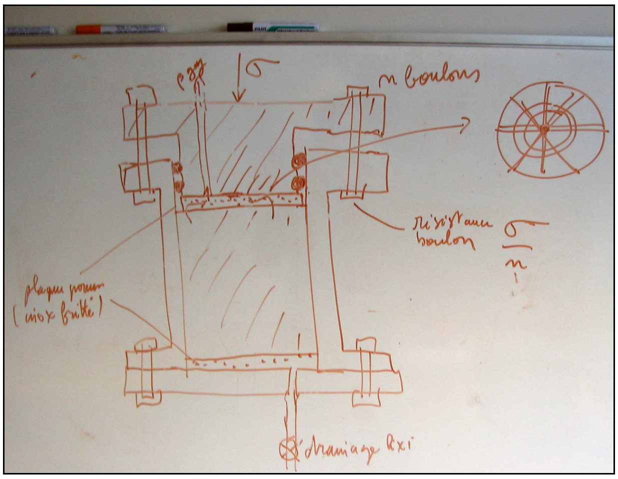
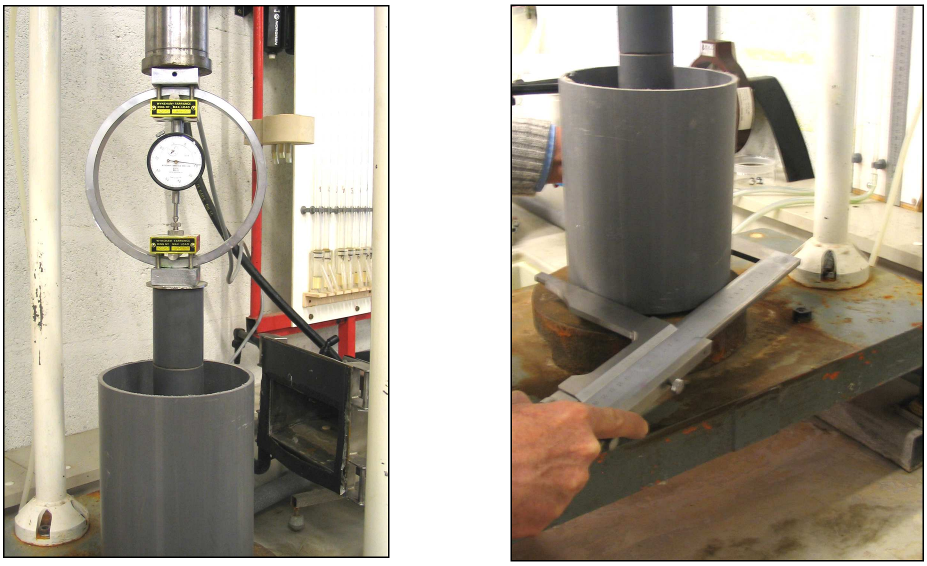
Annexe A

I Photos des expérimentations en laboratoire

1. Idée initiale et essais mécaniques



Idée initiale des cellules expérimentales



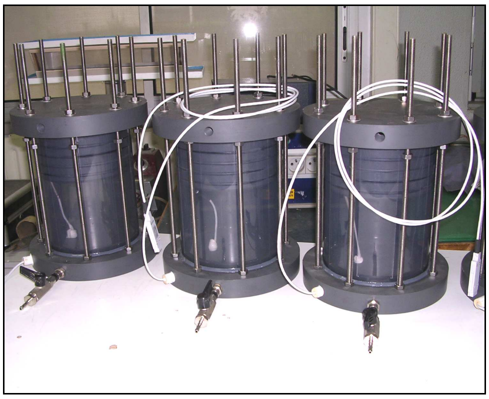
Essais mécaniques de résistivité des cellules en PVC avec application de la charge verticale pour compactage des déchets

2. Préparation des fractions de déchets selon l'Ademe 1993

25kg de déchets ont été coupés en petits morceaux de 1-2cm.

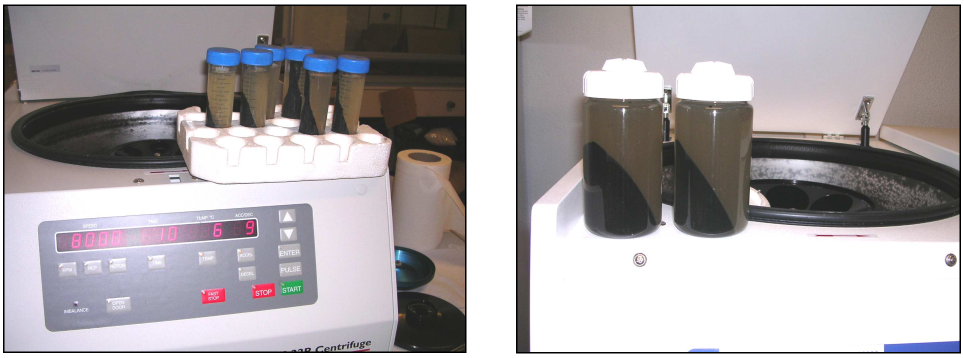
Découpage de différentes fractions de déchets (métaux, verre, tissue, papier, bois, plastique et déchet de cuisine) et préparation de 13 sacs poubelles de déchets avec les différentes fractions de l'Ademe (1993)

3. Préparation des cellules expérimentales



Préparation des cellules; installation des sondes de température dans les cellules et des joints isolants sur les couvercles

4. Préparation de la boue (sortie d'un digesteur méthanogène) pour inoculer les déchets

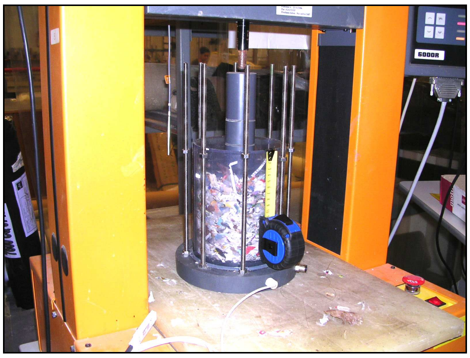


Centrifugation des boues liquides pour séparer l'eau de la partie solide, afin de ne par trop modifier les caractéristiques du lixiviat utilisé pour saturer les déchets

Liquéfaction des boues en les mélangeant avec 100ml de lixiviat

Inoculation des déchets avec le mélange de boue et de lixiviat

5. Remplissage, compactage et saturation des cellules




Remplissage des cellules et application de la charge verticale pour compacter les déchets

Injection de lixiviat avec la pompe

Application de la charge avec le couvercle

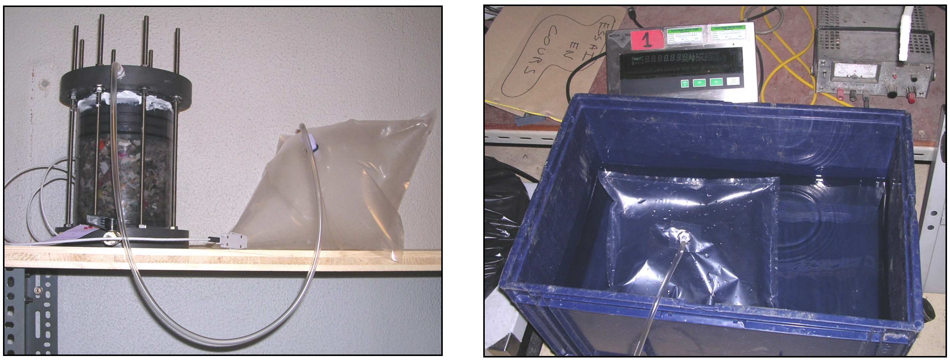
Drainage de lixiviat pendant un jour

Lixiviat sorti dans la poche de biogaz à cause de la pression de biogaz

Transport des pilotes dans la chambre à $33 \pm 2\,°C$

6. Production et quantification de biogaz

Les pilotes expérimentaux dans la chambre à $33 \pm 2\,°C$



Production de biogaz et quantification du volume de biogaz produit dans un bassin d'eau (par poussée d'Archimède)

Connexion des poches de biogaz au Micro-GC pour définir la composition du biogaz

7. Mesure de la température

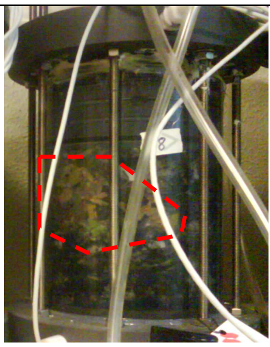
Mesure de la température des déchets à partir de capteurs de température

8. Echantillonnage de lixiviat pour les analyses biochimiques et microbiologiques

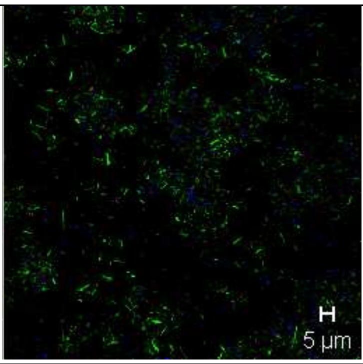
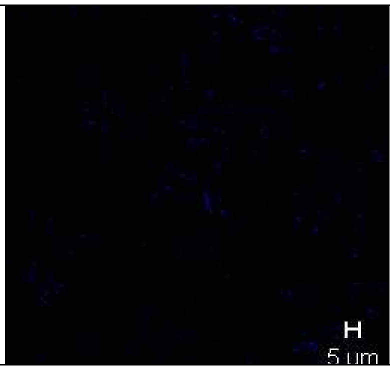
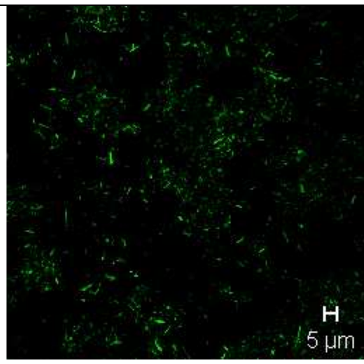
Echantillonnage du lixiviat

