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# Assessment of percolation through a solid leach bed in dry batch anaerobic digestion processes

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## H I G H L I G H T S

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- Percolation through solid cow manure was assessed.
  - An experimental procedure was set up to calibrate a multiphase flow model.
  - A percolation and drainage cycle was needed to reach water holding capacity.
  - Micro/macro-porosity and permeability were obtained for two compaction levels.
  - Numerical simulations successfully reproduced experimental dynamic water retention.
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## A B S T R A C T

This work aimed at assessing water percolation through a solid cow manure leach bed in dry batch AD processes. A laboratory-scale percolation column and an experimental methodology were set up. Water behaviour was modelled by a double porosity medium approach. An experimental procedure was proposed to determine the main hydrodynamic parameters of the multiphase flow model: the porosity, the permeability and the term for water exchange from macro- to micro-porosity. Micro- and macro-porosity values ranged from 0.42 to 0.70 m<sup>3</sup> m<sup>-3</sup> and 0.18 to 0.50 m<sup>3</sup> m<sup>-3</sup>. Intrinsic permeability values for solid cow manure ranged from 5.55 · 10<sup>-11</sup> to 4.75 · 10<sup>-9</sup> m<sup>2</sup>. The term for water exchange was computed using a 2nd order model. The CFD tool developed was used to simulate successive percolation and drainage operations. These results will be used to design leachate recirculation strategies and predict biogas production in full-scale dry AD batch processes.

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### Keywords:

Dry batch anaerobic digestion  
Leachate percolation  
Water distribution  
CFD  
Multiphase flow model

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## 1. Introduction

Dry anaerobic digestion is an effective, sustainable method for the treatment of solid wastes from agriculture, allowing the recovery of both energy and nutrients. For small-size farms, more and more rustic easily-maintained dry anaerobic batch technologies are being developed to treat wastes such as solid cow manure. Nevertheless, these types of dry anaerobic batch reactors have

several drawbacks: long degradation times that lead to a large bio-reactor volume and hence a high investment cost; a low methane conversion yield; a high inoculation rate using stabilized digestate kept from previous batches (Schäfer et al., 2006); and high sensitivity to inhibition if the substrate is mostly composed of readily biodegradable organic matter and if the water content is low (Vavilin et al., 2008; Abbassi-Guendouz et al., 2012). Appropriate leachate percolation through the solid waste body (amount of recirculated leachate, leachate injection geometry, injection flow rate and recirculation frequency) is a key point to solve these major issues and to increase the overall process efficiency (Benbelkacem et al., 2010).

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Leachate helps to provide water, microorganisms, nutrients and to dilute intermediate anaerobic digestion products (mainly volatile fatty acids VFAs,  $\text{NH}_4^+$  and  $\text{H}_2$ ) and potential toxins (Bilgili et al., 2007; Vavilin et al., 2003; Xie et al., 2012). Water is a key component as it acts as a solvent and contributes to mass transfer, the diffusion of microorganisms and the colonization of the substrate reactive surface (Vavilin et al., 2003; Le Hyaric et al., 2011; Bollon et al., 2011). A minimum water content is necessary for biological activity (Lay et al., 1997; Pommier et al., 2007). In dry batch processes, the overall performance and reliability depend on the extent to which VFA accumulation can be controlled through their leaching using water recirculation (Kusch et al., 2011). The amount of free water determines the efficiency with which intermediate reaction products are extracted (mainly as VFAs). To better understand water distribution and quantify the optimal amount of leachate to be recirculated, it is necessary to study the physical properties of the solids involved.

Various laboratory methodologies to determine the physical properties of the solid medium have been reported for municipal solid waste samples (Stoltz et al., 2012, 2010a,b; Wu et al., 2012). Physical parameters such as water content, bulk density, porosity and permeability have been measured in static (Agnew and Leonard, 2003; Richard et al., 2004; Schaub-Szabo and Leonard, 1999) and dynamic (Huet et al., 2012) conditions. Agnew and Leonard (2003) have reported porosity values for solid cow manure ranging between 90% and 92% and solid particle dry densities between 1200 and 1800  $\text{kg m}^{-3}$ . These research works have focused on landfilling and composting, and thus used static characterization with different initial water contents. However, in dry batch AD processes, the water content has been shown to vary in time and space. Therefore, hydrodynamic characterization is necessary to better understand the process of percolation through the solid leach bed.

