concentration influences VFA accumulation and, through product inhibition, controls hydrolysis, subsequent MB growth and the onset of methanogenesis (McDougall and Philp, 2001). It is evident that the most biodegradable AZ specimen produces the maximum MB accumulation (peaks at 8500 g/m$^3$) and CA2, the least biodegradable, produces the minimum (peaks at 1000 g/m$^3$).

Figures 4-6 (bottom left plots) illustrate the stock of SDF, which becomes almost completely mineralized during the course of the experiments (250 days) and all metabolic processes slowed down as a result. The initial ‘rise’ in the experimental data during the first 20–30 days is due to the initial adjustment of the solid components in the loose waste matrix and changes in the volume during the first one to three leachate recirculation events, after which the biodegradation process begins. The waste-rich AZ specimen has the maximum initial stock of organic fraction (65 kg/m$^3$), while the soil-rich CA2 specimen has the minimum value (10 kg/m$^3$). Also, with an increase in SDF availability in waste, the solids depletion rate increases. AZ solids deplete faster than TX2 which depletes faster than CA2. It is observed that MB growth depends on the amount of SDF remaining. With the complete depletion of solids at around 250-300 days, the MB concentrations also decay to negligible values. Note that during the experiments, there was no direct measurement of solids depletion. However, SDF is calculated indirectly by considering the consumption of degradable solids being proportional to the measured biogas production.

Methane generation is an output of the modelling effort and governed by the depletion of the solid degradable fraction. Figures 4-6 (bottom right plots) show the cumulative methane generation normalized by the initial specimen wet weight predicted by the model and compared with the experimental values. It is observed that the model predicts the methane generation reasonably well. As waste becomes more degradable, methane generation increases. Methane generation ranges from 6 L/kg wet for CA2 specimen (with 2% degradable solids) to 51 L/kg wet for AZ (with 16% degradable solids).

Figures 4-6 (top right plots) show the model calibration of the biodegradation and creep settlement against the measured settlement data. In these experiments, immediate compression occurs as a result of loose placement of the waste material in the column and due to changes in volume owing to softening associated with moistening, waste structure adjustment and particle movement, particularly during the first very few (one to three) recirculation events (in the first five to ten days). Active biodegradation occurs when most microbial species reach their maximum growth rates, and a robust microbial community has been established. Residual compression occurs when settlement slows down due to retarded microbial activity and creep becomes the major contributor to the settlement. In this study, only active biodegradation strain and residual compressions (in terms of creep strain) have been incorporated and presented. Immediate compression is not modeled in this study.

Biodegradation-induced strain, as predicted by the model, is a function of the volume of solids and the void ratio. The model trends are generally consistent with the experimental data. AZ (waste-rich) attained the maximum settlement (in terms of strain), while CA2 (soil-rich) attained the minimum settlement. The model predictions for the final settlement are close to the experimental values, with a difference ranging from 1.8%-7%. For example, for TX2, the final strain in the experimental setup was recorded as 15%, while the model predicts a strain of 12%.

The decomposition-induced void change parameter ($\Lambda$) is an effective constitutive link between the mechanical consequences of decomposition and their biochemical causes (McDougall and Pyrah, 2004). $\Lambda$ values are obtained in this study (Table 1) by fitting the slope of the biodegradation-induced settlement curve of the experiment with the model. As per Equation 10, when $\Lambda = -1$, the loss of solids results in a direct and equal increase in void volume, i.e., there is no change in overall volume and the increase in void ratio is at a maximum. An increase in void ratio occurs at all values of $\Lambda < e$, with the magnitude decreasing until $\Lambda = e$, whereon there is no change in void ratio. At higher $\Lambda$ values, the loss of solid matter leads to a more compact particle arrangement. $\Lambda$ values for the specimens range from -0.9 to -0.6, which indicates loosening and possible weakening of the MSW after degradation (McDougall, 2007; McDougall et al., 2013). Finally, the creep viscosity coefficient ($\chi$) (as shown in Table 1) ranges from 0.04 to 0.15 with waste-rich material having generally higher creep coefficients than other specimens. Reference time ($t_{ref}$), the time when the specimens start experiencing creep strains is around 150-200 days. Overall, it is observed that as organic material increases, the initial solid degradable fraction ($S_0$), the initial methanogenic biomass concentration
(m₀), maximum hydrolysis rate (b), and creep viscosity (j) increase. On the other hand, as SDF availability increases, the product inhibition factor (kVFA) decreases. Peak VFA concentrations are inversely related to the kVFA value, so more biodegradable material has higher peak VFA concentrations owing to greater hydrolysis rates. The half saturation constant (kMC), which defines the substrate concentration at which the specific cell growth rate (r) is exactly half its maximum value (k₀), i.e., k₀/2, is found to increase with increasing waste biodegradability.

