Waste Composition-Dependent "HBM" Model Parameters Based on Degradation Experiments

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Abstract

Municipal solid waste (MSW) is biodegradable in landfills under anaerobic conditions. The evolution of the hydro-biochemical-mechanical (HBM) processes during degradation is investigated first through experiments and subsequently via modelling. Three well-characterized MSW specimens with significantly different waste composition ranging from “waste-rich” to “soil-rich” were degraded in large-scale experimental setups that enabled simultaneous characterization of the processes with time. The closely-monitored processes are subsequently modelled using a two-stage anaerobic degradation model which is incorporated in the HBM model. This allows an assessment of model performance as a function of waste composition and derivation of waste composition-dependent model parameters. The model performed fairly well in capturing the biochemical and physical behaviour. An increase in biodegradable material in waste specimen corresponds to increase in anaerobic activity (volatile fatty acids and methanogenic biomass accumulation), higher rate of organic fraction depletion, increase in settlement and increase in methane production. However, the model is found to significantly over-predict methane production for all the specimens.

Keywords chosen from ICE Publishing list

Environmental Engineering; Energy; Geochemistry

List of notations

B_0 is the percentage of biodegradable waste

$\gamma_{d,I}$ is the dry unit weight of the specimen after immediate compression was practically completed in kg.m^{-3}

$\gamma_{B,I}$ is the density of biodegradable waste in kg.m^{-3}

c is the VFA concentration in aqueous medium in g.m^{-3}

m is the methanogen biomass (MB) in aqueous medium in g.m^{-3}

r_g is the rate of VFA accumulation in g VFA.m^{-3} aqueous.day^{-1}

θ_E is the effective volumetric moisture content

b is the maximum VFA growth rate in g.m^{-3} aqueous.day^{-1}

S is the solid degradable fraction in g.m^{-3}

n is the structural transformation parameter

ϕ is the relative digestibility of solid degradable fraction

k_{VFA} is the product inhibition constant in g.m^{-3}

P is the inhibition factor accounting for the inhibitory effect of high VFA concentration

k_0 is the maximum specific growth rate for cellulose in day^{-1}

k_{MC} is the half saturation constant in g.m^{-3}

r_j is the methanogenic biomass production rate in g.m^{-3}.day^{-1}

Y is the cell/substrate yield coefficient

k_2 is the methanogen death rate in day^{-1}

r_k is the methanogenic biomass decay rate in $g.m^{-3}.day^{-1}$

ρ_{H_2O} is the density of water in $kg.m^{-3}$

V_s is the solid phase volume

V_v is the void volume

Λ is the decomposition (or degradation)-induced void change parameter

e is the void ratio

χ is the creep viscosity coefficient

t_{eq} is the equivalent time in days

t_{ref} is the reference time in days

1. Introduction

Approximately 53% of the municipal solid waste (MSW) generated in the U.S. is disposed of in sanitary landfills (US EPA, 2014). The solid, liquid and gas phases of MSW in a landfill evolve with time due to coupled physical-biochemical-mechanical and hydraulic processes during anaerobic biodegradation (Fei and Zekkos, 2018). Under appropriate conditions, anaerobic microorganisms consume the biodegradable organic fractions including food waste, yard waste, paper and other organics eventually converting them to biogas. The process begins with bacterial hydrolysis of the complex organic waste into simpler forms, leading to mass loss and void creation in the waste matrix. As a consequence, physical properties, such as porosity and unit weight, change. Hydrolysis results in the dissolution of organic compounds in the leachate and is followed by acidogenesis and methanogenesis which produces biogas consisting primarily of methane and carbon dioxide (Barlaz et al., 2010). As a result, the phase relationships between solid, liquid and gas phases within the MSW matrix are altered. In order to understand, predict, and possibly control waste decomposition in sanitary landfills, one must first understand the anaerobic process of degradation occurring in the waste matrix. Batstone et al. (2002) developed a structured model, ADM1, including multiple steps describing biochemical as well as physicochemical processes in anaerobic digestion. Machado et al. (2008) developed a numerical model that incorporated the biodegradation of organic matter in landfilled MSW. Gawande et al. (2010) presented a numerical model, BIOKEMOD-3P, to investigate the biochemical processes with multi-phase systems that take place in bioreactor landfills. Chakma et al. (2016) developed a mathematical model to compute the chemical compositions of the organic portion of MSW.

Datta et al. (2017) studied the coupled degradation process and its evolution through numerical modelling of one MSW specimen with a unique composition. A biodegradation model was used to describe the anaerobic process and account for hydrolysis of degradable matter in relation to moisture content, product inhibition, cellulose digestibility and microbial controls. In this study, however, three significantly different waste compositions have been investigated in large-scale experimental setups and a two-stage anaerobic model was implemented and calibrated with the experiments. Specifically, the biodegradation component in the hydro-bio-mechanical (HBM)

formulation described by McDougall (2007) is adopted here to describe the hydrolysis, acidogenesis and methanogenesis reactions. The HBM model provides a constitutive framework for the integrated analysis of the hydraulic, biodegradation and mechanical behaviour of landfilled waste. HBM comprises three main component models that are coupled through link routines and continually updated to ensure each component model and its influence over others evolves during the simulation. The biodegradation component of the HBM model which is the focus of this study neglects the aerobic stage since the aerobic waste decomposition represents a minor part of the landfill lifetime and is considered less significant than anaerobic decomposition in terms of organic conversion. The outcome of this modelling effort is the derivation of model parameters that match the experimental data as a function of waste composition. In addition, limitations to the current formulation are identified.

2. Experimental Setup

2.1 Waste Characterization

The MSW specimens used in this study were collected from Austin Community Landfill in Texas (TX), Los Reales Landfill in Arizona (AZ) and Lamb Canyon Landfill in California (CA) and transported in sealed drums to the laboratory. Field composition was characterized according to the procedure described by Zekkos et al. (2010). The waste was segregated into finer and coarser fraction using a 20-mm sieve. The <20 mm fraction of each sample was soil-like. The >20 mm fraction was manually segregated based on waste constituent type. The three primary constituents by weight for all samples were paper, soft plastic and wood. Other minor constituents included hard plastic, metal, rock and miscellaneous objects. No distinguishable food waste was found in any sample. The segregated waste constituents from each sample were weighed and the corresponding percentages on a wet weight basis were calculated. The moisture content on a dry weight basis of the three primary >20 mm constituents and <20 mm fraction of each sample was measured by heating the material at 70°C. The content of volatile solids (VS) of each dried waste constituent was evaluated by heating the 70°C-dried waste at

550°C. The biodegradability of each specimen prior to degradation is evaluated using density of biodegradable waste ($\gamma_{B,I}$) (Fei, 2016) defined by equations 1 and 2.