Echantillons de lixiviat de différentes couleurs

9. Zones dégradées et intactes dans les déchets

 Pilote 8	 Pilote 11	 Pilote 5

10. Résultats de l'analyse microbiologique du pilote 3

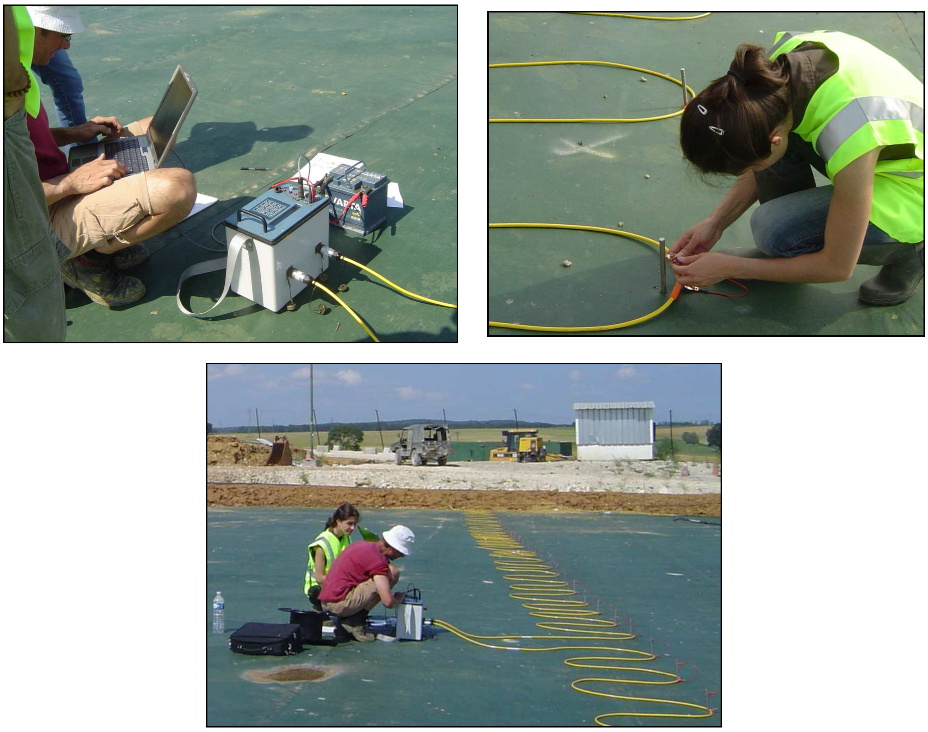
Ensemble de la population microbienne	Archae méthanogènes	Bactéries fermentatives
		

II Photo des expérimentations in situ

Centre d'ISDND-bioactive en France (pilote témoin et pilote test sont séparés par une couche d'argile)

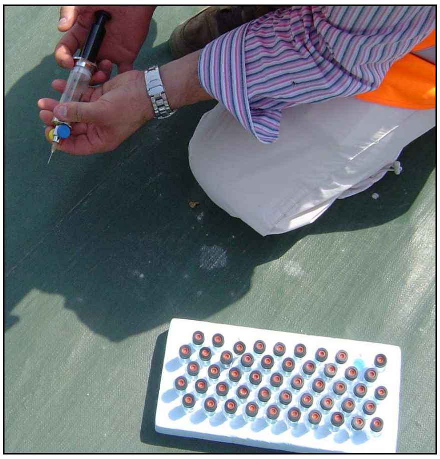
Stockage et compactage des déchets dans un pilote ouvert

Les couches de déchets stockés et de couverture d'argile



Installation des panneaux électriques sur le site pour mesurer la résistivité électrique

Système d'injection de lixiviat et de captage du biogaz sur site (à gauche), bassin de rétention du lixiviat et débitmètre (à droite)



Récupération de biogaz sur site (sous la couverture temporaire) et par les brins de biogaz pour faire des analyses de biogaz

Articles

Article de revu

Gholamifard S., Eymard R. and Duquennoi C. (2008), Modeling anaerobic bioreactor landfills in methanogenic phase: long term and short term behaviors, Water Research (2008), DOI: 10.1016/j. watres. 2008.09.040.

Articles de colloque

Gholamifard S., Eymard R. and Duquennoi C. (2008), Thermal Behavior of a Bioreactor Landfill During Leachate Injection, Global Waste Management Symposium & ICLRS, 2-7 September, Denver, Colorado, USA.

Gholamifard S., Eymard R. and Duquennoi C. (2008), Modeling Thermal and Biological Behavior of Bioreactor Landfills in Stationary Conditions Before Leachate Injection, The 23rd International Conference on Solid Waste Technology and Management, Philadelphia, PA, USA.

Gholamifard S., Eymard R. and Duquennoi C. (2008), Modeling Thermal Behavior of Bioreactor Landfills Before Leachate Recirculation, 5th International Symposium on Finite Volumes for Complex Applications (FVCA5), Aussois, France.

Gholamifard S., Duquennoi C. and Eymard R. (2007), Simulation of Two-Phase Flow in Anaerobic Bioreactor Landfills, Sardinia symposium, S.Margherita di Pula, Cagliari, Italy.

Gholamifard S., Duquennoi C. and Eymard R. (2007), A Multiphase Model of Bioreactor Landfill With Heat And Gas Generation And Transfer, Eurotherm Seminar N° 81, Reactive Heat Transfer in Porous Media, Ecole des Mines d'Albi, France.

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Modeling anaerobic bioreactor landfills in methanogenic phase: Long term and short term behaviors

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ABSTRACT

We have developed a mathematical model to simulate the behavior of real bioreactor landfills in the anaerobic methanogenic phase. This coupled model is composed of a two-phase flow and a biological model based on Darcy's law and Monod's model, respectively. This model considers bacterial activity and biological behavior as a function of temperature and makes it possible to study the thermo-biological behavior of bioreactor landfills with temperature changes. In this model we consider different effects of saturation on solid waste degradation. These effects consist of increasing hydrolysis with saturation and also decreasing the concentration of volatile fatty acids (VFAs) and activating the methanogenic biomass. This paper presents first the mathematical coupled model and the numerical methods used to solve the conservation equations. The numerical model is then used to simulate two bioreactor landfills. This paper presents the results of long and short (with leachate recirculation) term numerical simulations comparing them with site results. Finally results as well as advantages and drawbacks of the model are discussed. The results show that the mathematical model is able to reproduce the hydro-thermo-biological behavior of a bioreactor landfill in different conditions, with and without leachate recirculation, and leads to a better understanding of important thermal and biological parameters.

1. Introduction

Biodegradation of municipal solid waste (MSW) in landfills is a complex and variable process. Landfills develop distinctive microbial ecosystems during different decomposition phases. Waste decomposition phases ranging in number from three to six or more have been identified by different investigators depending on the data-base and purposes of each study. A four-phase characterization of refuse decomposition (Barlaz et al., 1989) consists of an aerobic phase, an anaerobic acid phase, an accelerated methane production phase and a decelerated methane production phase. In the accelerated methane production phase there is a rapid increase in the rate of methane production to some maximum value. Methane concentration of 50–70% is typical of this phase with the balance of the gas being mainly carbon dioxide. There is a little hydrolysis of solids during this phase and the methanogenic biomass increases. In the fourth phase of waste decomposition, decelerated methane production phase, the rate of methane production decreases even though methane and carbon dioxide concentrations remain constant at about 60% and 40%, respectively. Hydrolysis of solids controls the rate of methane production in that there is no longer an accumulation of carboxylic acids to serve as soluble substrate.

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It is believed that the major differences in waste decomposition under different conditions will be the time required for the different degradation phases, the methane production rate, and possibly the methane yield. These parameters are influenced by several conditions such as temperature and saturation (Barlaz and Ham, 1993).

The landfill bioreactor approach has been developed to optimize landfills as biological treatment systems and to reduce the time of landfill stabilization. Fast degradation rate in bioreactor landfills is an attractive feature of this innovative technology (Reinhart and Townsend, 1997). Enhancement in the biodegradation is usually achieved by recirculating the leachate collected from the bottom of the landfill. Recirculation of leachate helps the landfill to maintain a wet environment in addition to supplying nutrients needed for biodegradation. The idea of enhancing waste decomposition by the addition of supplemental water and/or recirculation of leachate was first proposed thirty-three years ago (Pohland, 1975).

Leachate recirculation is stimulatory for biodegradation because liquid movement distributes the inocula, minimizes local shortage of nutrients and dilutes potential toxins (Novella et al., 1997). However, in the absence of active acetogenic and methanogenic populations, recirculation of leachate may cause an accumulation of volatile fatty acids (VFAs). VFAs which are transferred from the acidogenic to the methanogenic areas, serve as the precursor for methane production. High VFA concentration inhibits both methanogenesis and hydrolysis/acidogenesis (Vavilin et al., 2002).

Mathematical models could be used to simulate the coupled biological and hydraulic behavior of bioreactor landfills and help to a better understanding of processes taking place over stabilization time and during leachate recirculation. These models could make it possible to predict gas and leachate production and to optimize the time and cost of operating bioreactor landfills by optimizing the volume of injected leachate, the number and spacing of injection devices and the duration of recirculation.

The mathematical models which have been developed previously (El Fadel and Findikakis, 1996; Vavilin et al., 2002) are usually based on constant values of temperature in controlled laboratory conditions or controlled industrial digesters and so they can hardly be directly applied to landfills where temperature is the result of coupled processes. In these models the growth and decay rates of biomass are supposed to be constant, at a given temperature.