3.2 Spatial variation results

Although the results presented earlier provide a macroscopic (or average) assessment of waste behavior, the spatial evolution of the coupled degradation process is of interest, particularly in support of the implementation of the model in field conditions. Figure 7 illustrates results at the centerpoint of the TX2 specimen with leachate being recirculated at every 5 days for a duration of 150 days. Note that in the actual experiment, leachate is recirculated every 3 days, but for computational efficiency purposes, the recirculation frequency is modified. Parametric analyses conducted by Datta (2021) indicated that the evolution of the processes is very similar for 3-day and 5-day recirculation intervals. The specimen begins with an initial moisture content of 20% and moisture content spikes up at the first recirculation. The initial peak of about 63% represents the saturated volumetric moisture content (θₛ), when the specimen is allowed to be submerged for 15 minutes during recirculation. θₛ is observed to be gradually decreasing with time since it is a function of the porosity of the specimen, which decreases as degradation proceeds, mass of solids is gradually lost and the specimen settles. The specimen reaches about 35% moisture content (approaching but not reaching field capacity in the experiment) in between recirculations when it is drained by gravity. The effective hydraulic conductivity shows alternate cycles of saturated hydraulic conductivity on the order of 10⁻²⁵ cm/s when the specimen is submerged and unsaturated hydraulic conductivity on the order of 10⁻⁴ cm/s when it is drained by gravity. The methane generation predicted by the model is in close agreement with the experimental data.

Figure 8 shows the spatial variation of moisture in TX2 specimen after leachate is recirculated every 5 days since the beginning of the experiment. As a reminder, the lower part of the model is included to ensure that the drainage of the waste column is properly modeled and does not represent the leachate behavior of the specimen. It is observed that on days 5, 50 and 100, the specimen is at saturation since the drainage valve at the bottom is closed to allow complete submergence of the specimen. Hence, the moisture content on these days represents the saturated moisture content (θₛ). 3 days later (on the right-hand side of the figure), it is observed that the specimen has drained significantly (close to field capacity) to about 35% moisture content. As previously mentioned, θₛ is dependent on the porosity of the specimen. With degradation, the waste matrix undergoes compression due to mass loss and porosity gradually decreases leading to decrease in θₛ. For example, on day 5 θₛ is around 62.5%, on day 50 it is 61.4% and on day 100 it is 60%.

Figure 9 shows the evolution of VFA concentrations during degradation. The VFA formation occurs due to hydrolysis of the organic material in MSW. Due to leachate recirculation, the degradation process is accelerated and high VFA concentrations develop early on in the process as observed in days 5 and 20, where concentrations are ranging between 2,400 mg/L – 3,700 mg/L in the specimen, beyond which they start decreasing as the hydrolysis process nears completion as seen in days 50 and 100 where VFA concentrations are in the range of 340 mg/L and 50 mg/L, respectively. By 150 days the concentrations are negligible indicating complete hydrolysis.