$$B_0(\%) = \frac{Dry\ mass\ of\ \ food, yard\ waste, paper\ and\ mass\ of\ VS\ in < 20\ mm\ fraction}{Total\ initial\ dry\ waste\ mass} \quad (1)$$

$$\gamma_{B,I}(\frac{kg}{m^3}) = \frac{B_0(\%)}{100} \times \gamma_{d,I}(\frac{kg}{m^3}) \quad (2)$$

where, B_0 (dry mass/dry mass %) is the percentage of biodegradable waste prior to degradation and is defined as the proportion by dry mass of food, wood and paper plus the mass of VS in 20 mm fraction of the entire dry waste mass (Fei and Zekkos, 2018). $\gamma_{d,I}$ is the dry unit weight of the specimen after immediate compression was practically completed. The initial total mass, mass percentage, initial average moisture content, volatile solids (VS) in <20 mm fraction and initial density of biodegradable waste (Fei and Zekkos, 2018) are provided in Table 1. Briefly, AZ waste is "waste-rich" containing the highest amount of biodegradable material per unit volume ($\gamma_{B,I}$ = 140 kg/m^3), while CA waste contains the least biodegradable material per unit volume ($\gamma_{B,I}$ = 67 kg/m^3) and is "soil-rich"; TX waste is intermediately biodegradable ($\gamma_{B,I}$ = 97 kg/m^3).

2.2 Degradation testing of MSW

The specimens were reconstituted based on their field waste composition and placed in 42-L columns for degradation testing. Three 42-L (d=300 mm, h=600 mm) columns accompanied by systems to recirculate and store leachate, and systems to monitor the long-term waste degradation process including biogas, changes in solid weight and volume were constructed as shown in Figure 1 and is described in detail by Fei et al. (2014). The materials were placed at a loose state, i.e., without significant compaction. Besides the vertical load from a plastic leachate distribution plate and a stainless steel rod for settlement measurement that impose <1 kPa vertical stress, no additional vertical stress was applied to the specimens. The temperature of each column remained constant at 40±3°C using a heating blanket. Leachate was recirculated three times a week and involved short-term (~15 min) submergence of the specimens followed

by gravity drainage of the liquids through the base of the column. Thus, the specimens were maintained at field capacity in between submergence events. Leachate samples were analysed for pH and soluble chemical oxygen demand (APHA 2005). The concentrations of volatile fatty acids (VFAs) in the samples were analysed using an ion chromatography system. The specimen total weights at field capacity and when submerged were measured with time using the weighing scales on which the setups were placed. Biogas composition was measured by a gas chromatograph equipped with a thermal conductivity detector using nitrogen as the carrier gas. The settlement in the specimen was measured continuously using a cable extension transducer positioned above the column. The volume of generated biogas was measured by a mass flow meter and was adjusted to standard temperature and pressure. The experiments were considered complete after 885 days, 1500 days and 850 days of operation for AZ, TX and CA specimens, respectively, since no additional biogas generation was observed, and further settlement was considered minimal, and the setups were disassembled.

3. Modelling Framework

3.1 HBM-Biodegradation Model

The HBM biodegradation model (McDougall 2007) is incorporated in MATLAB, considering the specimen column as a single element, with time as the evolving parameter. The model describes a two-stage anaerobic digester in which volatile fatty acid (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 kinetics of the stoichiometry are controlled by growth and decay parameters for each of the biodegradation variables,

- c [g.m^{-3}] the VFA concentration in aqueous medium;
- m [g.m^{-3}] the methanogen biomass (MB) in aqueous medium.

as set out in the following sections. In general, reaction rate equations are expressed in g.m^{-3}aqueous.day^{-1}.

Cellulose and hemicellulose are the major biodegradable components of MSW. However, in this model, mineralisation of only cellulose has been considered which accounts for 91% of the methane potential for typical MSW (Barlaz et al., 1989). The process of cellulose degradation is idealised in three main steps:

A. Enzymatic hydrolysis of solid cellulose ($C_6H_{10}O_5$) to glucose ($C_6H_{12}O_6$):

$$C_6H_{10}O_5 + H_2O \rightarrow C_6H_{12}O_6$$

B. Fermentation of glucose to acetic acid (CH_3COOH) which is considered a representative VFA:

$$C_6H_{12}O_6 + 4H_2O \rightarrow CH_3COOH + 8H_2 + 4CO_2$$

C. Methanogenesis:

(i) By acetate-cleaving methanogen:

$$CH_3COOH \rightarrow CO_2 + CH_4$$

(ii) By hydrogen-scavenging methanogen:

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O$$

Assuming the fermentation process (acidogenesis and acetogenesis) is relatively fast compared to methanogenesis and hydrogen-scavenging methanogens consume all gaseous intermediates (step C (ii)), the overall stoichiometry is

$$C_6H_{10}O_5 + H_2O \rightarrow CH_3COOH + 8H_2 + 4CO_2 \rightarrow 3CO_2 + 3CH_4$$

3.1.1 Hydrolysis and acidogenesis/acetogenesis

Hydrolysis and acidogenesis/acetogenesis is the first stage of the biodegradation, which represents the depletion of the complex organic content and its transformation into VFA. These latter intermediate products serve as a substrate for methanogenic biomass. However, high VFA concentration has inhibitory effects on those reactions, which is also taken into account in the model through an inhibitor factor.

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

$$r_g = \theta_E b \phi P \quad (3)$$

where, r_g denotes the rate of VFA accumulation [g VFA.m^{-3} aqueous.day^{-1}] and the four governing factors are:

- θ_E is the effective volumetric moisture content which is determined from the hydraulic model.
- b [g.m^{-3} aqueous.day^{-1}] is the maximum VFA growth rate under the most favourable environmental conditions, which normally occurs at the early stage of hydrolysis reaction.
- $\phi = 1 - [(S_0\text{-}S)/S_0]^n$ is the relative digestibility decreasing with the solid degradable matter depletion, where S_0 [g.m^{-3}] is the initial solid degradable fraction, S is the solid degradable fraction at any time and n is the structural transformation parameter.
- $P = exp(-k_{VFA}(c))$ is the inhibition factor accounting for the inhibitory effect of high VFA concentration, in which k_{VFA} [g.m^{-3}] is an inhibition constant.

Note that in the modelling effort, because the specimen was completely submerged and then drained to field capacity, the flow/hydraulics component of the model is not needed and the effective volumetric moisture content (θ_E) is treated as the average field capacity moisture content of the waste specimen after recirculation.

3.1.2 Methanogenesis

The second stage of the biochemical reactions occurring in the MSW transforms the VFA generated in the first stage to methanogen biomass. The MB production rate r_j is calculated through a Monod kinetic equation and the VFA consumption rate r_h is directly linked to the methanogen biomass accumulation through a substrate yield coefficient Y.

$$r_j = \frac{k_0 c}{(k_{MC} + c)} m \quad (4)$$

$$r_h = \frac{r_j}{Y}$$

5.

where k_0 [day^{-1}] is the maximum specific growth rate, k_{MC} [g.m^{-3} aqueous] is the half saturation constant. The MB decay r_k is given by

$$r_k = k_2 m$$

6.