Olivier and Gourc (2007) have studied the hydrodynamic behaviour in municipal solid waste in a 1  $\text{m}^3$  pilot-scale compression cell with leachate recirculation and have investigated the correlation between biological degradation and settlement in a landfilling process with leachate recirculation. They have also studied the effect of the degradation degree and the leachate flow rate on the variations of the effective porosity and permeability of the solid waste. Moreover, modelling approaches have been proposed to improve our understanding of water distribution and physical changes during percolation processes. Tinet et al. (2011) and Capelo and De Castro (2007) considered municipal solid waste as a double porosity medium so as to take better account of water transport and distribution inside the waste. In the double porosity medium theory, two types of heterogeneities are distinguished: macro-porosity, where two-phase (gas and liquid) flow transport occurs, and micro-porosity, where both the gas and liquid phases are static. The comparison of numerical simulations with experiments should help to determine the main physical changes (such as porosity and permeability variations) during the waste treatment as a whole and thus to better understand the physical processes involved and finally to optimize biogas production.

The aim of this work was to study water distribution through a solid bed of cow manure during the percolation and drainage processes. Firstly, a double medium porosity was defined and static and dynamic water saturations were determined. From experimental data, the main hydrodynamic parameters, i.e. micro-porosity, macro-porosity and permeability were identified. A water balance on the solid bed was established to estimate micro-saturation/macro-saturation values and to quantify water transport from macro-porosity to micro-porosity. These results were used to implement a CFD modelling tool running numerical simulations to predict water distribution in full-scale percolation processes.

## 2. Methods

### 2.1. Solid waste characterization

Cow manure solid waste was taken from an experimental farm of the French National Polytechnic Institute located in the area of Toulouse (Midi-Pyrénées, France). Samples were rapidly placed at 4 °C. To measure the initial gravimetric water content, a representative total mass of 1 kg of each sample was shredded to 1 cm and dried at 105 °C for 24 h. For percolation operations, no mechanical pretreatment was carried out. To assess the repeatability of the results, two experiments ( $M_1$  and  $M_2$ ) were performed using the same solid cow manure.

As solid cow manure is a highly heterogeneous medium, a double-medium porosity model was used to represent the physical heterogeneity of the solid waste. This model is based on a theoretical assessment similar to those used by Tinet et al. (2011) and Capelo and De Castro (2007) for municipal solid waste. The distinction between micro-pores and macro-pores was based on the assumption that the micro-porosity was completely filled with static water when the solid bed reached its water holding capacity (Stoltz et al., 2010a; Huet et al., 2012).

The solid fraction ( $\varphi_s$ ) was defined as the ratio between the volume of dry mass and the total volume of the leach bed and was considered constant during the whole testing period. The different fractions satisfied the obvious relationship

$$\varphi_M + \varphi_s + \varphi_m = 1 \quad (1)$$

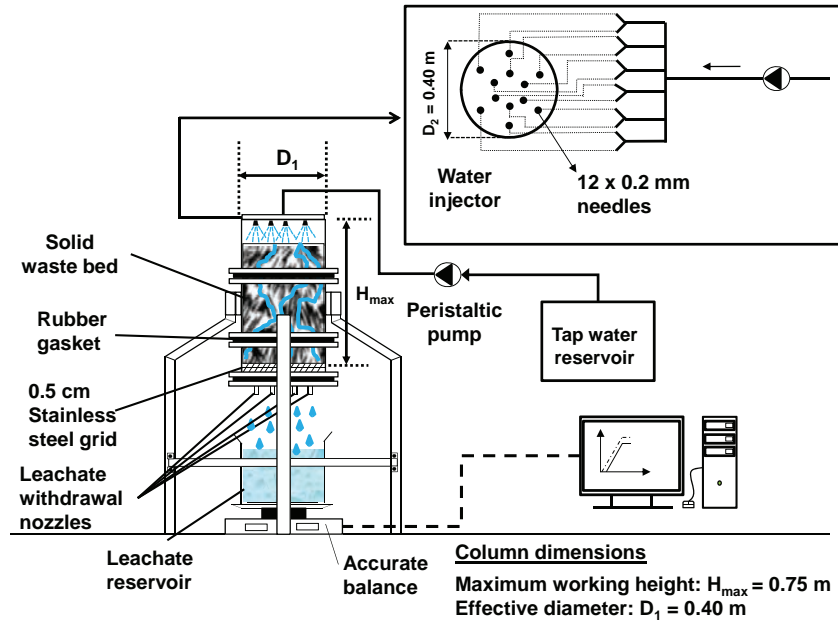
where  $\varphi_M$  and  $\varphi_m$  are the macro-porosity and micro-porosity fractions, respectively. The dry mass lost with the drained liquid (total amount of solubilized mineral matter and COD) made up less than 1% of the total dry mass fed into the percolation device, and was thus neglected. The initial characteristics of the solid waste beds are reported in Table 1.