Figure 10 shows the variation of cumulative methane generation along specimen height, which is practically uniform within the specimen (upper part of model). At 5 days and 20 days, when the VFA concentrations are high, the acidic environment inhibits methanogens to thrive and hence, methane production is low (~50-90 L). Once, the hydrolysis is completed, active methanogenesis dominates and higher methane yield are observed in the specimen as observed in 50 days (220-240 L) and 100 days (450-460 L). The difference in methane yield from 100 to 150 days is not significant (<2%) indicating that the majority of the
organic solids have been depleted leading to a decrease in gas production rate.

As expected, the biodegradation induced settlement observed in the specimen is in agreement with the coupled behavior as shown in Figure 11. During the hydrolysis phase (5 and 20 days), the settlements observed are low (0.7-1.3%). During active methanogenesis, with more solids being depleted waste settlements are higher (3.2%-6.3%) as observed at 50 days and onwards. It is to be noted that the simulation was run for 150 days to optimize computational cost since the majority of the biodegradation activity was completed within 150 days, although the experiment was conducted for a much longer duration (>1,000 days). The settlements recorded beyond this duration was primarily due to residual creep.
4 MODEL IMPLEMENTATION AT FIELD SCALE: DEER TRACK EXPERIMENT

Based on the model calibration of the laboratory experiments, it becomes evident that the model has the capacity to accurately capture the temporal variations of the degradation process. Thus, the model was implemented at field scale to demonstrate its capability to simulate field conditions. Specifically, a 37 m$^3$ field-scale lysimeter at the Deer Track Park (Watertown, Wisconsin) conducted by Bareither et al. (2012) was modeled. There are two main differences between the laboratory-scale modeling and the field-scale modeling. First, the lysimeter is two orders of magnitude larger in volume than the laboratory experiments. Second, as we transition from laboratory scale to field scale, the ability to control and monitor the experiment is reduced, leading to more limited measurements compared to the laboratory conditions. Despite these limitations, modeling field conditions (with inherent uncertainties and lack of proper control) is critical for full-scale energy harvesting implementation.

4.1 Field experimental setup and monitoring

The Deer Track Bioreactor Experiment (DTBE) consists of a cylindrical steel drainage lysimeter that was placed in an operating landfill and filled with MSW. The leachate in the MSW was introduced in seven quarterly doses starting on Day 399 of the experiment. During the experiment, the average dry unit weight of the waste was reported to increase by 28% and the dry-weight water content increased by 18%; field capacity of the waste was reported to be 44% to 48% on a dry-weight basis. The pH of the effluent increased, whereas biochemical oxygen demand (BOD), chemical oxygen demand (COD), and BOD to COD ratio all decreased during dosing. The DTBE was intended to replicate, as closely as practical, full-scale landfill conditions while maintaining a defined experimental control volume for assessing mechanisms of waste behavior.
A schematic of the field setup is provided in Figure 12. The lysimeter was a steel cylinder of 8.2-m height and 2.4-m diameter placed vertically in the waste mass of an MSW landfill to maintain typical landfill temperatures. Approximately 0.5 m of gravel was placed at the base of the lysimeter for leachate drainage. Waste excavated from the landfill was used for filling the lysimeter in 0.3-m-thick waste lifts. The waste sampled for this study had been in place in the landfill approximately 4 months before excavation for placement in the lysimeter. The lysimeter contained 27.2 Mg of waste, the initial waste thickness was 6.85 m, and the corresponding average dry unit weight was 6.3 kN/m³. A detailed analysis of the initial waste composition is described in Bareither et al. (2012). Composite waste specimens were analyzed for water content, VS, biochemical methane potential (BMP), cellulose, hemicellulose, and lignin contents. The initial waste composition on a dry weight basis consisted of paper (16.1%), flexible plastic (4.9%), rigid plastic (4.6%), textile (2.7%), wood (7%), gravel/inert (8.2%), yard waste (0.1%), food waste (0.2%), metal (5.3%), glass (0.4%), miscellaneous (2.3%) and <25 mm fraction (48.2%). The miscellaneous fraction was equally divided into degradable and inert components for the analysis. The initial average moisture content was 33% (dry basis) and the initial VS of the mixed waste and <25 mm fraction was measured as 52.9% and 16.5%, respectively. Food waste was negligible, probably degraded before the waste was placed in the lysimeter. The larger contribution of <25-mm material was caused by cover soil left in place during waste filling. The average cellulose, hemicellulose, and lignin contents of the waste were 19.6%, 5.4% and 28.8%.