In this work we have developed a numerical model based on the finite volume method (Benard and Eymard, 2005; Eymard et al., 2000) incorporating basic concepts from hydrodynamics and microbiology to simulate the hydraulic, thermal and biological behavior of anaerobic bioreactor landfills during the methanogenic phase. This model is composed of a two-phase flow model of leachate and biogas based on Darcy's law, coupled with a biological model of heat and gas generation, considering the effects of saturation and temperature changes on the biological behavior. The two-phase flow model is based on mass conservation equations of each fluid phase and on energy conservation equations, considering heat transport by conduction, convection and a heat production term which is introduced by the biological model. The development of the biological model of heat and gas production is based on Monod's model for microbial growth (Monod 1942, 1949, 1950), later adapted to anaerobic digestion by Findikakis and Leckie (1979), Halvadakis (1983) El Fadel (1991) and El Fadel and Findikakis (1996). Moreover, our biological model includes features introduced by Vavilin et al. (1999, 2000, 2002), who considers the effects of generation of volatile fatty acids (VFA) on methanogenic bacterial activity and methane production. In this paper we present first the mathematical equations and basic assumptions of the model. The model is then applied to the simulation of two real bioreactor landfills in France. We present the results in terms of temperature changes, solid degradation, VFA production, methanogenic biomass growth and gas production and we finally present and discuss the simulation results in the case of leachate recirculation.

2. Mathematical model

2.1. Main assumptions

Some important assumptions of this model are: 1) the landfill is considered as a homogeneous and anisotropic three-phase porous medium. The three phases are solid waste, gas and leachate, where gas is considered as a mixture of CO_2 and CH_4 with equal percentages (50% for each gas), 2) the solid phase of the landfill is considered to be non-deformable which means that we just consider the effects of degradation on gas and heat production and not on deformation and settlement of the landfill, 3) the gas and liquid phases are considered to be immiscible and Darcy's law is applicable for both fluid phases, 4) there is a thermal equilibrium between the three phases and 5) thermal radiation is neglected. Finally to simplify the model and to avoid a long list of undefined parameters, only one type of biomass (methanogenic biomass) is considered in the biological model.

2.2. Hydrodynamic model

As mentioned, the hydrodynamic model is based on mass and energy conservation equations. The mass conservation equation for each liquid phase is expressed by:

$$\frac{\partial m_P}{\partial t} + \nabla . \left[\rho_P \frac{k_{rP} k}{\mu_P} (- \nabla P_P + \rho_P g) \right] = \alpha_{bP} \quad (1)$$

where $m_l = \phi S \rho_l$, $m_g = \phi(1-S)\rho_g$ are the masses of phase $P = l, g$ per unit volume of porous medium (kg/m^3), α_b is the production rate for each phase, S is the liquid saturation and ϕ and k are the porosity and absolute permeability (m^2) of the porous medium (landfill waste). ρ_P, k_{rP} and P_P are the density, relative permeability and pressure for phase p. Gas density is defined by the perfect gas law, $\rho_g = P_g M_g / RT$, where M_g is the molecular mass of the mixture of methane and carbon dioxide and R is the universal gas constant. Gas pressure is defined as the sum of the liquid and capillary pressures. For the relative permeability and capillary pressure we have used Van Genuchten's (1980) model:

$$k_{rl} = S_e^2 \left[1 - \left(1 - S_e^{1/m} \right)^m \right], \quad k_{rg} = (1 - S_e)^2 \left[1 - S_e^{1/m} \right]^m \quad (2)$$

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where k_{rl} and k_{rg} are the relative permeabilities of fluid and gas, respectively, m is Van Genuchten's parameter, $S_e = (S - S_r)/(1 - S_r)$ is the effective saturation which defines the mobile fluid in a medium and S_r is the residual saturation. Capillary pressure is expressed by:

$$P_c = -\rho_l g \psi(S), \quad \psi(S) = \frac{1}{\alpha}\left(S_e^{-1/m} - 1\right)^{1/n} \tag{3}$$

where $n = 1/(1 - m)$ and $\alpha(m^{-1})$ is a parameter which defines the capillary height. The energy equation can be written as follow:

$$T\frac{d\eta}{dt} + \sum_{P=l,g} g_P \frac{dm_P}{dt} = -\nabla.(h_P \rho_P V_P) - \nabla.q + \alpha_q \tag{4}$$

V_P, $h_P(P_P, T)$, q, g and T are respectively Darcy's velocity of phase P, enthalpy per mass unit of phase P, conductive heat flux defined by Fourier's law as $q = -\lambda\nabla T$, gravity acceleration and temperature. g_P is defined as: $g_P(P_P, T) = h_P(P_P, T) - T\eta_P(P_P, T)$, where $\eta_P(P_P, T)$ is the entropy per mass of phase P. α_q is the heat source added to the model using the biological model of gas and heat production.

2.3. Biological model

The biological model consists of different parts corresponding to different steps of the degradation process. Fig. 1 shows the structure of this model and different steps of degradation of the solid substrate: production of volatile fatty acids (VFA) from hydrolysis and finally methanogenic biomass and biogas production. VFAs serve as the precursor for methane production. High VFA concentrations inhibit both methanogenesis and hydrolysis/acidogenesis (Vavilin et al., 2002). This model is presented by Eqs. (5)–(10).

2.3.1. Hydrolysis

The most important part in the biodegradation process of the solid substrate in landfills is hydrolysis, which is represented using first order kinetics (El Fadel and Findikakis, 1996):

$$\frac{dA}{dt} = -\sum_{i=1}^{3}[A_i.\lambda_i(T).fs(A_s).S_l] \tag{5}$$

A_i is the fraction of each biodegradable component of solid substrate which could be rapidly: $i = 1$, fairly: $i = 2$ and slowly: $i = 3$ biodegradable and $\lambda_i(T)$ is the degradation kinetic for each biodegradable component A_i defined by Arrhenius Law:

$$\lambda_i(T) = \beta_i \exp(-Ea_i/RT) \tag{6}$$

β_i (s^{-1}) is a constant and Ea_i is the activating energy of each component. S_l is the saturation function described as: $S_l = (S - S_{min})/(1 - S_{min})$, where S_{min} is considered as a minimum saturation in which hydrolysis could be stopped. This function considers the direct effect of saturation on acceleration of hydrolysis with increasing saturation. $fs(A_s)$ describes VFA inhibition of hydrolysis. The inhibition terms of hydrolysis and methanogenesis write respectively:

$$fs(A_s) = \frac{1}{1 + \{A_s/k_1(1 + k_a/\mathrm{Hb})\}^2}, \quad gs(A_s) = \frac{1}{1 + \{A_s/k_2(1 + k_a/\mathrm{Hb})\}^2} \tag{7}$$

where A_s is the VFA concentration produced from hydrolysis of the solid substrate, k_a is the dissociation constant, Hb is the proton constant and k_i is the inhibition constant for $i = 1$; hydrolysis and $i = 2$; methanogenesis.

2.3.2. Gas production

VFA:

$$\frac{dA_s}{dt} = -\beta\frac{dA}{dt} - \rho_{App}(T).gs(A_s).\left(\frac{A_s}{K_{s_A} + A_s}\right).X - \alpha_A.q\frac{dA_s}{dZ} \tag{8}$$

Biomass:

$$\frac{dX}{dt} = Y.\underbrace{\rho_{App}(T)}_{production}.gs(A_s).\frac{A_s}{K_{s_A} + A_s}.X - \underbrace{K_d(T)}_{decay}X - \alpha_X.q\frac{dX}{dZ} \tag{9}$$

Biogas:

$$\frac{dB}{dt} = (1 - Y)\rho_{App}(T).gs(A_s).\left(\frac{A_s}{K_{s_A} + A_s}\right).X \tag{10}$$

β is a stoichiometric coefficient, α is the fraction of VFA and biomass transferred by liquid flow, q is the volumetric liquid flow rate per surface area, K_{s_A} is the half saturation constant for acidogenesis (kg/m^3) and Y is the methanogenic biomass formed per mass of VFA utilized (kg/kg). $K_d(T) = K_{d\,max}.dD/dT$ and $\mu_{App}(T) = Y.\rho_{App}(T) = \mu_{max}.gs(A_s).A_s/K_{s_A} + A_s.dGr/dT$ are the decay and production rates of methanogenic biomass (d^{-1}), respectively, where $K_{d\,max}$ and μ_{max} are the maximum specific decay and production rates of methanogenic biomass (d^{-1}) at optimal temperature. In the existing biological models that we have found in the literature, the temperature is considered to be constant and these models are based on a constant value for production and decay of the biomass (El Fadel and Findikakis, 1996; Vavilin et al., 2002). As our model is based on a coupled behavior as a function of temperature it is very important to consider the temperature-dependence of

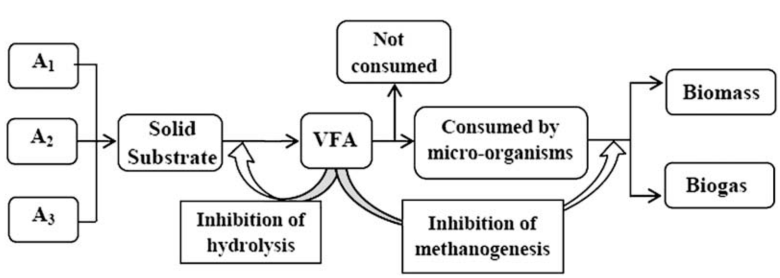

Fig. 1 – Simplified biological model of solid waste degradation and biogas production.