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

3.1.3 Governing Equations

The combined growth and decay of VFA and MB in the biodegradation model are described by the following two equations:

$$[r_g - r_h] = \frac{\partial c}{\partial t}$$

7.

$$[r_j - r_k] = \frac{\partial m}{\partial t}$$

8.

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 solid degradable fraction is depleted in each timestep using the following equation

$$S^{t+\Delta t} = S^t - \theta . \frac{162}{60} r_g \Delta t$$

9.

where, S^t is the solid degradable fraction remaining in timestep t and $t + \Delta t$ is the next time step. This is derived from the overall stoichiometry which indicates 60 g of acetic acid (representative VFA) is a result of the solubilisation of 162 g of cellulose. The stoichiometry of the hydrolytic step shows that 162 g of cellulose consumes 18 g of water; hence the effective volumetric moisture content is also decreased in each timestep by

$$d\theta = \frac{18}{162\rho_{H_2O}} dS$$

10.

where, ρ_{H_2O} is the density of water.

3.1.4 Methane Generation

In the HBM model, methane production (and carbon dioxide) is estimated from the stoichiometry of the digestion process. In other words, assuming the solid degradable fraction to consist purely of cellulose, then 1 mole of cellulose gives rise to 3 moles of carbon dioxide and 3 moles of methane.

3.1.5 Biodegradation-induced settlement

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_V (McDougall and Pyrah 2004), is implemented to calculate the change in void ratio in the MSW matrix and subsequently, the strain due to biodegradation ε_B is calculated.

$$dVv = \Lambda dVs$$

11.

$$de = (e - \Lambda)\frac{dV_S}{V_S}$$

12.

$$\varepsilon_B = \frac{de}{1 + e_0}$$

13.

where, Λ is the decomposition (or degradation)-induced void change parameter, e is the void ratio and e_0 is the initial void ratio.

3.1.6 Settlement due to time-dependent creep

Creep behaviour is incorporated within the HBM model using the 'equivalent time' method (Yin and Graham, 1989). '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 modelled by

$$\frac{de}{dt} = \left(\frac{\chi}{t + t_0 + t_{eq} + t_{ref}} \right) \quad (14)$$

where, e is the void ratio, t is the current time, t_0 is the time at which the current creep stage commences t_{eq} is the equivalent time, t_{ref} is the reference time to indicate when creep straining commences (a curve fitting parameter unique to each specimen) and χ is the creep viscosity coefficient. χ is a material parameter and constant for a given waste specimen. In this study, it is considered that the equivalent time is close to the duration of the increments since the specimen is in normally consolidated range of loading (Yin and Graham, 1994).

3 .2 Initial Conditions

The solid fraction is segregated into degradable and inert phases for input into the HBM model. Initial solid degradable fraction (S_0) and degradable phase density are determined from the waste composition as provided in Table 2. Determination of each phase density requires the density of each solid constituent in the waste (paper, wood, plastic and <20 mm fraction). Particles <20 mm (soil-like fraction) consist of inorganic soil intermixed with degradable and inert particles. The organic content (in terms of volatile solids) in that fraction varies from 7% to 13% in the three specimens. Since, the specific gravity (G_s) of soil decreases with increasing organic content (Radforth et al., 1996), the soil-like fraction G_s varies from 2.0 to 2.4 (solid phase density of 2000 kg/m^3 to 2400 kg/m^3). Nevertheless, it is observed that the modelling results are not affected significantly by different G_s for the range of waste composition considered. Based on this calculation, 24%, 12% and 9% by dry weight of the MSW solids are degradable in AZ, TX and CA specimens, respectively, as shown in Table 3.

3 .3 Modelling Strategy

As-placed physical waste characteristics, i.e., initial mass, height and volume, are defined. Initially a simulation was run adopting the default parameters as provided in McDougall (2007). However, the results fit the experimental data poorly which is not surprising. As a result, all the biodegradation parameters were adjusted to improve the fit between the HBM model and experimental data. The parameter values that most closely match the experimental data for the selected MSW specimens are provided in Table 4.

4. Results

4.1 VFA and MB concentrations

Figure 2 shows the evolution of VFA as a function of time for all specimens. A good fit is observed between the experimental data and the model prediction of the VFA concentrations. Initial VFA and MB concentrations are chosen as shown in Table 4 to best match the experimental data. The VFA concentration starts from an initial value, reaches a peak and decays soon reaching a long-term residual value indicative of established methanogenesis. It is interesting to note that initial VFA for "waste-rich" AZ specimen is high (8500 g/m^3) indicating that anaerobic process readily established in the waste mass. The AZ specimen produced the highest VFA (peaks at 13500 g/m^3) indicating more biodegradable material available for hydrolysis and fermentation to take place, followed by the intermediate biodegradable TX specimen (peaks at 4750 g/m^3) and then the "soil-rich" least biodegradable CA specimen (peaks at 450 g/m^3). The model predicts lower decay rates of VFA than the measured data in TX and CA specimens. For example, the measured VFA concentration in TX specimen reduces to a value close to zero at day 50, whereas the model reaches zero at around day 90. The maximum VFA concentrations are constrained by the product inhibition factor, whereas their subsequent decay is triggered by MB accumulation. The residual input parameters were therefore tuned by a combination of an increase in 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). Peak VFA concentrations increases with increase in maximum hydrolysis rates and decreases with increase in product inhibition factor (Table 4). As the waste becomes more degradable, maximum VFA

concentrations occur later. For example, AZ peaks after around 25 days while CA peaks after 5 days.

Figure 3 shows methanogenic biomass accumulation predicted by HBM. Initial MB concentration ranges from 1200 g/m^3 for "waste-rich" AZ specimen to 10 g/m^3 for "soil-rich" CA specimen as shown in Table 4. 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 more-biodegradable AZ specimen produces the maximum MB accumulation (peaks at 19000 mg/L) and CA produces the minimum (peaks at 4700 mg/L), with TX reaching an intermediate value (peaks at 8200 mg/L). However, it is interesting to note that, with the increase in biodegradability, the MB peaks are slightly delayed with AZ reaching the maximum in 85 days while TX and CA attaining its peak in 75 days. The methanogen yield coefficient (Y) controls the mass of methanogenic biomass produced per unit mass of substrate with AZ and TX having higher values than CA (Table 4). As the biodegradability of waste increases, an increase in half-saturation constant and methanogen death rate and a decrease in specific growth rate of cellulose is observed (Table 4). Note that MB concentrations were not measured during the experiment (Fei et al., 2015).