The DTBE was instrumented to monitor water content, pore pressure, temperature, and settlement at various depths in the waste column. Gas generation and composition were not monitored. Positive pore pressures were never measured by the pressure transducers installed within the waste (i.e., the waste remained unsaturated). Leachate from the lysimeter was collected in a dosing basin to monitor effluent volumes and to obtain samples for chemical analysis (pH, COD, BOD, ammonia). Water content and matric suction were monitored by using time domain reflectometry (TDR) sensors placed at a depth of 4.7 m and 6.6 m.

### 4.2 Model parameters and initial conditions

To simulate the entire degradation process in the DTBE experiment, a two-dimensional finite element mesh geometry with triangular elements (maximum size 0.36 m) is created in COMSOL. Figure 12 shows the waste column of diameter 2.4 and thickness 6.85 m modelled in the FE based platform COMSOL. The model domain is extended vertically by 4 m to reduce any boundary/edge effects on the simulation results. The lateral boundaries were subjected to no flow and the lower boundary is set up as free drainage surface where the leachate from the lysimeter drains under gravity. The scheduled leachate dosing is injected from the top boundary by simulating a time dependent mass flux function. Over the first 399 days, no leachate was added to the DTBE to allow the system to reach an equilibrium water content distribution after construction. For the degradation model all boundaries were subjected to no flow due to advective-diffusive transport of VFA and MB.

The van Genuchten model parameters: air-entry suction ($α$) and curve fitting parameters- $n'$ and $l$ for the DTBE experiment are adopted from the water retention curves derived using inversion analysis from TDR measurements as described in Breitmeyer et al. (2020). $n'$ was considered to be constant at 1.4 while $α$ and $l$ varied with each leachate dosing. $α$ ranged from 0.08 to 0.17 and $l$ varied from 0.0004 to 4.49. Note that the inversion analysis was done by Breitmeyer et al. (2020) considering DTBE domain into two conceptual zones. The model parameters in this study for the entire domain are adopted as the average values of the upper zone which is around 5 m thick. The saturated hydraulic conductivity ($k_{sat}$) values decrease with density or void ratio ($e$) as degradation proceeds and are determined using the following equation derived by Breitmeyer et al. (2019) for a wide range of laboratory and field-scale hydraulic conductivity tests conducted on the DTBE specimen.

$$k_{sat} = 1.5 \times 10^{-4} \left( \frac{8.36}{1 + e} \right) m/s$$  \hspace{1cm} (13)

The saturated volumetric moisture content ($θ_s$) is equal to the porosity of the specimen and the residual moisture content ($θ_r$) is assumed as 5% as per Breitmeyer et al. (2020).

The degradation model requires degradable and inert solid fractions and the corresponding phase densities as initial conditions to be determined from waste composition. Paper, food, yard waste, textile and organic fraction of <25 mm material are considered as degradable components. Bareither et al. (2012) considered wood to be non-degradable in their experiment. The anaerobic biodegradability of different waste constituents is assumed based on literature data (Barlaz et al, 1990, Lee et al. 2017): paper is 50% degradable, food is on average 64%, yard waste is 25% and textile is 50% degradable consider half of it is cotton. The VS content of <25 mm fraction is considered as the degradable portion. Based on the above-mentioned assumptions, the waste is found to be around 20% degradable with 1120 kg/m³ and 1456 kg/m³ as the degradable and inert phase densities, respectively. The initial density of waste biodegradability ($γ_{bi}$) determined from the waste composition is 230 kg/m³. It is important to note that...
available DTBE data is not comprehensive enough to derive all model parameters accurately (e.g., there is no VFA concentration or biogas generation data recorded). Hence, judgement has been used to get approximate estimates of the parameters, specifically initial methanogenic biomass, maximum hydrolysis rate, product inhibition factor, half-saturation coefficient and yield coefficient.