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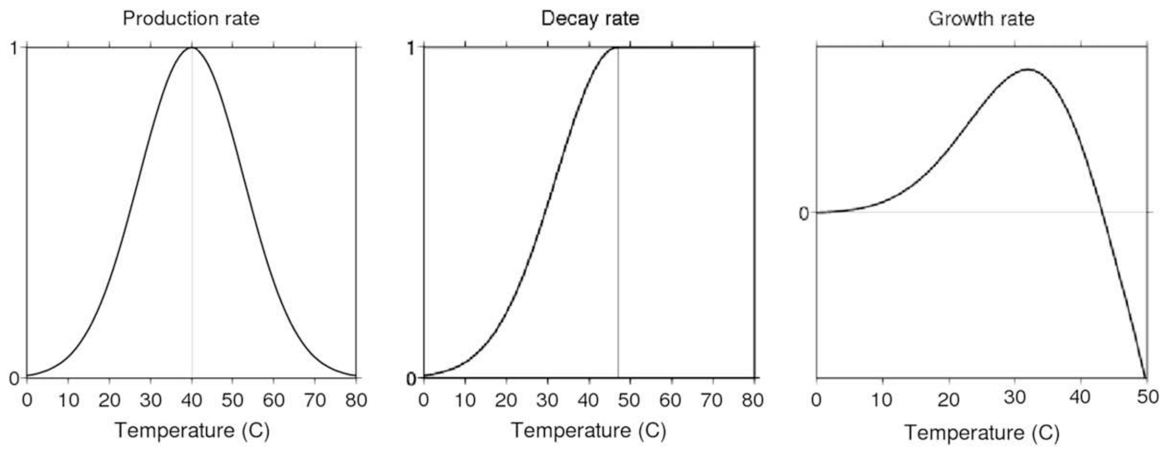

Fig. 2 – Production, decay and growth rate of the biomass as a function of temperature.

bacterial activity. Fig. 2 shows the production, decay and growth rate curves used in our model. The growth rate is represented as the difference between production and decay rates and is usually introduced in the literature with Rosso's law, without the negative side. Our conjecture is that it seems reasonable to prolong the curve, following the slope. This negative part shows that the decay rate of the biomass could exceed the production rate. Considering the maximum specific values of production and decay rates, it could be noticed that the negative part of this curve does not represent negative values of growth rate, but a higher rate of biomass decay compared to its production. Numerical tests show that if we do not introduce the negative part, the temperature does not remain at the asymptotic level observed on the landfill sites (see Fig. 3). The maximum value for this curve is obtained at the optimum temperature of 35 °C which is considered to be the optimum temperature of mesophilic bacteria.

2.3.3. *Heat generation*

Heat generation in landfills is a result of energy released from a large set of parallel and sequential biochemical reactions, beginning with the hydrolysis of solid substrates and ending with carbon dioxide and methane production. The reactions which happen in a landfill are too numerous and complex to be all considered in a numerical model (El Fadel, 1991; El Fadel and Findikakis, 1996). The heat generation rate in our model is obtained from the hydrolysis and methane production rates, as follow:

$$\alpha_q = \beta\frac{H_{hyd}}{M_{VFA}}.\frac{dA}{dt} + \frac{H_{meth}}{M_{meth}}.\frac{\alpha_b}{2} \tag{11}$$

where H (kJ) is the energy released per mole of VFA/methane produced and M is the molar mass of VFA/methane. α_b is the biogas production rate (dB/dt). As mentioned above, there is a large set of reactions with different enthalpy release during biogas production and so a large range of values for H_{meth} can be found in the literature. Lanini (1998) proposed values between 40 and 255 kJ, Aran (2000) proposed values between 2 and 60 kJ, Augenstein et al. (1999) proposed an average value of 68 kJ and finally El Fadel and Findikakis (1996) Proposed 108 kJ. In our model we used the following values for heat production: $H_{hyd} = 170$ kJ/mol and $H_{meth} = 80$ kJ/mol.

2.4. *Numerical methods*

The numerical solution of the mathematical model is obtained using a finite volume method, which consists in a set of nonlinear discrete balance equations in grid blocks, coupled with the conservation equations. This system is solved using Newton's method (Benard and Eymard, 2005; Eymard et al., 2000). The advantage of this method over other methods for solving the conservation equations is that nonstructured grid blocks could be solved as well as structured ones. Moreover there is flux conservation between two control volumes K and L, which can be expressed by the relation: $F^n_{K,L} = F^n_{L,K}$. Considering a finite volume mesh of the domain, consisting of N grid blocks indexed by ($i = 1,\ldots, N$), the volume of cell i is denoted by V_i. The subscript j stands for any cell having a common interface $i|j$ with cell i. Eq. (12) gives, after time and finite volume discretization, a set of coupled nonlinear equations (Eymard et al., 2000):

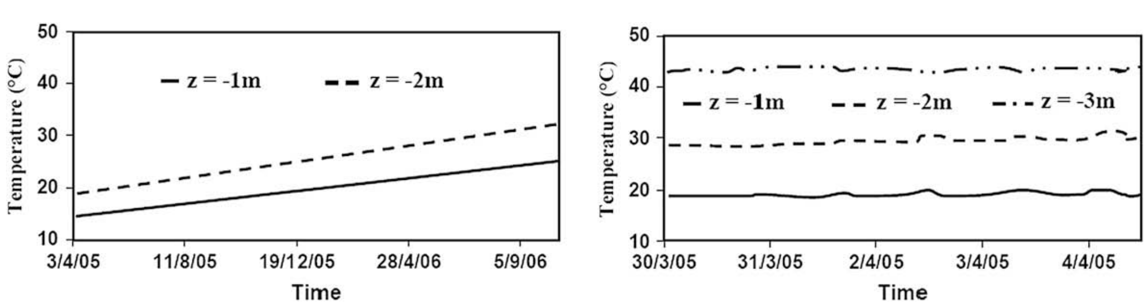

Fig. 3 – Temperature measured at two locations of a landfill bioreactor in France.

$$V_i \frac{m_{Pi}^{n+1} - m_{Pi}^n}{t^{n+1} - t^n} + \sum_{i|j} F_{wij}^{n+1} = 0$$

$$V_i \left[T_i^{n+1} \frac{\eta_{hi}^{n+1} - \eta_{hi}^n}{t^{n+1} - t^n} + \sum_{P=l,g} g_{P_i}^{n+1} \frac{m_{Pi}^{n+1} - m_{Pi}^n}{t^{n+1} - t^n} \right] + \sum_{i|j} F_{hij}^{n+1} = \overline{Q}_i^{n+1} \quad (12)$$

where $\overline{Q}_i^{n+1}$ is the heat source term in grid block i. The water flux F_{wij}^{n+1} and the energy flux F_{hij}^{n+1} across the interface $i|j$ are evaluated using an implicit finite difference scheme with respect to the pressures, the saturations and the temperatures of grid blocks i and j. For each time step, we applied an adapted Newton's method to find an approximate value of thermodynamic and hydraulic unknowns of the whole system of equations (saturation, pressure and temperature). This method, classically used in the oil reservoir simulation setting (Aziz and Settari, 1979), appears to be very stable and efficient for solving conservation equations of mass and energy.

2.5. *Model parameters*

The biological parameters are chosen from values proposed by El Fadel and Findikakis (1996), Vavilin et al. (2002) and the literature review done by Meima et al. (2007). The values are presented in Table 1. The initial values are 0 and 0.2 (kg/m^3) for VFA and humid biomass concentrations, respectively. The initial biodegradable solid is considered to be 30% of the waste mass where 5% is rapidly, 10% is fairly and 15% is slowly biodegradable. The degradation kinetics is between 0.0086 (d^{-1}) and 0.086 (d^{-1}) for different degradability rates.

We applied the coupled model to an 80 m × 10 m homogeneous two dimensional domain representing a typical cross section of a bioreactor landfill. The boundary conditions are defined as follow: a zero-flux condition on the top horizontal boundary; zero-flux condition on the vertical boundaries and atmospheric pressure (perfect drainage) on the bottom horizontal boundary. The temperature of the vertical and base boundaries is 10 °C, and on the top boundary changes between 2 and 26 °C within a year using a sinusoidal function. The initial saturation is obtained indirectly from electrical resistivity measurement which is a function of saturation (Grellier et al., 2003). Using these measurements and sensibility analysis by the numerical model, we obtained 25 percent (by volume) of saturation for the site. We assumed that the initial gas pressure in the landfill is atmospheric. The porosity and hydraulic conductivity are 0.3 and 1e−4 m/s, respectively.

Thermal conductivity of the landfill is found to be a very important parameter for the temperature results during ten years of numerical simulation and also during leachate recirculation. For the first case, it influences the thermal gradient between layers and thermal exchange with the outside. For the second case, during leachate recirculation, it influences the curve of temperature increase just after leachate recirculation. Higher thermal conductivities lead to a faster increase of temperature. For a ten year simulation, we first considered an average thermal conductivity proposed by other authors (Aran, 2000) of about 0.4 W/m K corresponding to relatively dry waste. As the initial saturation of the landfill is estimated at about 0.25, it seemed to be a good assumption. We present first the results with this thermal conductivity. The numerical simulations in the case of leachate recirculation and their comparison with site data lead us to a better estimation of this parameter between 0.6 and 0.8 W/m K.

Table 1 – Biological parameters for numerical simulation

μ_{max} (d^{-1})	K_d (d^{-1})	Y (kg/kg)	K_{S_A} (kg/m^3)	β	α
0.35	0.002	0.08	1.0	0.5	0.1

2.6. *Site temperature*

Under the assumption that temperature distribution in a landfill could provide essential insights in its coupled thermo-hydro-biological behavior, we studied the site data of a French pilot bioreactor landfill. The temperatures were measured with preinstalled sensors at different depths under the top cover layer (Fig. 3). We observed that temperature varies with seasonal changes between 15 and 20 °C in the upper layers and 35–40 °C in deeper layers (maximum 3 m under the horizontal injection points).

Temperature changes can range between 6 and 15 °C during 1–2 years, depending on the depth of the layer and the external temperature. Upper layers are more affected by the external temperature and show larger changes. The differences in terms of temperature values and trends at different locations but at the same depth could be explained by the heterogeneity of the waste which could cause different local values of gas and heat production. It could also be attributed to a difference between different stages of degradation of an initially homogeneous waste. A vertical thermal gradient of 1–10 °C/m is also generally observed.