4.2 Solid degradable fraction

Figure 4 shows that the stock of solid degradable fraction or organic fraction was almost completely mineralised during the course of the experiment (in 300 days) and all metabolic processes slowed down as a result. The solid degradable fraction depletion predicted by HBM fit fairly well with the experimental data except for the initial 20-30 days. 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 1-3 leachate recirculation events, after which the biodegradation process begins. It is evident that "waste-rich" AZ specimen has the maximum initial stock of organic fraction (102 kg/m^3), while "soil-rich" CA specimen has the minimum value (50 kg/m^3). Also, with increase in biodegradability in waste, the solids depletion rate increases. AZ solids deplete faster than TX and CA solids. It is

observed that MB growth depends on the amount of solid degradable fraction remaining. With the complete depletion of solids at around 300 days, the MB concentrations also decay to negligible values. Note that during the experiments there was no direct measurement of solids depletion. However, it is calculated indirectly by considering the consumption of degradable solids being proportional to the measured biogas production which is a reasonable assumption considering the stoichiometry of the anaerobic process. It is interesting to note that the reduction of moisture at the end of degradation (Equation 10) is approximately 3.2% for AZ, 1.6% for TX and 1.4% for CA, which is not considered significant.

4.3 Methane generation

Figure 5 shows the cumulative methane generation for the specimen. As waste becomes more degradable, methane generation increases. It is observed that the total volume of methane collected during the experiment is 1110 L for AZ, 476 L for TX and 187 L for CA (methane: carbon dioxide measured approximately equal to 60:40) while the model predicts a total of 1681 L for AZ, 1110 L for TX and 879 L for CA (methane: carbon dioxide equal to 50:50). Clearly, methane production is over-predicted by the model – more for the “soil rich” specimen than “waste rich”, possibly due to the simplified stoichiometric assumption of solid degradable fraction of MSW consisting of only cellulose. Apart from cellulose, hemicellulose is another principal biodegradable component of MSW; while the other major organic component, lignin, is at best only slowly degradable under methanogenic conditions and acts as a recalcitrant. (Barlaz et al., 1990). In addition, the experimental methane yield could be lower due to a number of factors such as part of the organic material inaccessible due to binding in particles or structural organic matter, utilization of a fraction of the substrate to synthesize bacterial mass, ammonia toxicity or limitation of other nutrient factors (Angelidaki et al., 2004, Labatut et al., 2010).

4.4 Settlement

Figure 6 shows the model prediction of the evolution of the biodegradation and creep strain (settlement) along with the experiment settlement data. Long-term settlement of waste specimen is found to follow three phases – immediate compression, active biodegradation and

residual compression, as discussed in Fei and Zekkos (2013). 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 few recirculation events. 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 a major contributor to the settlement (Fei and Zekkos, 2013). For a typical waste composition as in AZ, immediate compression of the waste mass was observed between days 1-10. Active biodegradation took place from day 10 to day 270 and the remaining was residual compression.

In this study, only active biodegradation strain and residual compression (in terms of creep strain) has been modelled. Biodegradation induced strain, as predicted by the model, is a function of the volume of solids and the void ratio. The trend of the model is generally consistent with the experimental data. AZ ("waste-rich") has attained the maximum settlement (in terms of strain) while CA ("soil-rich") attained the minimum settlement. The model predictions for the final settlement (except immediate compression) are close to the experimental values. For example, for AZ, the final strain in the experimental setup was recorded as 32%, while the model predicts a strain of 28.5%. Immediate compression occurs in the specimens during the first 1-3 leachate recirculation events (till 5-10 days), which is not modelled in this study as shown in Figure 6. AZ and CA specimens have significant immediate strains of 10.9% and 10%, respectively, while TX has a noticeably low value of 1.5%.

The decomposition-induced void change parameter (Λ) is an effective constitutive link between the mechanical consequences of decomposition and their biochemical causes (McDougall and Pyrah, 2004). Λ values are obtained in this study (Table 4) by approximately matching the slope of the biodegradation-induced settlement curve of the experiment with the model. Values are similar for all three specimens (0.8 for AZ, 0.7 for TX and 0.8 for CA) and are estimated to be less than the specimen void ratio at any time (e estimated to be between 1.7 and 2.4 for AZ, 1.4 and 1.7 for TX, 1.7 and 1.8 for CA), which indicates loosening and possible weakening of the

material after degradation. Although the parameter values are not found to change significantly with waste composition in this study, factors such as compaction effort and vertical stress that were not investigated in this study, may influence them. In addition, a systematic trend in creep viscosity coefficient (χ) in the three specimens (Table 4) is observed with AZ having the highest value of 0.018 and CA the lowest value of 0.007. This is indicative of the fact that waste constituents have an impact on the time-dependent creep of a soil-waste mixture (Zekkos et al., 2016). For example, AZ has the maximum amount of paper and soft plastic than the other two specimens which results in significant creep in the specimen. Note that in this study, both the experiment and the model consider negligible vertical stress on the specimen.

5. Conclusion

The coupled biochemical-physical-mechanical processes documented during the degradation of MSW in a large-scale experimental setup is modelled using the HBM-biodegradation framework. The VFA and MB concentrations (biochemical) are dependent on the solid degradable fraction (physical), which is in turn coupled with biodegradation-induced vertical strain. For a given waste composition, the model parameters are calibrated against the experimental data. Influence of waste composition on this coupled behaviour has been investigated through three significantly different MSW specimens. The most significant findings from this work are as follows:

- The model predicts aspects of biochemical and physical behaviour fairly well.
- Biodegradation and creep induced settlement trends (mechanical behaviour) agree well with the experimental results.
- Specific gravity of soil-like (<20 mm) fraction (varying from 2.0-2.4) does not have significant effect on the degradation process.
- Decomposition-induced void-change parameter values do not change significantly with waste composition.
- With increase in biodegradability of waste, an increase in VFA and MB accumulation, increase in depletion rate of organic fraction, increase in methane production, and increase in settlement are observed.

- The model systematically over predicts methane production; more so for the “soil-rich” specimen than the “waste-rich”.

Note that the model formulation considered here is one-dimensional and time is the only varying parameter. This is adequate for the purposes of modelling these specific laboratory experiments. However, MSW in the field is essentially heterogeneous and anisotropic, hence, it becomes critical that spatial variability is considered as well. In addition, in this study, both experimentally and numerically, the degradation process was conducted under negligible vertical stress.

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Figure captions

Figure 1. Schematic of the experimental setup (Datta et al., 2017)

Figure 2. VFA concentrations measured experimentally and predicted by HBM-biodegradation model

Figure 3. Methanogenic biomass concentrations predicted by HBM-biodegradation model

Figure 4. Solid degradable fraction measured experimentally and predicted by HBM-biodegradation model

Figure 5. Cumulative methane generation measured experimentally and predicted by HBM-biodegradation model

Figure 6. Settlement (strains) measured experimentally and predicted by HBM-biodegradation model (*Immediate compression observed during the experiment, not modelled in this study)

Table captions

Table 1. Composition, moisture content, volatile solids and biodegradability of three MSW specimens.