DTBE experienced average immediate compression of 19% due to the placement of the upper gravel layer and self-weight of the waste in the first 1-2 weeks following waste placement. This short-term load induced compression is not modelled herein, as the focus is on the moisture distribution due to leachate recirculation and the subsequent influence on the coupled hydraulic, biochemical, and long-term compression behavior of the waste during the duration of the experiment.

4.3 Results

The evolution of the modelled biodegradation behavior in the lysimeter (at the centre point) is shown in terms of hydraulic characteristics (Figure 13-15), time dependent settlements (Figure 16), and biochemical characteristics (Figure 17). The seven leachate recirculation doses (shown with a dashed line in Fig. 13) and the corresponding changes in the effective saturation and volumetric moisture content during MSW degradation at a depth of 4.7 m in the lysimeter are shown in Figure 13. This depth was selected in order to compare the moisture content measurements measured by the TDR sensors at the same location during the experiment.

As observed in Figure 13b, the initial saturation in the specimen is predicted by the model to be approximately 35%. The specimen does not show a change in saturation until the leachate dosing is initiated at day 399. However, since moisture is absorbed at higher elevations and it takes time for the liquid to reach the specimen at a depth of 4.7 m, the
degree of saturation does not rise until the second recirculation at 500 days. This is in agreement with the TDR measurements (lower and upper bound) taken at the same depth, which show a spike in saturation around the same time. With subsequent leachate dosing, there is progressive wetting of the waste and the degree of saturation increases and eventually reaches field capacity after the 5th dose (at 70-78% saturation) as predicted by the model. Bareither et al. (2012) stated that the specimen reached field capacity after the 4th leachate dosing. With the assumptions considered in the model, this difference is not considered significant. When compared with the saturation range provided by the TDR measurements, the model prediction is in good agreement with the data.

Since the specimen is continuously draining by gravity, before the first dose at day 399 is injected, the initial moisture content drops as the specimen drains and attempts to reach field capacity as shown in Figure 13c. The moisture content profile after dosing also indicate progressive wetting until field capacity is reached. The field capacity moisture content achieved is 31-32% by volume or 38-39% by dry weight. The TDR measurements reported field capacity moisture content of 30-60% by dry weight.

The unsaturated hydraulic conductivity, \( k(\theta) \) is shown as a function of saturation in Figure 14. The \( k(\theta) \) values are on the order of \( 10^{-7} \) m/s to \( 10^{-8} \) m/s and increase with saturation or decomposition in this study. These fall within the range of hydraulic conductivity determined by Breitmeyer et al. (2020) using inversion analysis of the TDR measurements.

It is important to understand the effect of leachate recirculation and the corresponding effect of moisture distribution on the degradation along the depth of the waste column in the field-scale setup. The spatial variability is investigated in terms of effective saturation, methane generation and long-term settlements predicted by the model. Figure 15 shows the evolution of saturation in the specimen with leachate dosing and degradation. For the first 200 days, the saturation remained at 40% throughout the depth of the specimen. After the initiation of leachate dosing from day 399, on days 550 and 700 progressive increase in saturation of the specimen to around 60-70% is observed, as the liquid flows downward and drains by gravity. At the end of the experiment, on day 1050, the specimen reaches field capacity (75%-78% saturation).

Time dependent settlements were monitored at four different points along the depth of the lysimeter. In Figure 16, the average settlement value of the four points is shown. Suitable model parameters were chosen to best match this behavior. The decomposition induced void change parameter in the model (\( A \)) is found to be -0.3, which indicates softening/weakening of the waste post degradation. The onset of biodegradation induced settlement before leachate dosing (Day 399) is caused by anaerobic waste decomposition that began when the waste was originally buried and prior to placement in the lysimeter.