### 2.2. Experimental set-up

The laboratory scale bed reactor consisted of a stainless steel column of 100 L total working volume, and 0.75 m maximum working height. The height was divided into three sections enclosed with rubber gaskets to ensure water tightness (Fig. 1). The water was injected by means of a peristaltic pump at the top of the vessel using an injector that divided the flow among twelve injection points distributed above the whole working surface. The applied surface hydraulic loads (SHLs) were 16, 48, 96, 128 and 176  $\text{L h}^{-1} \text{m}^{-2}$ , corresponding to injection flow rates of 2, 6, 12, 16 and 22  $\text{L h}^{-1}$ , respectively. To avoid side-effects at the vessel walls, a surface/volume ratio of 2.5 was chosen in agreement with Huet et al. (2012). Thus, 0.4 m of waste height was used. The leachate was recovered by five withdrawal nozzles as shown in Fig. 1 It fell into a reservoir placed on a precision balance for online acquisition of the drained leachate flow. To avoid migration of solid

**Table 1**  
Initial characteristics of leach beds.

Initial characteristics	$M_1$	$M_2$
Solid bed total wet mass (kg)	16.7	17.8
Solid bed total dry mass (kg)	6.60	7.50
Bulk density ( $\text{kg m}^{-3}$ )	313	337
Total height (m)	0.40	0.40
Total volume (L)	50.27	50.27
Initial gravimetric water content ( $\text{kgH}_2\text{O kgW M}^{-1}$ )	0.58	0.58
Initial static water saturation ( $\text{m}^3 \text{m}^{-3}$ )	0.18	0.20
Initial solid fraction $\varphi_s$ ( $\text{m}^3 \text{m}^{-3}$ )	0.08	0.08
Initial total porosity ( $\varphi_m + \varphi_M$ ) ( $\text{m}^3 \text{m}^{-3}$ )	0.92	0.92



**Fig. 1.** Schematic diagram of the leachate percolation bed used to characterize the water behaviour. At right, zoomed top view of the water injector (12 needles of diameter 0.2 mm set on a plastic disc of diameter 0.40 m).

particles with the leachate, a 0.5 cm stainless steel grid was placed at the vessel bottom.

The head of the vessel was not sealed and allowed air to flow through the porous medium. Aerobic biological activity was neglected because of the short duration of the experiment. Water evaporation from the solid waste bed was monitored throughout the test. Total water evaporation represented less than 0.2% of the injected water for the lowest SHL ( $16 \text{ L h}^{-1} \text{ m}^{-2}$ ).

Once the device had been filled with the waste, a round plate (diameter 0.38 m) was placed at the waste surface to avoid any vertical movement of the bed due to flotation.

### 2.3. Percolation and drainage cycle

The experiments were performed at ambient temperature ( $20^\circ\text{C}$ ). Firstly, water was injected from the vessel bottom at a moderate SHL ( $96 \text{ L h}^{-1} \text{ m}^{-2}$ ) until the solid waste bed was completely filled with water. The water flow was then stopped and the system was maintained closed for two hours to make this first wetting process more efficient. Then, the withdrawal nozzles were opened, allowing the leachate to drain out.

When the first water saturation and drainage had been completed, constant SHLs were applied to the vessel following this order: 16, 176, 48, 128 and  $96 \text{ L h}^{-1} \text{ m}^{-2}$ . This sequence was chosen in order to alternate low and high SHL, thus avoiding a possible dependence of the waste wetting on the SHL increase. The cumulated injected water mass and the cumulated drained leachate mass were continuously monitored and plotted versus time (Fig. 2). The difference between the two curves represented the injected water present in the solid bed. Fig. 2 illustrates a complete percolation/drainage operation at a given SHL with four different stages: stage 1, water first accumulates inside the bed and no water leaves it; stage 2, the liquid begins to flow out of the bed and the inlet flow rate becomes equal to the outlet flow rate, leading to a first plateau on the curve representing the total water retention (dynamic and static water retention); stage 3, as soon as the influent flow is stopped the outlet flow decreases until the end of the drainage; stage 4, when all the mobile water has drained out, a constant value of the retained water was obtained (second

plateau in Fig. 2). For all experiments, the amount of water injected was not totally recovered in the leachate. This means that a fraction of the injected water was retained inside the micro-porosity. The successive values of the retained water volume allowed micro-saturation ( $S_m$ ) values to be calculated throughout the experiment.

### 2.4. Mathematical model and measurement of hydrodynamic parameters

#### 2.4.1. Micro-saturation $S_m$ and the term for water exchange ( $q_{L,M \rightarrow m}$ )

The two-phase flow (gas and liquid) in the porous medium (solid cow manure) was assumed incompressible and isothermal. In these conditions, the mass conservation equations can be written as (Aziz and Settari, 1979; Gerritsen and Durlofsky, 2005; Horgue et al., 2014):

$$\nabla \cdot U_L + \nabla \cdot U_G = -q_{L,M \rightarrow m} \quad (2)$$

$$\varphi_m \frac{\partial S_m}{\partial t} + \nabla \cdot U_L = -q_{L,M \rightarrow m} \quad (3)$$

$$\varphi_m \frac{\partial S_m}{\partial t} = -q_{L,M \rightarrow m} \quad (4)$$

where  $U_G$  and  $U_L$  are the gas and the liquid velocities, respectively, in the macro-pores and  $q_{L,M \rightarrow m}$  is the term for water exchange between the macro- and micro-porosity (Köhne et al., 2004; Huang et al., 2004; Di Donato et al., 2003).