3. Results and discussion

In this part we compare the temperatures measured in the pilot bioreactor landfill and the thermal results of numerical simulations performed with our model, using the above mentioned parameters and settings and we discuss the results.

3.1. *Long term results: during 10 years without leachate recirculation*

Fig. 4a shows the thermal evolution in a landfill during 10 years of simulation. The results are presented at different depths under the upper clay layer. The temperature increases up to 55 °C in the deeper layers. There is also a vertical gradient of 1–10 °C/m between different layers. The higher layers are more affected by external temperature and seasonal changes and there is a shift of peaks between the external and waste temperatures which is due to the thermal diffusivity, $\alpha = \lambda/\rho.C$ of the waste and top cover layer. This shift is also observed in the site results. In this bioreactor, the maximum values of temperature are observed in September–October and the minimum values in March–April, however the weather data of this region show that the external temperature is higher in spring than in autumn. As we can see in this figure, this shift is more remarkable in deeper layers of

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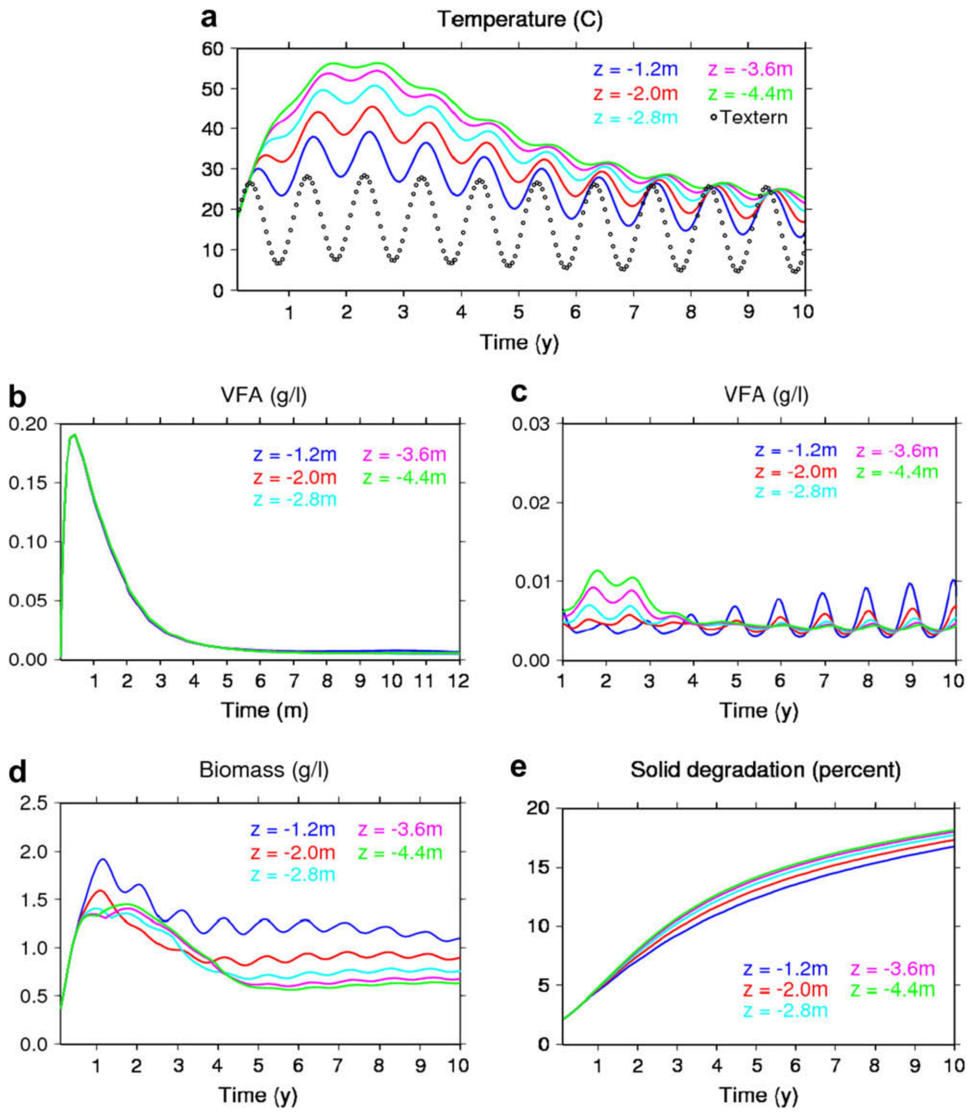

Fig. 4 – Thermal and biological behavior of a bioreactor landfill during 10 years (numerical simulation).

the landfill. These results could give good estimation of the thermal conductivity of the cover layer and waste and also show the necessity to install temperature sensors in deeper layers of real bioreactors to study how these values change in deeper layers with time. The results for solid degradation, VFA production and biomass changes are presented in Fig. 4 at different depths.

Different phases are observed from the numerical results and can be summarized as follow: during about six first months of simulation, there is a high production of VFA (Fig. 4b). Meanwhile, the methanogenic biomass increases in different layers consuming more and more VFA until VFA concentration drops to lower levels. This process increases the degradation of solid waste and methanogenesis. During the next three years the temperature and gas production of the landfill increases and depending on the temperature at different depths, different behaviors are observed for biomass growth and VFA changes. In deeper layers where the temperature is above 45 °C, the biomass decreases because of the decay rate which is the maximum in this range of temperature and so VFA concentration increases. However, in upper layers the biomass values are large enough to consume the VFA produced by hydrolysis and to keep the values under 0.01 g/l (Fig. 4c). After five years, temperature and gas production decrease and the temperature in different layers tends to the external temperature. The hydrolysis rate decreases because most of the rapidly and moderately degradable waste fractions have been degraded. In this phase hydrolysis is a key factor to generate biomass. As we can see in Fig. 4 biomass decreases, allowing a slight VFA increase in different layers. These two last phases could be considered as accelerated and decelerated methanogenic phases, as explained in Section 1.

The maximum degradation of solid waste is about 17% of the total mass in 1 m^3 (Fig. 4e), considering that degradation decreases and tends to a constant value after 10 years of simulation. This value is comparable with the data given by Olivier (2003) who advocates a maximum absolute mass loss of 20% for a standard bioreactor landfill after 30 years. This value could help us to estimate orders of magnitude of biodegradation kinetics.

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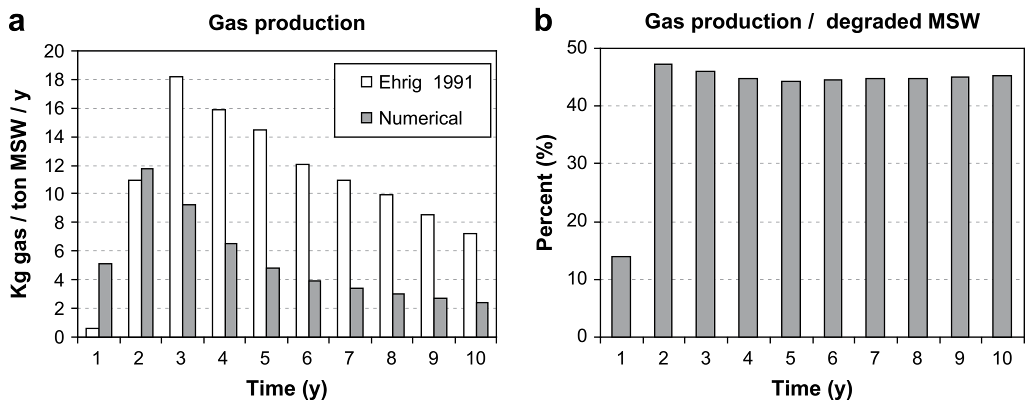

Fig. 5 – Annual gas production in a landfill, estimated from laboratory experimentations on municipal solid waste (Ehrig, 1991) compared to numerical results (a), and numerical results of gas production from degraded solid waste each year (b).

Fig. 5b shows that 13–47% of degradable waste is transformed into biogas. This value is validated by Ehrig (1991) who has observed that less than 50% (7.2–41.7%) of the initial mass of organic waste is transformed into biogas. This author estimated that maximum gas production in a landfill occurs about 3–5 years after the waste has been landfilled (Fig. 5a). Haarstad (1997) extrapolated gas production during 40 years and proposed a value of 168 kg gas/ton of solid MSW for total biogas production. Thomas (2000) proposed that maximum biogas production is about 215 kg gas/ton of MSW. Gas production obtained by our numerical simulation is about 53 kg gas/ton MSW within 10 years. This value is less than the values proposed in the literature but it should be considered that the latter values are based on laboratory experimentations and are usually under condition of maximum gas production potential. Gas production reaches a peak after one year of numerical simulation. Comparing this time with Ehrig's estimation (3–5 years), it should be noted that the numerical model does not consider the aerobic and acidogenic phases.

3.2. Short term results

3.2.1. *Site-1: during two weeks with leachate recirculation*

Here we present the numerical results of temperature changes in the same bioreactor landfill during leachate recirculation. The results are presented at two different depths under the top clay cover compared to site data. Site data are obtained from temperature sensors placed at different depths under the top clay layer at about one meter distance from the injection line. To install the temperature sensors, five meters long and three meters deep trenches have been excavated near the injection lines and the sensors have been installed on vertical lines at 0 m, −1 m, −2 m and −3 m under the top cover layer. Then the trenches have been backfilled with compacted waste to minimize density heterogeneities. Waste settlement and aging of materials can cause disconnection problems in sensors and measurement errors. For these reasons only part of the sensors are still operating after about seven years and little data are available. The maximum error of these sensors when manufactured is $0.3\,^{\circ}\mathrm{C} + 0.005T$, where T is the measured temperature.