The biochemical behavior is shown in Figure 17. In the absence of VFA data which is the primary variable in the degradation model, the available pH and COD measurements were used as guidelines to make VFA predictions. The biochemical measurements were conducted after the first leachate dosing by Bareither et al. (2012). Although data is unavailable, if the pH data is extrapolated, it is most likely to be below 6.5 before first leachate dosing. Hence, acidogenesis and acetogenesis should occur during that period with VFA concentrations accumulating and then depleting. After the first dosing, pH values were close to neutral indicating active methanogenesis has already started. Peak VFA concentrations are generally 50-60% of COD concentrations as observed from large-scale laboratory studies described by Fei and Zekkos (2018) and Zhan et al. (2017). Although, it is difficult to estimate COD concentrations before leachate dosing, it is reasonable to assume that 5,000-5,500 mg/L is the peak value of COD, from where it subsequently decays. These assumptions were implemented when selecting model parameters to predict the VFA concentration behavior. The VFA concentrations peak at 2,200 mg/L at around 50 days and then decay to negligible levels at around 300 days.

The organic solids depletion and the corresponding methane generation predicted by the model is shown in Figure 18. 130 kg/m³ of solids (20% of solids are degradable) are completely depleted by day 800. The estimated methane yield is 26 m³ or 2.4 ml CH₄/g-dry VS. Biogas composition or measurements were not measured during the experiment. However, BMP tests conducted by the authors reported methane yield of 51 mL/CH₄/g-dry VS. The difference may be attributed to the fact that BMP tests are conducted under ideal operating conditions on a specimen size that is not
representative of the waste in landfills. Field-scale operations are suboptimal, and they are expected to generate biogas at a significantly slower rate than BMP tests.

Fig. 15. Spatial variation of effective saturation during degradation of MSW in the DTBE field-scale setup for day 1, 200, 550, 1050.

Fig. 16. Time dependent biodegradation and creep settlements measured in DTBE and predicted by the model.

Fig. 17. Biochemical characteristics of effluent leachate: a) pH and b) COD measured during the experiment in the effluent leachate and c) VFA concentrations predicted by the model.

In summary, the model is able to account for the coupled behavior observed in the lysimeter and provides insights on parameters that are not measured in the field. However, uncertainties exist and those can be reduced or eliminated with advances in monitoring.
5 CONCLUSIONS

A coupled hydraulic-biochemical-mechanical model that is based on the HBM model with some modifications, is implemented in large scale laboratory and field scale studies to understand the degradation process of MSW with the goal to support energy generation in modern landfills and the envisioned Sustainable Energy Reactor Facilities (SERFs). The biodegradable organic fraction in the waste is consumed by microorganisms, eventually leading to biogas generation, and changing solid waste and leachate characteristics.

The model was first tested against 0.04 m³ laboratory experiments to assess its ability to predict the observed behavior. The model was successful in capturing the coupled degradation behavior and yield reasonable energy predictions. The model was calibrated against large scale degradation experiments and reasonable range of model parameters were derived. It was found that several parameters are affected by waste composition. Specifically, with an increase in the biodegradability of waste, an increase in VFA and methanogenic biomass accumulation, an increase in the depletion rate of organic fraction, an increase in methane production and an increase in settlement is observed. The initial methanogenic biomass (\(m_0\)), maximum hydrolysis rate (\(b\)), product inhibition factor (\(k_{Ic}\)), half-saturation constant (\(k_{MC}\)), maximum specific growth rate (\(k_0\)), and the creep viscosity coefficient (\(\chi\)) are found to be waste composition dependent.

The model was then implemented on a 37m³ field lysimeter at Deer Track Park Landfill. It is to be noted that field-scale tests are operated with limited control on the process. Leakage of gas or leachate, suboptimal testing conditions and monitoring lead to partial understanding of the process. A comprehensive model such as the one formulated in this study helps to fill in the gaps by making reasonable predictions of the coupled process. The presented data indicates that the model could be used for field scale deployment and be used to optimize landfill management. The analysis presented provides insights on the interdependencies of the various processes that occur in a landfill which will help make decisions to optimize energy yield.

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