The water holding capacity (WHC) corresponded to the situation where the micro-porosity was completely filled with water and therefore the micro-saturation ( $S_m$ ) value was 1. However, obtaining a realistic value of WHC was not trivial because complete filling of the micro-porosity of the waste required several days of soaking. A water saturation of the waste by simply submerging the waste and then draining was not sufficient to reach the water holding capacity. For that reason, the value of WHC (and so the micro-porosity  $\varphi_m$ ) was deduced from the water retention measured at the end of a complete experimental cycle, when successive water retention measurements over several days showed

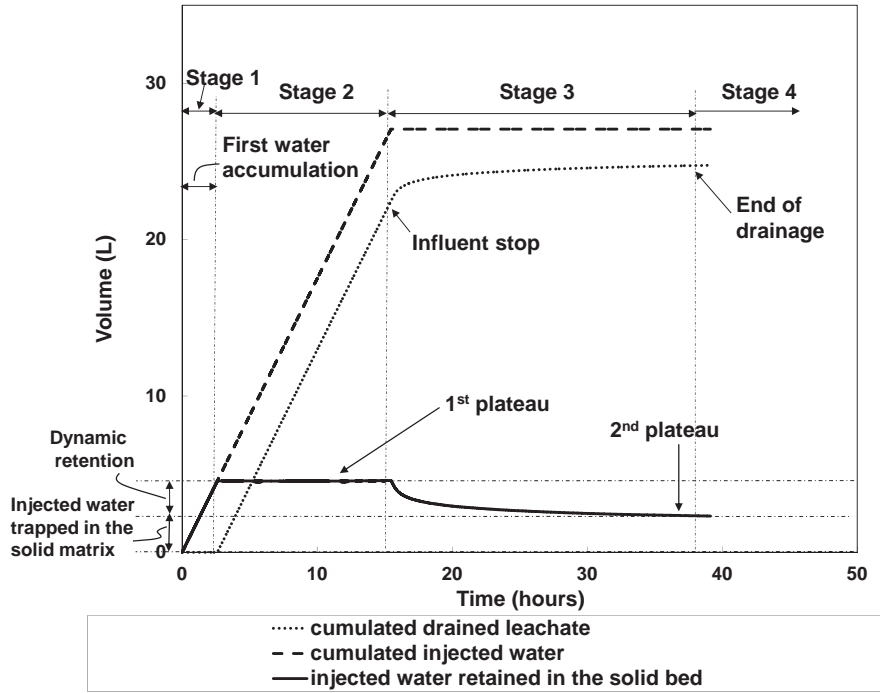


Fig. 2. A typical percolation and drainage curve for a surface hydraulic load (SHL) of  $96 \text{ L h}^{-1} \text{ m}^{-2}$ .

that a steady state value had been reached. This value was then used to compute the initial waste micro-saturation ( $S_m$ ).

The successive values of water retention obtained during the experimental process can be used to define the micro-porosity filling rate, i.e. the value of the term for water exchange  $q_{L,M \rightarrow m}$ . A second order model between  $q_{L,M \rightarrow m}$  and micro-saturation ( $S_m$ ) simulating the transient transfer of water from the macro-porosity to the micro-porosity is proposed in Section 3.1.

#### 2.4.2. Macro-saturation $S_M$ and apparent permeability $K_i$

The macro-saturation ( $S_M$ ) values were computed as the ratio between the dynamic water retention volume (difference between the 1st and 2nd plateaux in Fig. 2) and the volume of the macropores.

Considering a multiphase flow in a porous medium, the difference between gas and liquid pressure in a control volume is generally defined as the macro-scale capillary pressure  $P_c$  (Leverett, 1940), and depends on the macro-saturation ( $S_M$ ):

$$P_c(S_M) = P_G - P_L \quad (5)$$

Phase velocities can then be expressed as a function of gas pressure ( $P_G$ ) and liquid macro-saturation ( $S_M$ ) using the generalized Darcy's law (Aziz and Settari, 1979; Gerritsen and Durlafsky, 2005; Horgue et al., 2014; Muskat, 1949):

$$U_G = -\frac{\mathbf{K}k_{r,G}(S_M)}{\mu_G}(\nabla P_G - \rho_G \mathbf{g}) \quad (6)$$

$$U_L = -\frac{\mathbf{K}k_{r,L}(S_M)}{\mu_L}(\nabla P_G - \rho_L \mathbf{g} - \nabla P_c(S_M)) \quad (7)$$