The injection parameters are presented in Table 2. Earlier works of the authors (Gholamifard et al., 2007) show that hydraulic parameters as porosity, hydraulic conductivity and also recirculation discharge and leachate temperature are key parameters affecting temperature variation trends under injection points. Many numerical simulations have been taken out to estimate the best values for these parameters. For this model we used a porosity and a hydraulic conductivity of 30 percent and 1e−4 m/s, respectively. Unfortunately, leachate temperature was not measured but is believed to be close to the outdoor temperature because leachate used for recirculation was stored in an open pond on the landfill site. The best leachate temperature values deduced from test numerical simulations are 3 °C and 8 °C, respectively for the first and second injection. As waste temperature before leachate injection is about 33 °C at $z = -1$ m, we started simulating the first injection when the temperature of the model is close to this value, after about 1.5 years (512 days) of numerical simulation. The first simulated injection is on day 527 and the following simulated injection is one week later. The site data are available up to two days after the second injection.

As mentioned previously thermal parameters such as waste heat capacity and thermal conductivity are estimated with the use of a series of numerical simulations. We observed that with a thermal conductivity of 0.4 W/m K which was used for the long term simulation during 10 years, the model is not able to reproduce the same shape of temperature increase after leachate injection and gives a linear form. We thus increased the thermal conductivity to 0.6 and 0.8 (Aran (2000) proposed a value between 0 and 0.8 W/m K and Yessiler et al. (2005) used 1 W/m K). The numerical results presented in Fig. 6 are obtained in grid elements situated at −0.8 m and −1.6 m under the top clay cover and a 1 m distance from the injection

Table 2 – Injection parameters

Parameters	First injection	Second injection
Injection duration	4h00	2h10
Discharge (m^3/h)	6.0	11.0
Outdoor temperature (°C)	2.6	10.7

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point and seem to be the closest to the site data at −1 m and −2 m. It should be considered that because of waste settlement and degradation, the actual position of the sensors is probably not exactly the initial one. The temperature results are presented in Fig. 6 with two thin solid lines which are: $T_{Site} \pm (0.3 + 0.005 \times T_{Site})$, to take into account the possible errors of measurement. It should be noted that this error is the manufacture error and the effects of aging and disconnection are not considered in it.

As we can see in Fig. 6, the best value for this model is 0.8 W/m K which matches the best the site data especially at depth −1 m. Waste heat capacity for this landfill is estimated between 1000 and 1100 J/K kg which is in the range of the values proposed by different authors (Meima et al., 2007). As the model presented herein is a homogeneous model, it cannot consider the effects of heterogeneity of thermal and hydraulic parameters which are believed to affect the measured site data. Increasing the thermal conductivity of the landfill from 0.4 W/m K to 0.8 W/m K affects the long term thermal and biological behaviors. Temperature gradients decrease with an increasing thermal conductivity and the maximum temperature obtained with 0.8 W/m K is 40 °C instead of 55 °C obtained with 0.4 W/m K. Studying the long and short terms behaviors of the landfill could lead us to conclude that thermal conductivity of the waste varies throughout the landfill not only because of waste heterogeneity but also because of variations in waste saturation. Waste saturation and hence thermal conductivity are probably higher under the injection lines.

Hydraulic and thermo-biological changes during leachate recirculation are presented in Fig. 7 for different layers. To study how leachate injection affects the thermo-biological behavior, numerical simulations are conducted for periods of about 160 days. It should be mentioned that in this short term simulation leachate does not have any biological character; it means that there is no VFA and biomass injected with the leachate, however the model is able to consider these parameters.

As we can see in Fig. 7, leachate injection decreases the temperature in different waste layers below the injection point during recirculation. Temperature increases very rapidly just after leachate injection is stopped because of diffusivity of the warmer surrounding waste. Temperature changes after leachate recirculation are due to seasonal changes of outdoor temperature and it seems that leachate injection does not thermally affect the long term behavior. Saturation increases during leachate recirculation in the layers situated directly below injection point. Leachate injection seems to affect hydrolysis of the solid waste most of all and the first injection period has a very significant effect. The first cause of this phenomenon is that degradation is related to the difference between initial saturation of the landfill and saturation after injection. The second cause is temperature changes which are more important during the first injection. VFA concentration decreases slightly during leachate injection as the nutriments are displaced by flow and increases immediately after leachate injection because of the acceleration of hydrolysis. Methanogenic biomass also decreases during the injection, 10 percent of the biomass being displaced by the leachate flow, and it increases after consuming the high concentration of VFA. This process increases gas production just after each period of leachate recirculation, as we can see in Fig. 7f.

The results show that leachate recirculation does not affect the long term thermal behavior of the landfill but has a significant effect to accelerate degradation of solid waste and to increase gas production. It should be noted that both waste degradation and gas production reach a peak during leachate recirculation and drop back to previous values within a few months without further leachate injection.

3.2.2. *Site-2: overall results*

In the second bioreactor landfill site that we have studied, only a limited number of data under each injection line is available. Nevertheless temperature data at different points of the site were available and could lead to overall interpretation and discussion. A significant number of temperature sensors were out of order and showed either negative values or inconsistent values around 80–90 °C but nevertheless showed significant temperature changes. Temperature changes in waste are visibly affected by seasonal changes of external temperature, like in site-1. In some places, the thermal gradient is about 5–6 °C/m whereas in other locations almost no gradient is observed. The gradients observed on site-2 are

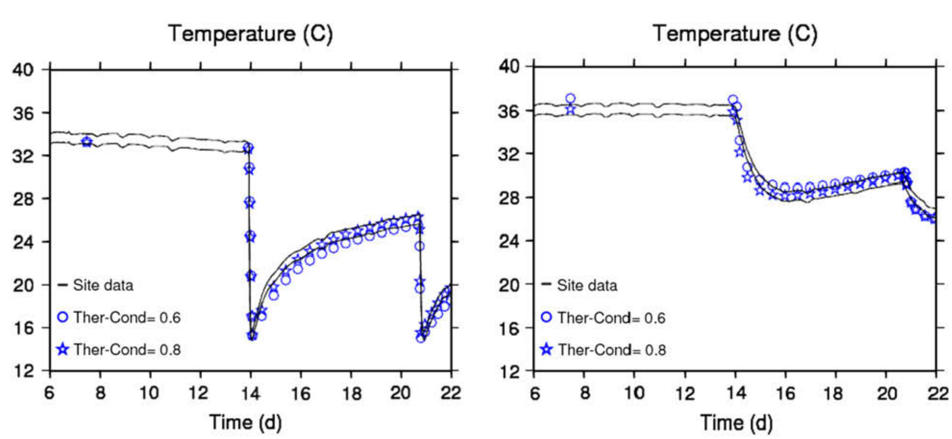

Fig. 6 – Simulated temperature changes during leachate recirculation at z = −1 m (left) and z = −2 m (right) compared to site data, using two values of thermal conductivity.

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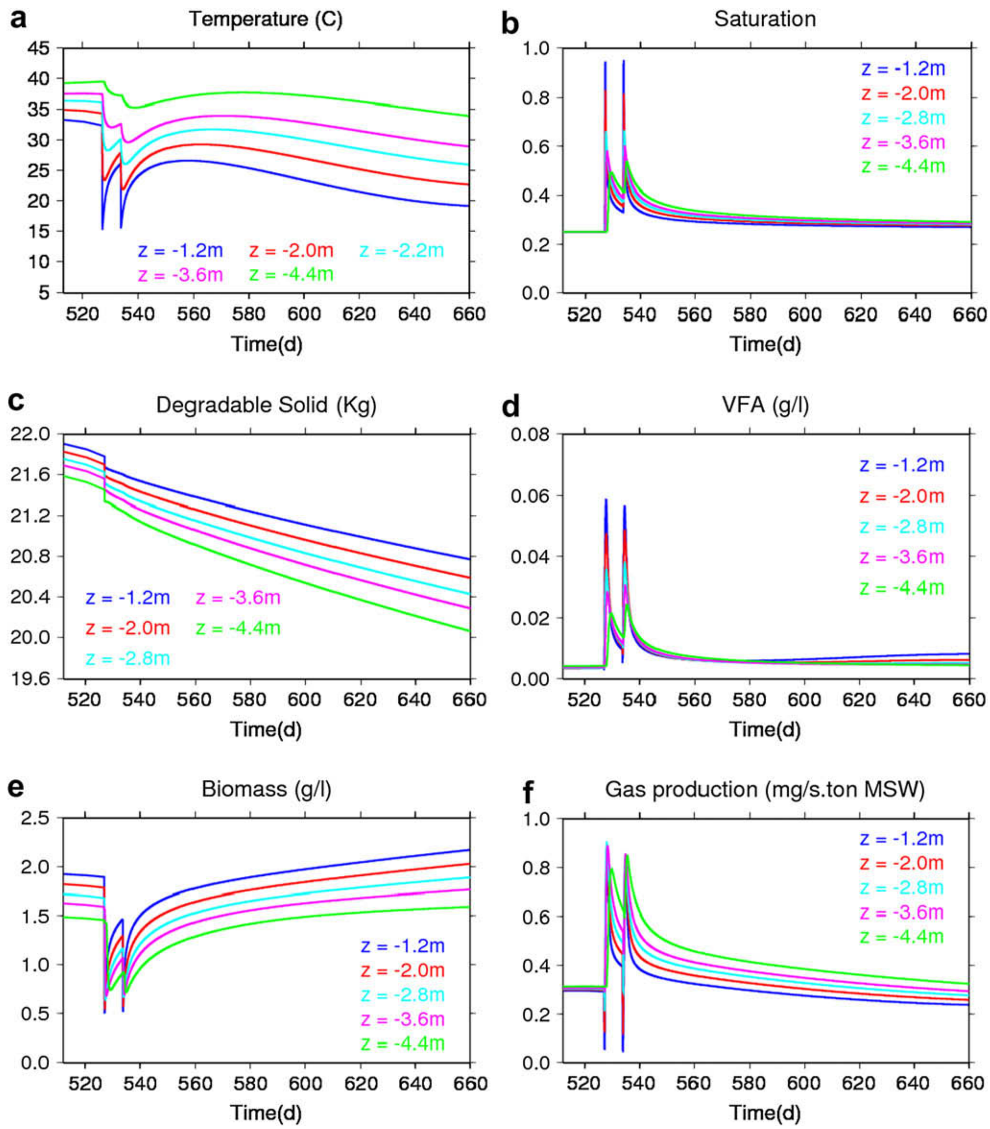

Fig. 7 – Biological behavior of the landfill.

smaller than on site-1 which could be a result of a higher thermal conductivity of waste in site-2. This assumption is in accordance with the temperature curves observed during leachate recirculation. Indeed, temperature rises more rapidly after leachate injection in site-2 as compared to site-1. This could also be attributed to waste heat capacity, channeled flow or injection parameters such as leachate discharge and temperature.