Table 2. Example calculation of solid degradable fraction from initial waste composition of TX specimen.

Table 3. Initial modelling conditions for the three MSW specimens.

Table 4. HBM-Biodegradation calibrated model parameters for the three MSW specimens.

Figure 1

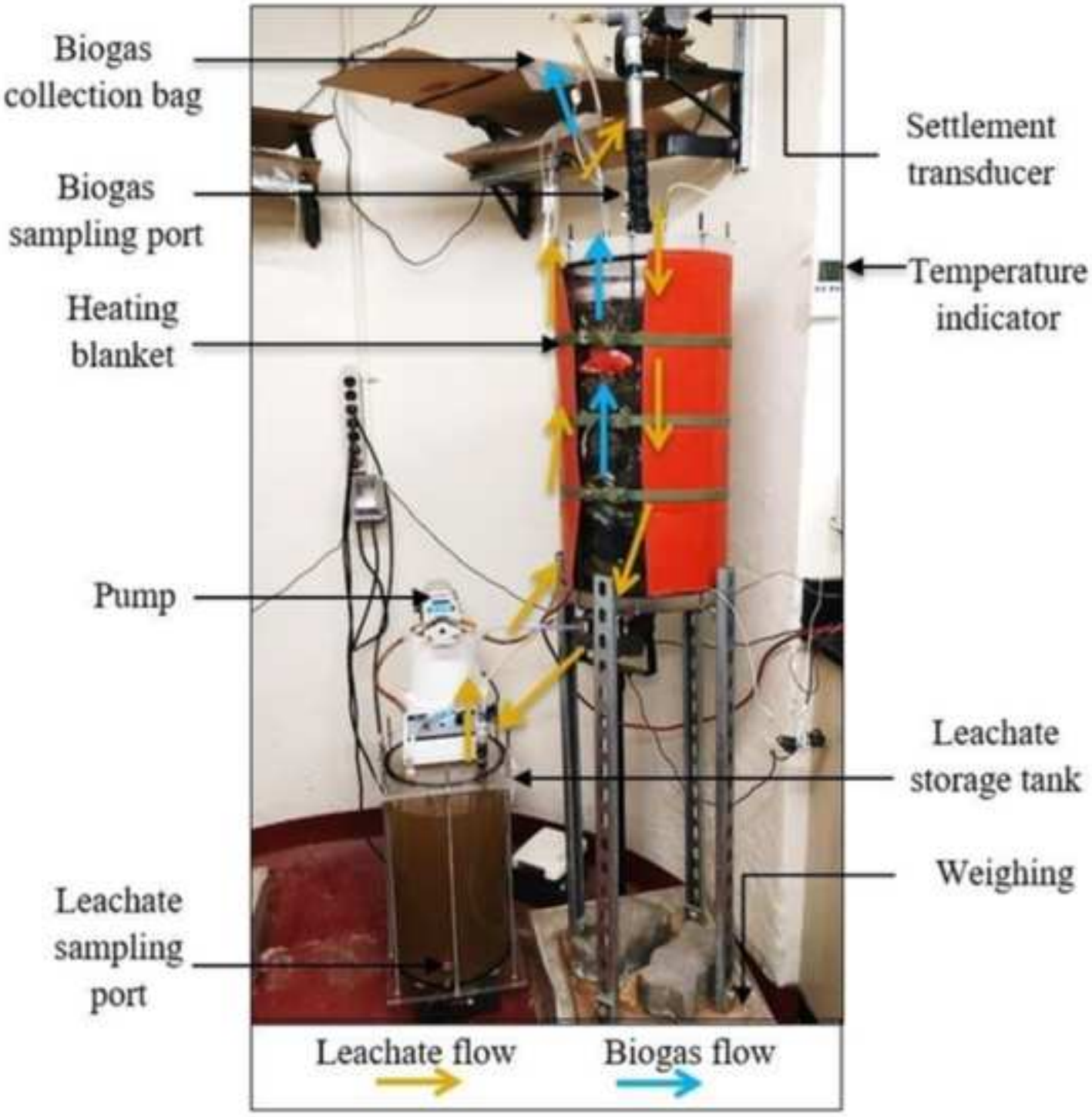

Figure 2

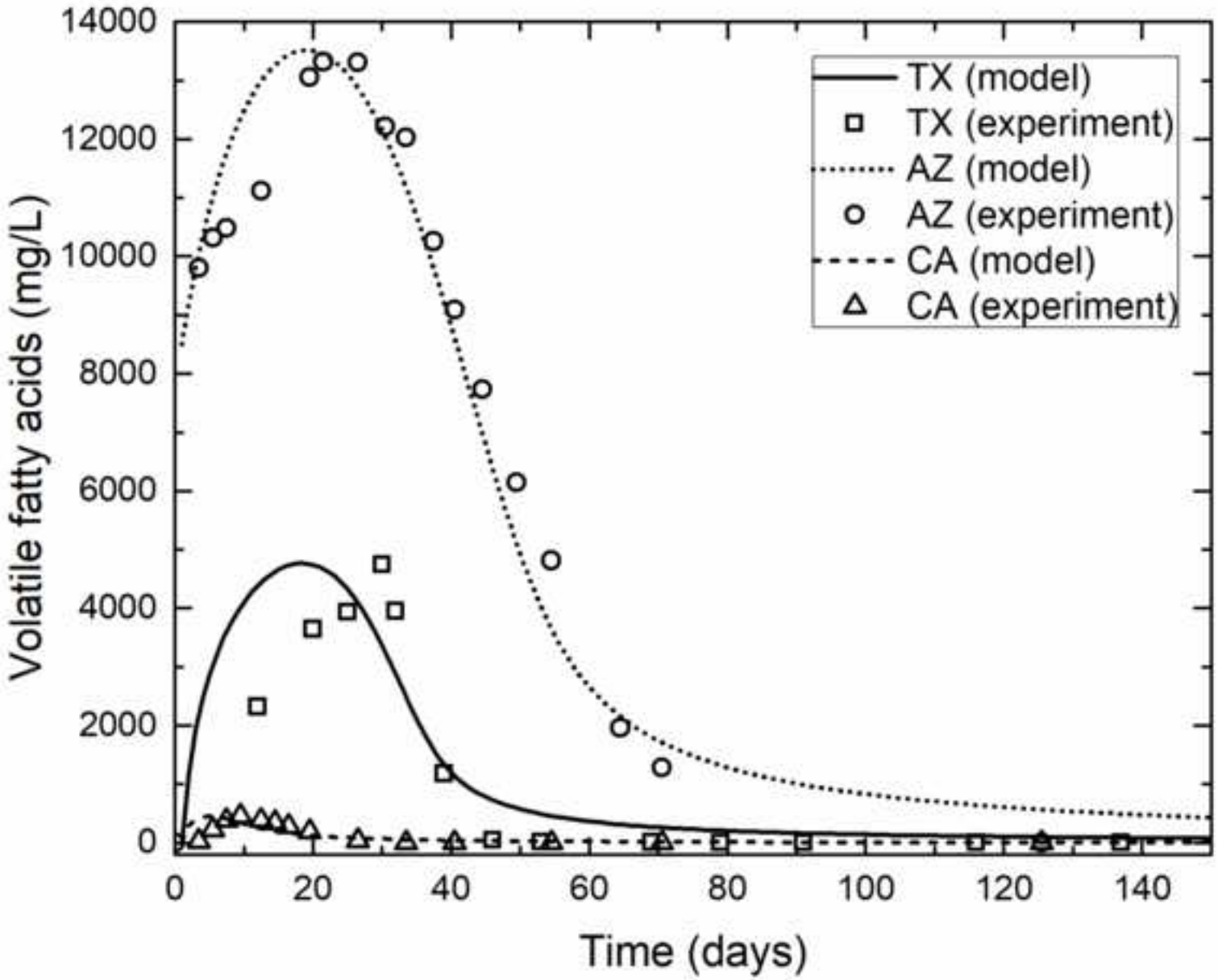

Figure 3

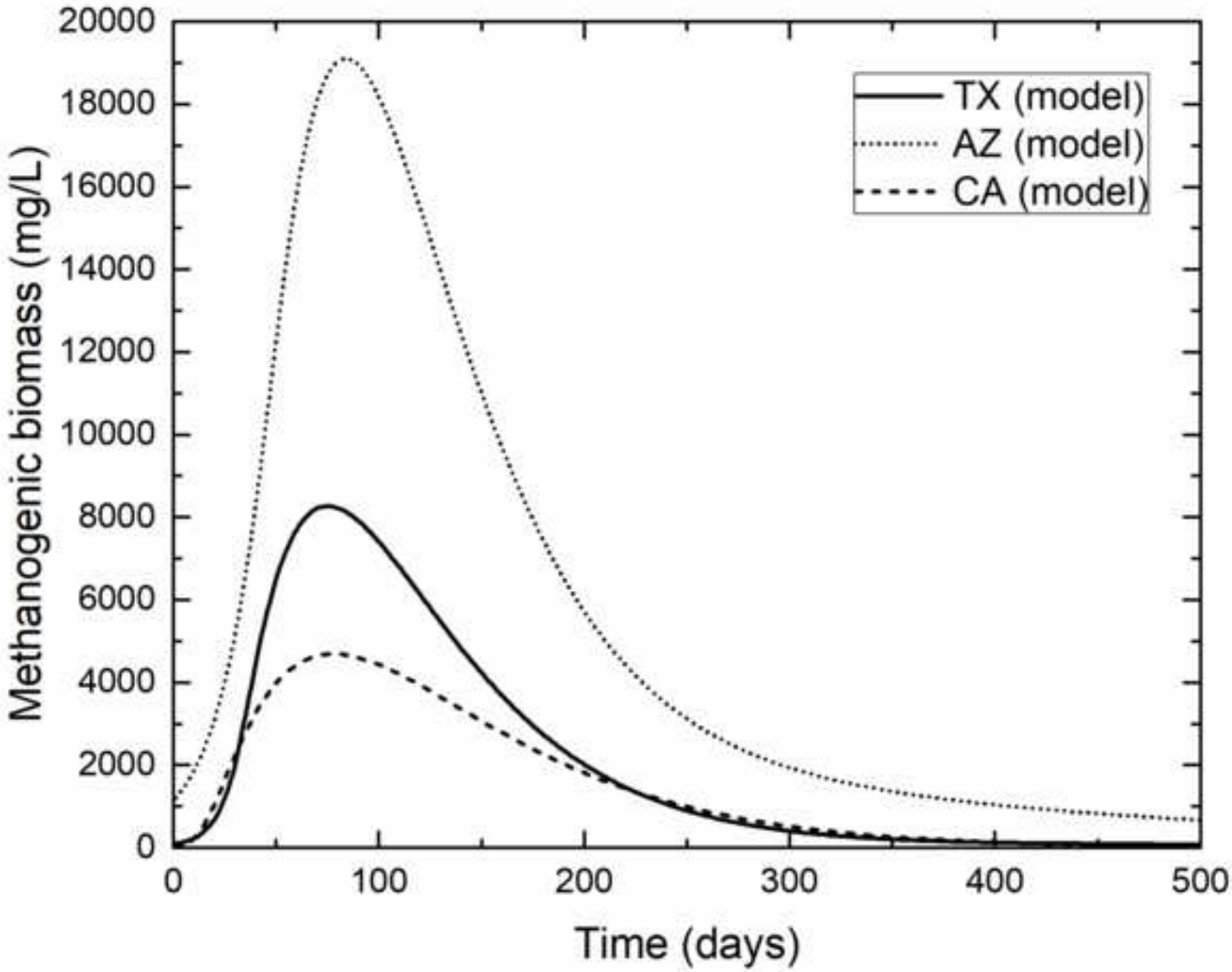

Figure 4

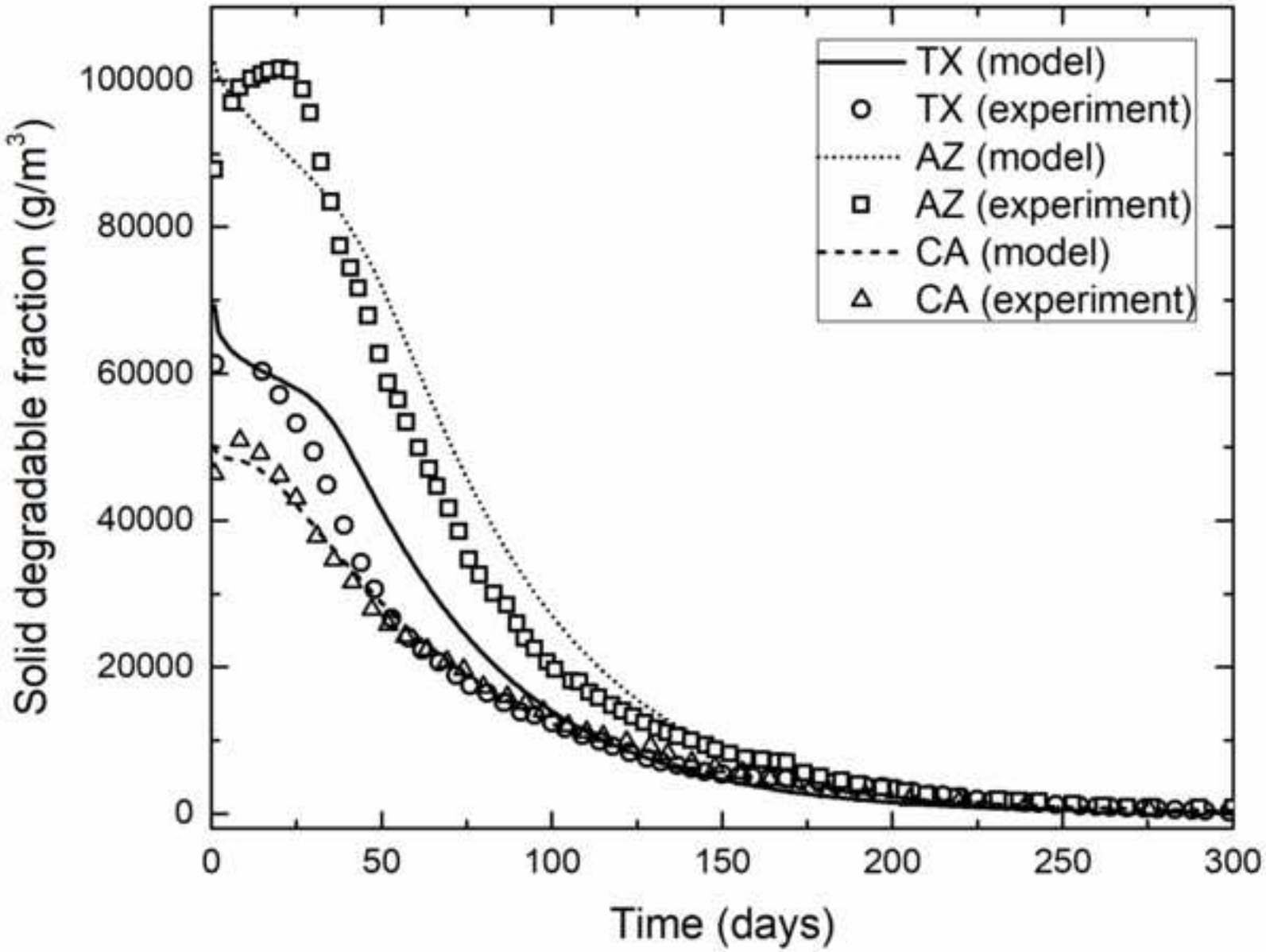

Figure 5

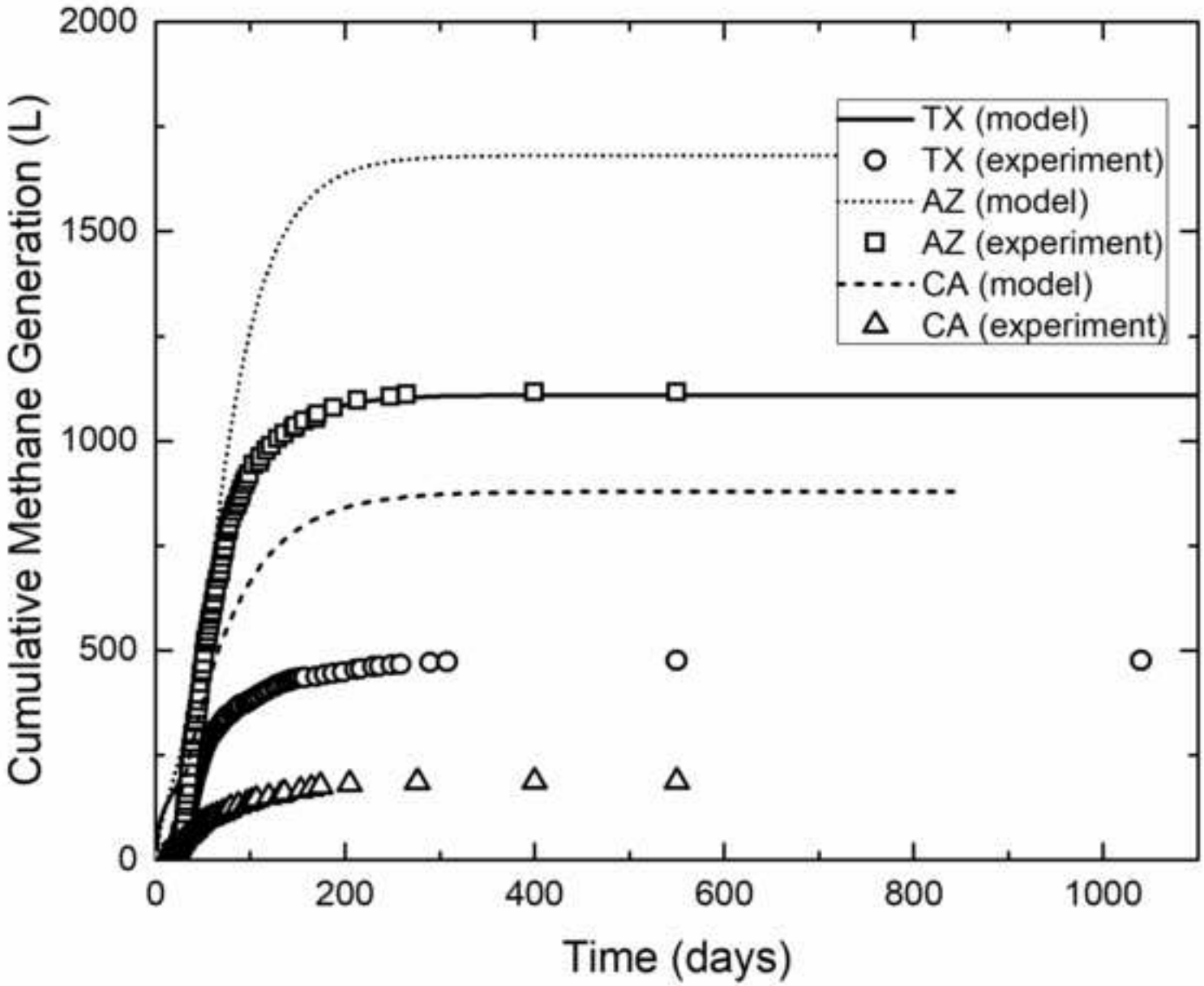

Figure 6

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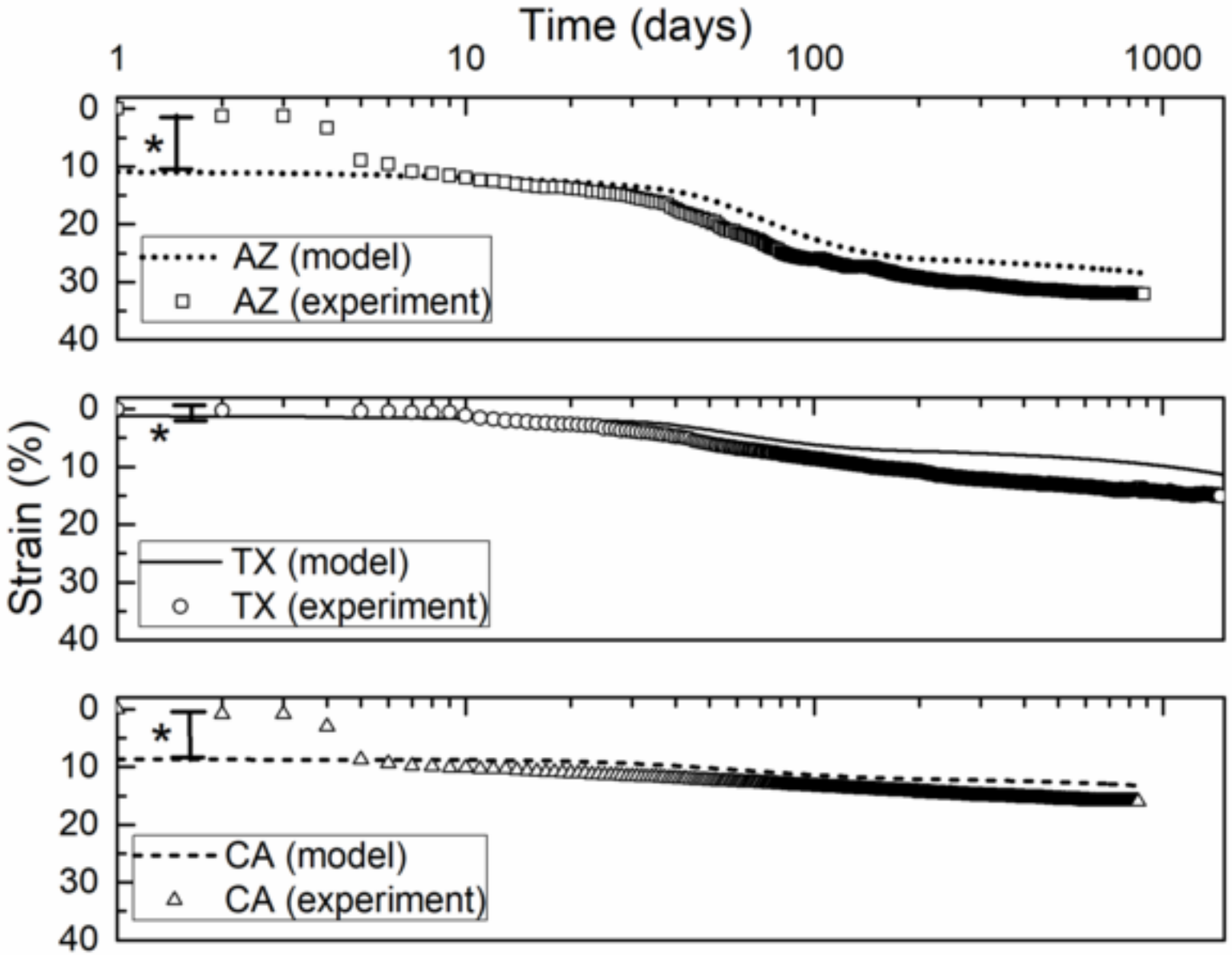

Table 1. Composition, moisture content, volatile solids and biodegradability of three MSW specimens.

Parameter	Value		
	AZ	TX	CA
Paper (%)	21.4	10.7	4.6
Wood (%)	2.1	4.3	5.8
Soft plastic (%)	8.5	5.9	3.9
<20 mm particles (%)	68.0	79.1	68.5
Hard plastic, metal, cobbles (%)	0	0	17.2