where  $\mathbf{K}$  is the intrinsic permeability tensor of the porous medium,  $k_{i,L}(S_M)$  is the relative permeability of phase  $i$  depending on the macro-saturation  $S_M$ , and  $\rho_i$  and  $\mu_i$  are the density and the viscosity, respectively, of phase  $i$ . In the experimental set-up presented in Section 2.2, we assumed that the liquid phase was uniformly distributed on the top of the waste and that the saturation gradient inside the waste was thus negligible, as was the capillary pressure

gradient. There was no gas injection inside the device and both sides (top and bottom) were open to the ambient air, i.e. at atmospheric pressure. Consequently, the gas flow was almost zero and the hydrostatic pressure gradient  $\nabla P_G$  was of the order of  $\rho_G \mathbf{g}$  and therefore negligible relative to the contribution of the liquid pressure ( $\rho_L \mathbf{g}$ ). The liquid phase velocity along the vertical ( $z$ ) axis in the solid waste could then be simplified as

$$U_{L,z} = \frac{K_z \cdot k_{r,L}(S_M)}{\mu_L} \rho_L \quad (8)$$

where  $K_z$  ( $\text{m}^2$ ) is the absolute permeability along the  $z$ -axis. The apparent permeability  $K_i$  of a phase  $i$  for a given porous medium was thus defined as the product of the intrinsic and relative permeabilities (Horgue et al., 2014):

$$K_i(S_i) = K_z \cdot k_{r,i}(S_i) \quad (9)$$

The relative permeability took a value between 0 and 1 and mimicked the fact that the presence of another fluid phase reduced the apparent permeability for a given phase. For a given flow rate inside the solid waste, the liquid apparent permeability could be deduced directly from

$$K_L(S_M) = \frac{\mu_L Q_L}{\rho_L g A_{exp}} \quad (10)$$

where  $A_{exp}$  is the section ( $\text{m}^2$ ) of the percolation column presented in Section 2.2.

#### 2.4.3. Solid bed compaction

Once a complete experimental cycle had been performed, a surface load of  $320 \text{ kg m}^{-2}$  (3 kPa) was placed above on the surface of the solid bed (a 30 cm-diameter-disc weighing 40 kg). This load simulated a height of 66 cm of solid cow manure exerting a pressure on the solid leach bed. Hence a new total volume of solid bed was determined by measuring the new solid bed height and its new water content. Thereafter, the solid volume was considered to be constant and not affected by compaction. Using this assumption, a new complete percolation and drainage cycle was per-

formed. The new micro/macro-porosities and saturation values were computed following the same approach.

### 3. Results and discussion

#### 3.1. Micro-saturation $S_m$ and the term for water exchange $q_{L,M \rightarrow m}$

The evolutions of micro-saturations were computed for the two solid beds tested. The values presented in Fig. 3A are the averages obtained over the two solid beds  $M_1$  and  $M_2$ . At the end of the first water saturation/drainage process, the solid bed had decreased from 50.27 L to 47.75 L (named settlement 1 in Fig. 3A). During the experiment, the solid mass held a certain amount of water by capillary forces. The structure was affected by water suction leading to a settling of the waste body. Afterwards, no further settling was observed during successive percolation and drainage experiments and the physical structure of the porous medium remained stable. The micro-porosity was filled progressively until it was completely full ( $S_m = 1$ ). Then, micro-porosity ( $\phi_m$ ) values were assessed as 0.406 for solid bed  $M_1$  and 0.426 for solid bed  $M_2$ .

From the micro-saturation ( $S_m$ ) values obtained experimentally, the term for water exchange between macro-porosity and micro-porosity was assessed. Experimental values of  $q_{L,M \rightarrow m}$  were computed as the difference between the initial and final

micro-saturation of each percolation/drainage operation (see Eq. (4)). In Fig. 3B, the term  $q_{L,M \rightarrow m}$  is plotted as a function of micro-saturation ( $S_m$ ). As can be observed, it is extremely difficult to establish a correlation, mainly because the term for water exchange varies strongly when the value of  $S_m$  is close to 1.