Numerical simulation results show the importance of thermal conductivity and heat capacity of the waste on temperature and especially on temperature increase after leachate recirculation. It is also observed that temperature after leachate recirculation is influenced more by external temperature than by waste temperature prior to leachate injection. It is thus believed that thermal conductivity of the cover layer is a very important parameter. The results (Fig. 8) show that the model could not reproduce the temperature increase after leachate recirculation as fast as what is observed on site-2. Numerical results shown on Fig. 8 are

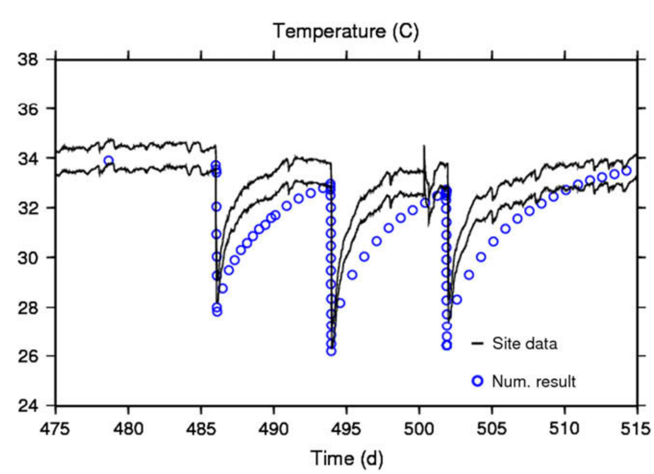

Fig. 8 – Temperature changes during leachate recirculation with a discharge of 7 (m^3/h) and a duration of 2h30, 1h36 and 1h40 for the first, second and third injections, respectively.

obtained using the largest thermal conductivity (1 W/m K) and the smallest heat capacity (900 J/K kg) in the range of waste thermal parameters found in the literature. Fitting the experimental temperature curve would require inconsistent values for these parameters. One explanation could be that trenches in which the temperature sensors were installed could have been filled with less compacted waste. In this case, whereas thermal properties would be similar to surrounding waste, hydraulic conductivity of waste in these trenches could be higher. This could lead to channeled flow in the trenches, and to lesser cooling of the surrounding waste than what is numerically simulated. The warmer surrounding waste would then transfer heat to the trench faster than what is simulated. Testing this hypothesis would require representing volumes with different hydrodynamic properties in the numerical simulation. We did not perform these tests in the present study. Fig. 8 shows small oscillations which could be attributed to daily thermal radiation on top cover layer. These oscillations are also observed on site-1.

3.3. *Discussion on the temperature data*

For our purpose site temperature data were only available down to three meters deep. It was thus impossible to study the behavior of deeper layers and to find out which temperature could reach a bioreactor landfill at larger depths. Knowledge of temperature in deeper layers would lead to a better understanding of thermal behavior of methanogenic bacteria, a better idea of their production and decay curves, as well as thermal parameters of the top cover layer.

4. Conclusions

We have developed a 3D mathematical model based on conservation equations and using a finite volume method to model bioreactor landfills. The coupled model contains a two-phase flow model based on Darcy's law and a biological model based on the simplified Monod's model (1949) and considering the biogas production via degradation of the biodegradable solid wastes and VFA production. In contrast to general biological waste degradation models, this model considers the effect of temperature on growth rate of biomass. The results of numerical simulations lead to the following conclusions:

(a) The stationary condition of temperature observed through site data is the result of heat exchange at the surface of the landfill combined with temperature-dependent microbial activity generating heat in the waste.
(b) Parameters influencing the thermo-biological behavior of landfills the most appear to be the thermal conductivity and heat capacity of waste, the initial values of biological parameters and of VFA concentration, and also the parameters controlling biomass growth such as production and decay rates. Waste thermal conductivity could be estimated using thermal gradients in different layers of a landfill. The thermal and hydraulic parameters are characteristic of each site; they change from one landfill site to another and should be obtained and estimated separately for each site.
(c) Short term simulation during leachate recirculation on two different sites show that short term temperature changes in the waste depend essentially on temperature and discharge of the injected leachate, heat capacity and thermal conductivity of waste in the neighborhood of the injection point, and variation of outdoor temperature which is observed in the upper layers.
(d) Short term simulations show one of the limitations of our homogeneous model. Representing waste heterogeneities is nevertheless complex. Numerous parameters may vary in space which affect the hydro-thermo-biological behavior of the landfill. Although our model is homogeneous, it showed good sensibility to changes in these parameters and could explain different in-situ observations.
(e) Simulations of leachate recirculation show that leachate injection increases the saturation of the waste and decreases the VFA concentration. Both phenomena increase the rate of solid degradation and methane production.

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Thermal behavior of a bioreactor landfill during leachate recirculation

Shabnam Gholamifard[1], Robert Eymard[2] and Christian Duquennoi[3]

ABSTRACT: The goal of a bioreactor landfill is to control gas and liquid flows and to optimize the valorization of waste as well as to decrease the stabilization time and increase the degradation rate by controlling microbiological activity. Enhancement in the biodegradation is usually achieved by re-circulating leachate collected from the bottom of the landfill. This is believed to help the landfill maintaining a wet environment in addition to supply nutrients needed for biodegradation. We have developed a two-phase flow model based on Darcy's law which is coupled with a biological model based on Monod's model. The biological model considers the hydrolysis of biodegradable solid waste, VFA production, growth of biomass and gas and heat production. This model makes it possible to study the coupled behavior of an anaerobic bioreactor landfill in methanogenic phase during leachate recirculation. This paper presents first the mathematical model and numerical methods for heat and gas generation. The model is applied to simulate a real bioreactor landfill in France. It then presents results of thermal numerical simulations comparing them with field experimental results. Finally the authors discuss the results as well as advantages and inconveniences of the model.

INTRODUCTION

A landfill bioreactor approach has been developed to optimize landfills as biological treatment systems and to reduce the time of landfill stabilization. Fast degradation rate in bioreactor landfills is an attractive feature of this innovative technology (Reinhart and Townsend, 1998). Enhancement in the biodegradation is usually achieved by recirculating the leachate collected from the bottom of the landfill. Recirculation of leachate helps the landfill to maintain a wet environment in addition to the supply of nutrients needed for biodegradation. Leachate recirculation is stimulatory for biodegradation because liquid movement distributes the inocula, minimizes local shortages of nutrients and dilutes potential toxins (Novella et al., 1997). Modeling biological and hydraulic phenomena within bioreactor landfills is a crucial issue to improve the efficiency of engineered landfills. The need to manage landfill sites and to monitor leachate injection and biogas production makes it more and more necessary to be able to predict the behavior of a bioreactor landfill during leachate recirculation. The objective of this work was to develop a numerical model

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incorporating basic concepts from hydrodynamics and microbiology to simulate the coupled behavior of a bioreactor landfill during leachate recirculation. The coupled model contains two parts: a two-phase flow model of leachate and biogas based on Darcy's law, using a finite volume method (Benard *et al.* 2005, Eymard *et al.* 2000), and a heat and gas generation model. The two phase flow model is based on mass conservation equations of each liquid phase and the energy conservation equations, considering heat transport by conduction, convection and a heat production term which is an output of the biological model. The development of the biological model of heat and gas production is based on Monod's model for microbial growth (Monod 1942, 1949, 1950), later adapted to anaerobic digestion by Finidikas *et al.* 1979, Straub and Linch 1982, Halvadakis 1983 and El-Fadel *et al.* 1991 and 1996, Vavilin *et al.* 1999, 2000 and 2002). This paper presents first the mathematical formulations and basic assumptions of the model. The model is applied to simulations of a real bioreactor landfill in France. The results of leachate recirculation simulations are presented and discussed, especially in terms of simulated temperature which is compared to measured temperature.

METHODS

General assumptions

The important assumptions of this model are: The landfill is considered as a homogeneous and anisotropic three-phase porous medium. The three phases are solid waste, gas and leachate, where gas is considered as a mixture of CO_2 and CH_4 with equal percentages. The solid phase of the landfill is considered to be non-deformable which means that we just consider the effects of degradation on gas and heat production and not on deformation and settlement of the landfill. The gas and liquid phases are considered to be immiscible and Darcy's law is applicable for both fluid phases. We assume that there is a thermal equilibrium between the three phases and thermal radiation is neglected. Finally to simplify the model and to avoid having a long list of undefined parameters, only one type of biomass (methanogenic biomass) is considered in the biological model.

Biological model

The biological model is explained in Fig. 1 and by Eq. 1 to 4. This figure shows the succession of different steps: degradation of the solid substrate (hydrolysis), production of volatile fatty acids (VFA) from hydrolysis and finally methanogenic biomass and biogas production. VFAs serve as precursor for methane production. High VFA concentrations inhibit both methanogenesis and hydrolysis/acidogenesis (Vavilin et al. 2002).