Initial total weight (kg)	21.69	29.31	29.02
Initial average moisture content (%)	32.7	37.7	28.1
Volatile solids in <20 mm fraction (g/g dry)	0.128	0.252	0.086
Initial percentage of biodegradable waste, B_0 (% dry)	30.1	16.2	10.5
Initial density of biodegradable waste, $\gamma_{B,I}$ (kg/m^3)	140	97	67

Table 2. Example calculation of solid degradable fraction from initial waste composition of TX specimen.

Waste constituent	Dry weight (kg)	Degradable solids (kg dry)	Inert solids (kg dry)	Density of solid constituent (kg/m^3)	Degradable phase density, ρ_{sd} (kg dry/m^3)	Inert phase density, ρ_{si} (kg dry/m^3)
Paper	2.3	1.5[a]	0.8	810[b]	1044[g]	1727[g]
Wood	0.9	0.1[d]	0.8	600[e]		
Soft Plastic	1.3	0	1.3	970[b]		
<20 mm	16.9	0.9[f]	15.9	2200[c]		
Total (% dry weight)		11.7	88.3			

[a]Degradable fraction in paper waste is considered to be 20-80 % (Barlaz et al. 1990). For TX specimen, it is 65 %
[b]Olivier and Gourc, 2006
[c]Specific gravity of soil-like fraction with organic content considered to be the average of the range 2.0-2.4. (Radforth et al., 1996)
[d]Degradable fraction in wood is considered to be 0-20 % (Milke et al., 2010). For TX specimen, it is 10 %.
[e]Zobel and van Buijtenen,1989
[f]Degradable fraction in <20 mm soil-like material has been assumed to be the organic fraction as determined by volatile solids content
[g]Phase density = $\frac{\sum Dry\ weight\ of\ each\ constituent\ in\ a\ phase}{\sum Volume\ of\ each\ constituent\ in\ the\ same\ phase}$

Table 3. Initial modelling conditions for the three MSW specimens.

Parameter	AZ	TX	CA
Volumetric moisture content, θ_E (%)	38	49	42
Solid degradable fraction (%)	24	12	9
Degradable phase density, γ_{sd} (kg/m^3)	955	1044	1338
Inert phase density, γ_{si} (kg/m^3)	1716	1727	1660

Table 4. HBM-Biodegradation calibrated model parameters for the three MSW specimens

Model parameters	Value		
	AZ	TX	CA
Initial VFA concentration, (g/m^3)	8500	0	0
Initial MB concentration, (g/m^3)	1200	100	10
Maximum hydrolysis rate (b), gVFA/m$^3_{aqueous}$/day	6500	3500	2700
Product inhibition factor (k_{VFA}), m^3/g	1.2x10^{-4}	4.2x10^{-4}	6.3x10^{-3}
Structural transformation parameter (n)	1.0	0.7	0.7
Maximum specific growth rate for cellulose (k_0), day^{-1}	0.07	0.15	0.75
Methanogen death rate (k_2), day^{-1}	0.004	0.0005	0.0005
Half saturation constant (k_{mc}), g/m^3	3500	1500	700
Cell/substrate yield coefficient (Y)	0.4	0.4	0.3
Biodegradation-induced void change parameter (Λ)	0.8	0.7	0.8
Creep viscosity coefficient (χ)	0.018	0.010	0.007
Reference time (t_{ref}), day	270	220	290

Author photographs

Author photographs

Author photographs

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