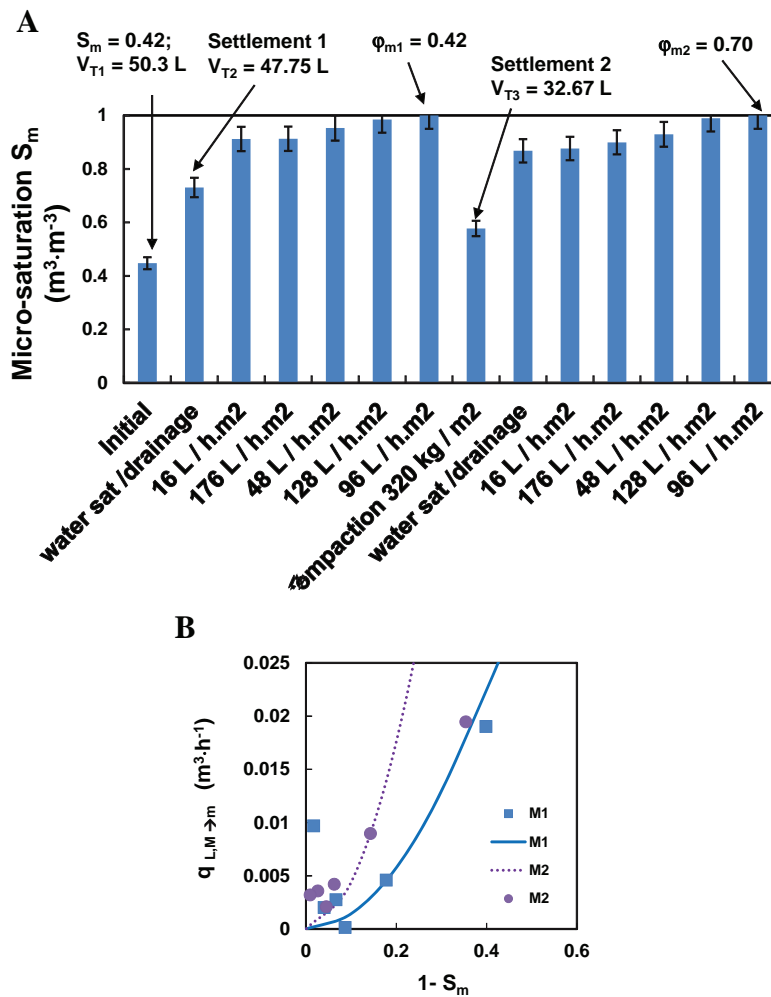
For the modelling approach, first-order and second-order terms were tested in different configurations and media, as well as with different hydraulic properties. The second-order term, originally derived for water transfer between fractures and the matrix in fractured materials, was adapted for a very different saturated solid medium. We used the second-order model as suggested by Köhne et al. (2004):

$$q_{L,M \rightarrow m} = C(1 - S_m)^2 \quad (11)$$

where coefficient  $C$  was computed to fit the experimental value of  $q_{L,M \rightarrow m}$  for the first percolation and drainage cycle. The kinetics constants had the values of 0.144 and 0.44 for solid beds  $M_1$  and  $M_2$ , respectively.

#### 3.2. Macro-porosity saturation $S_M$ and apparent permeability $K_L$

The macro-porosity ( $\phi_M$ ) had values of 0.514 and 0.494 for solid wastes  $M_1$  and  $M_2$ , respectively. The macro-saturation ( $S_M$ ) values



**Fig. 3.** (A) Micro-saturation evolution with percolation and drainage cycles. Values are averages over both solid beds,  $M_1$  and  $M_2$ . (B) Correlation between  $1 - S_m$  and the term for water exchange  $q_{L,M \rightarrow m}$ . Points are values from experimental data for both non-compacted solid beds,  $M_1$  (squares) and  $M_2$  (circles). Curves are the modelled data for  $M_1$  (continuous line) and  $M_2$  (dotted line).

were computed and are plotted versus the surface hydraulic load (SHL) in Fig. 4A.

The macro-saturation ( $S_M$ ) values for each SHL were averaged for both solid beds,  $M_1$  and  $M_2$ . Non-compacted and compacted conditions were studied. Similar relationships were observed between compacted and non-compacted solid beds. Between SHLs of 0 and  $48 \text{ L h}^{-1} \text{ m}^{-2}$ , macro-saturation increased strongly. From 48 to  $176 \text{ L h}^{-1} \text{ m}^{-2}$ , macro-saturation increased slowly with increasing SHL, reaching an almost a stable plateau for high SHL. Maximum macro-saturation values for the non-compacted solid bed and for the compacted bed were  $15.67 \pm 5\%$  and  $51.33 \pm 5\%$ , respectively. This behaviour is commonly observed in multiphase flow in porous media and can be easily modelled by a classical approach. Fig. 4B shows the apparent permeability ( $K_L$ ) curves plotted as a function of macro-saturation ( $S_M$ ). A law widely applied to multiphase flow in porous media was used to fit the experimental data (Brooks and Corey, 1964):

$$K_L = K_z S_M^p \quad (12)$$

where  $K_z$  is the z-axis intrinsic permeability of solid waste ( $\text{m}^2$ ) and  $p$  is an empirical model parameter varying between 0.1 and 10. The identification of these two parameters was carried out. The values for intrinsic permeability ( $K_z$ ) and  $p$  were  $4.75 \cdot 10^{-9} \text{ m}^2$  and 3.62, respectively. As shown in Fig. 4B, simulated and experimental data values of apparent permeability ( $K_L$ ) were in good agreement.

The results presented confirmed that, if leachate was homogeneously injected over the whole surface of the solid leach bed, a multiphase porous medium approach could correctly fit the dynamic water retention for a given SHL in solid cow manure. This is essential for quantifying the dynamic water content in the system during steady state leachate recirculation mode, as this free water controls the transport of intermediate reaction metabolites and overall process efficiency.

### 3.3. Impact of compaction

The effect of compaction was studied by applying a surface load of  $320 \text{ kg m}^{-2}$  (3 kPa) and performing a new complete cycle. The solid bed decreased by 30% of its initial total volume and 1 L of leachate was drained during the settlement. As presented in Fig. 3A, during the two successive percolation and drainage cycles, micro-saturation ( $S_m$ ) increased until it reached a stable value. From these experimental data, solid, micro-porosity and macro-porosity fractions were computed. All the results are summarized in Table 2. The new micro-porosity ( $\varphi_m$ ) had value of 0.70 and

**Table 2**

Volumes of solid bed including porosity ( $V_T$ ), solids ( $V_s$ ), micro-pores ( $V_m$ ) and macro-pores ( $V_M$ ) before and after compaction for solid bed  $M_1$ . In brackets, solid fraction, micro-porosity and macro-porosity values (in  $\text{m}^3 \text{ m}^{-3}$ ) deduced from experimental measurements.

	Solid bed $M_1$	Solid bed $M_1$ compacted
Total volume $V_T$ (L)	47.75	32.67
Solid volume $V_s$ (L)	3.82 (0.08)	3.82 (0.12)
Micro-pores volume $V_m$ (L)	19.48 (0.41)	22.86 (0.70)
Macro-pores volume $V_M$ (L)	24.35 (0.51)	5.88 (0.18)

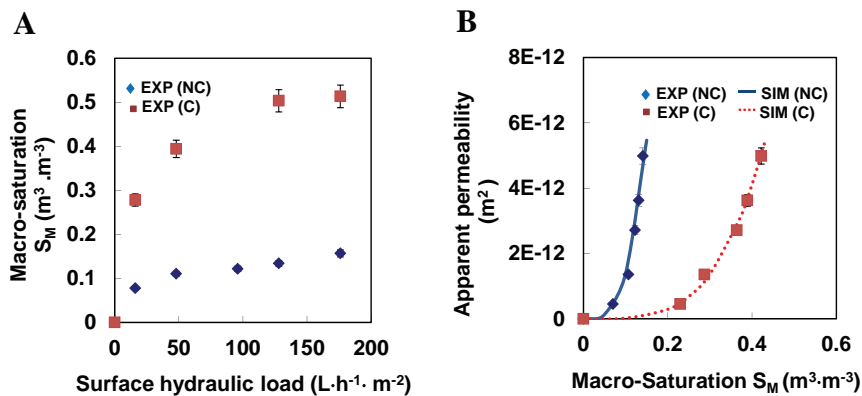
the new solid fraction ( $\varphi_s$ ) was 0.12. Finally, the new macro-porosity ( $\varphi_M$ ) was deduced as 0.18.

After compaction, the macro-porosity volume decreased by 18.47 L (75.80% of its initial volume). The micro-porosity volume increased by 3.38 L (17.35% of its initial volume). The total volume of the solid bed decreased ( $\Delta V_T$ ) by 15.08 L. This difference was lower than the macro-porosity volume variation (18.47 L). A possible explanation could be that after compaction, both the capillary pressure and the water retention capacity of the solid waste increased. Thus, in agreement with the definition of micro-porosity proposed in Section 2.1, the fraction of macro-pores lost was transformed into an additional volume of micro-pores. Subtracting micro-porosity volume gain (3.38 L) from the macro-porosity loss (18.47 L), gave 15.08 L, which corresponded to the total volume loss measured for the solid bed (15.08 L).

Macro-saturation values for the same range of SHL were higher for the compacted waste (see Fig. 4A) and reached a value of  $51.33 \pm 5\%$ . This was due to the decrease of both macro-porosity and apparent permeability values, which let the same amount of water pass through a much smaller volume. Using the Brooks & Corey law, experimental data were correlated and gave an intrinsic permeability ( $K_z$ ) of  $5.55 \cdot 10^{-11} \text{ m}^2$ . The model fitting parameter ( $p$ ) had a value of 3.77.

### 3.4. Numerical simulations

Considering that water injection was homogeneous at the top of the experiment, we neglected changing gradients perpendicular to the flow (pressure and saturation) and consequently, the experiments could be simulated using a 1D multiphase flow model. The system of Eqs. (2–4) was solved using the IMPES (Implicit Pressure and Explicit Saturation) algorithm. The code was developed in the open-source CFD platform OpenFOAM® which has already



**Fig. 4.** (A) Surface hydraulic load (SHL) correlation with macro-saturation ( $S_M$ ). Each point (experimental data) is an average value over both solid beds,  $M_1$  and  $M_2$ , for non-compacted (NC) and compacted (C) conditions. (B) Correlation between macro-saturation ( $S_M$ ) and apparent permeability ( $K_L$ ). Each point (experimental data) is an average over both solid beds,  $M_1$  and  $M_2$ , for non-compacted (NC) and compacted (C) conditions. Curves (SIM) are modelled with a Brooks & Corey law for non-compacted bed (continuous line) and compacted bed (dotted line).



been used for other two-phase flows (Horgue et al., 2014; Sheldon and Cardwell, 1959; Soullaine et al., 2014).

The waste height (40 cm) was discretized using 96 computational cells (each  $\Delta x$  of 4 mm). A fixed pressure was imposed on the bottom and top of the waste. The boundary conditions alternated between fixed liquid velocity (injection) and fixed zero-liquid macro-saturation (drainage) according to the same procedure as the experimental one. By integrating macro-saturation and micro-saturation terms over the whole domain, it was possible to directly compare the total liquid holdup for numerical simulations and experimental measurements. Fig. 5 shows a comparison between numerical simulations and experimental data for the solid beds  $M_1$  (Fig. 5A) and  $M_2$  (Fig. 5B). Initial values of liquid holdup for the comparison were set to 0.

There was good overall agreement between simulations and experiments. Nevertheless, two main differences were observed. On the one hand, the wetting process of solid waste in experiments was faster than in simulations. On the other hand, the draining process was faster in numerical simulations. This could be explained by the term for water exchange, which did not take all the physical features involved in the wetting process into account. Actually, the term for water exchange in double porosity theory is generally defined in terms of capillary pressure difference between the macro-porosity and the micro-porosity as reported in Huang et al. (2004) and Di Donato et al. (2003). It would therefore be nec-

essary to describe the capillary pressure curves accurately for the macro-porosity and micro-porosity domains. This would require other experimental determinations that are not within the scope of this paper, even though some improvements are possible for this model. However, as observed in Fig. 5 the simple model proposed here for the assessment of water exchange allows the transient solid waste wetting processes to be correctly described.

#### 4. Conclusions

A double porosity medium approach was proposed to model water distribution through a solid leach bed during percolation in dry batch AD processes. A lab-scale experimental procedure was set up to obtain the main hydrodynamic parameters: micro-porosity, macro-porosity, apparent permeability and the term for water exchange from macro- to micro-porosity. Numerical simulations successfully matched experimental dynamic water retentions. Capillary effects would need to be studied in greater depth to improve the model prediction capability. The CFD tool combined with a simplified biological model should allow leachate recirculation strategies to be evaluated for the optimization of full-scale processes.

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#### Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.biortech.2014.10.017>.

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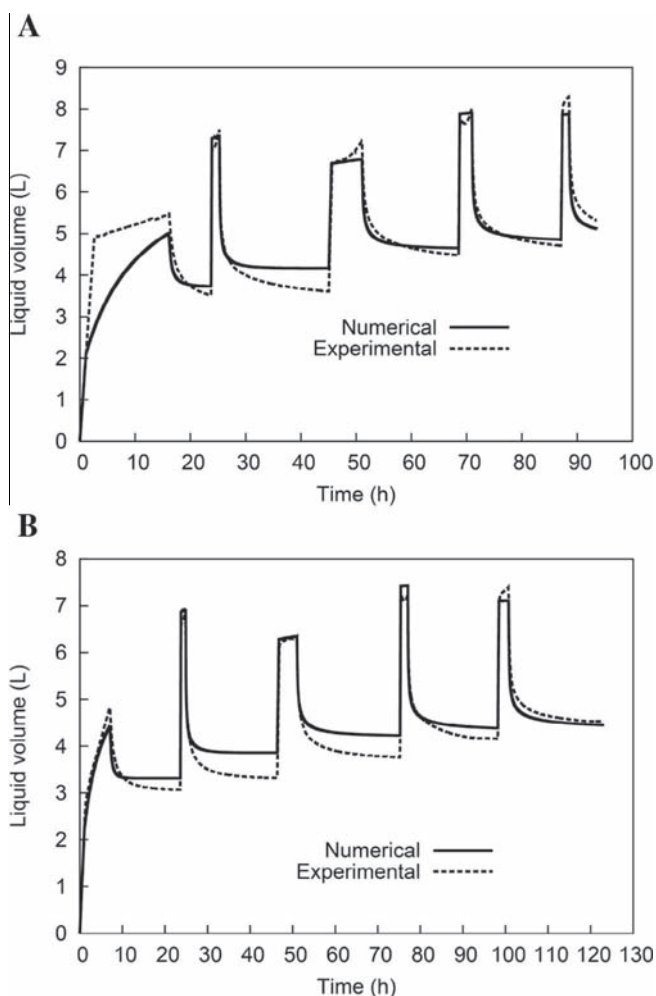


Fig. 5. Evolution of the liquid holdup (L) for solid beds  $M_1$  (A) and  $M_2$  (B) during percolation and drainage cycle. Dotted lines represent experimental data and continuous lines are for simulations.

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