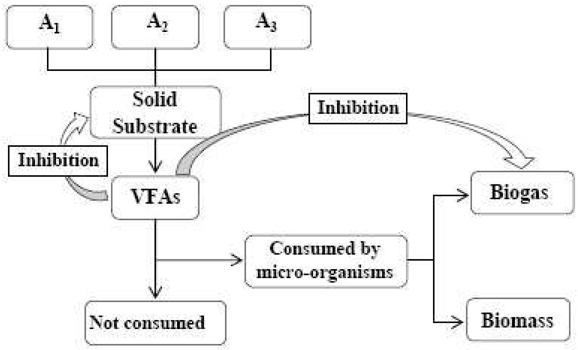

Fig. 1. Simplified biological model of digestion of solid waste

Solid substrate:

$$\frac{dA}{dt} = -\sum_{i=1}^{3}\left[A_i.\lambda_i(T).fs\right] \tag{1}$$

VFA:

$$\frac{dA_S}{dt} = -\beta\frac{dA}{dt} - \mu_{App}(T).gs.\left(\frac{A_S}{K_{S_A}+A_S}\right).X - \alpha_A.q\frac{dA_s}{dZ} \tag{2}$$

Biomass:

$$\frac{dX}{dt} = Y.\left(\underbrace{\mu_{App}(T)}_{Birth}.gs.\frac{A_S}{K_{S_A}+A_S}.X\right) - \underbrace{K_d(T)}_{Decay}X - \alpha_X.q\frac{dX}{dZ} \tag{3}$$

Biogas:

$$\frac{dB}{dt} = (1-Y)\mu_{App}(T).gs.\left(\frac{A_S}{K_{S_A}+A_S}\right).X \tag{4}$$

β is a stoichiometric coefficient, α is the fraction of VFA and biomass transferred by liquid flow, q is the volumetric liquid flow rate per surface area, K_{S_A} is half saturation constant for acidogenesis $\left(kg/m^3\right)$ and Y is the methanogenic biomass formed per mass of VFA utilized (kg/kg). In this model we have divided the *growth rate* (introduced in the literature with Rosso's law) into two curves: birth and decay rates and the growth rate will be the subtraction of decay rate from birth rate. Using this definition we have introduced an exponential birth curve (Meima et al., 2007) with a maximum at 40°C which is close to the optimum temperature of mesophilic bacteria. Decay rate as a function of temperature is defined by a *S* shape curve with a maximum at 46°C. The growth curve obtained from these two curves matches very well the curves which are usually presented in the literature as net growth rate of methanogenic bacteria (the positive side). So in this model $K_d(T) = K_{d\max}.dD/dT$ and $\mu_{App}(T) = \mu_{\max}.Y.gs.\frac{A_S}{K_{S_A}+A_S}.dGr/dT$ are the decay and birth rates of methanogenic biomass$\left(day^{-1}\right)$, respectively, where $K_{d\max}$ and $\mu_{\max}$ are the maximum decay and birth rates of methanogenic biomass (day^{-1}) at optimal temperature.

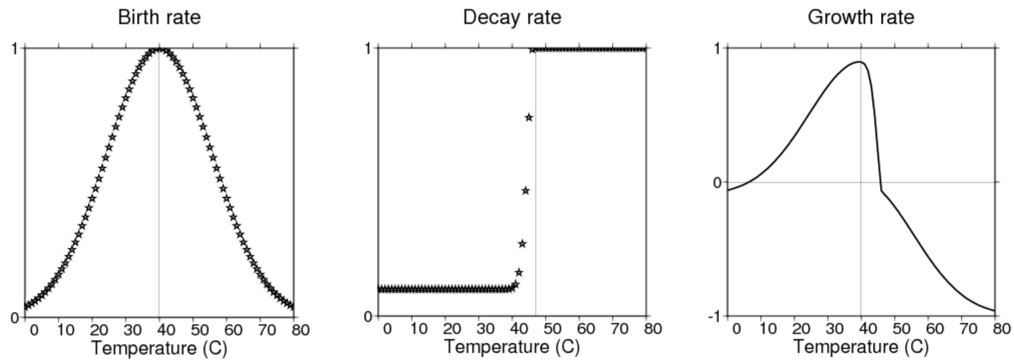

Fig. 2. Birth, decay and growth rate of the biomass as a function of temperature

The heat generation rate is defined as follow:

$$\alpha_q = \beta \frac{H_{hyd}}{M_{VFA}} . \frac{dA}{dt} + \frac{H_{meth}}{M_{meth}} . \frac{\alpha_b}{2}, \tag{5}$$

where $H(KJ)$ is the energy released per mole of VFA/methane produced and M is the molar mass of VFA/methane. α_b is the biogas production rate (dB/dt). As there is a large set of reactions with different enthalpy release during biogas production, we can find a large range of values for H_{meth} in the literature (Lanini, 1998; Aran, 2000; Augenstein *et al.*, 1999; El-Fadel *et al.*, 1996). In this model we used the following values for heat production: $H_{hyd} = 170$ and $H_{meth} = 80$ KJ/mol.

Numerical techniques

The numerical solution of the mathematical model is obtained using a finite volume method, which consists in a set of nonlinear discrete balance equations in grid blocks, coupled with the conservation equations. This system is solved using Newton's method (Benard *et al*. 2005; Eymard *et al*. 2000). The advantage of this method over other methods for solving the conservation equations is that non structured grid blocks can be solved as well as structured ones and there is flux conservation between two control volumes K and L, which can be expressed by relation: $F^n_{K,L} = -F^n_{L,K}$. Considering a finite volume mesh of the domain, consisting of N grid blocks indexed by (i =1, . . . ,N), the volume of cell i is denoted by V_i. The subscript j stands for any cell having a common interface $i|j$ with cell i. Equation 6 gives, after time and finite volume discretization, a set of coupled nonlinear equations (Eymard et al., 2000):

$$
\begin{aligned}
&V_i \frac{m_{Pi}^{n+1} - m_{Pi}^{n}}{t^{n+1} - t^{n}} + \sum_{\mathrm{i|j}} F_{wij}^{n+1} = 0 \\
&V_i \left[T_i^{n+1} \frac{\eta_{hi}^{n+1} - \eta_{hi}^{n}}{t^{n+1} - t^{n}} + \sum_{P=l,g} g_{P_i}^{n+1} \frac{m_{Pi}^{n+1} - m_{Pi}^{n}}{t^{n+1} - t^{n}} \right] + \sum_{\mathrm{i|j}} F_{hij}^{n+1} = \overline{Q}_i^{n+1}
\end{aligned}
\tag{6}
$$

where $\overline{Q}_i^{n+1}$ is the heat source term in grid block i. The water flux F_{wij}^{n+1} and the energy flux F_{hij}^{n+1} across the interface $i|j$ are evaluated using an implicit finite difference scheme with respect to the pressures, the saturations and the temperatures of grid blocks i and j. For each time step, we applied an adapted Newton's method to find an approximate value of thermodynamic and hydraulic unknowns of the whole system of equations (saturation, pressure and temperature).

RESULTS

We used a homogeneous two dimensional model on a rectangular $80m \times 10m$ domain. The boundary conditions are defined as follow: a zero-flux condition on the top horizontal boundary; zero-flux condition on the vertical boundaries and atmospheric pressure (perfect drainage) on the bottom horizontal boundary. The temperature is 15°C on both vertical and base boundaries, and 25°C on the top boundary. The results of temperature changes during leachate recirculation are presented at two different depths under the top clay cover and are compared to the site data. Site data are obtained from temperature sensors placed at different depths under the top clay layer at about one meter horizontal distance from the injection line. To install the temperature sensors, five meters long and three meters deep trenches have been excavated near the injection lines and the sensors have been installed on vertical lines at 0m, -1m, -2m and -3m under the top cover layer. Then the trenches have been backfilled with compacted waste to avoid density heterogeneities, as much as possible. Waste settlement and aging of materials can cause disconnection problems in sensors and measurement errors. For these reasons only part of the sensors are still operating after about seven years and little data are available. The maximum error of these sensors when manufactured is 0.3°C+0.005T, where T is the measured temperature.

The injection parameters are presented in Table 1. Earlier works of the authors (Gholamifard et al., 2007 and 2008) show that hydraulic parameters as porosity, hydraulic conductivity and also recirculation discharge and leachate temperature are key parameters affecting temperature variation trends under injection points. Many numerical simulations have been taken out to estimate the best values for these parameters. For this model we used a porosity and a hydraulic conductivity of 30 percent and 1e-4m/s, respectively. Unfortunately, leachate temperature was not measured but is believed to be close to the outdoor temperature because leachate used for recirculation was stored in an open pound on the landfill site. The best leachate temperature values we have deduced from test numerical simulations are 3°C and 8°C, respectively for the first and second injection. The initial waste temperature is about 33°C at z = -1m, from measured site data. Waste temperature data are available up to two days after the second injection.

Table 1. Injection parameters

Parameters	First injection	Second injection
Injection duration	4h00	2h10
Discharge (m^3/hr)	6.0	11.0
Outdoor Temperature (°C)	2.6	10.7

Thermal parameters as waste heat capacity and thermal conductivity are estimated with the use of a series of numerical simulation. We observed that the best value for thermal conductivity is between 0.6 and 0.8 W/m.K (Aran 2000 proposed a value between 0 and 0.8 W/m.K and Yessiller et al., 2005 used 1 W/m.K). The numerical results presented in Fig. 2 are obtained in grid elements situated at -0.8m and -1.6m under the top clay cover and a 1m distance from the injection point and appear to be the closest curves to the site data at -1m and -2m. It should be considered that with waste settlement and degradation effects on the situation of the temperature sensors, we could not be sure that these sensors are still in the exact initial positions that are mentioned in the reports. Site temperature results are presented in Fig.3 with two thin solid lines which are: $T_{Site} \pm (0.3 + 0.005.T_{Site})$, to consider the possible errors of measurement. It should be noted that this error is the manufacture error and the effects of aging and disconnection are not considered in it.

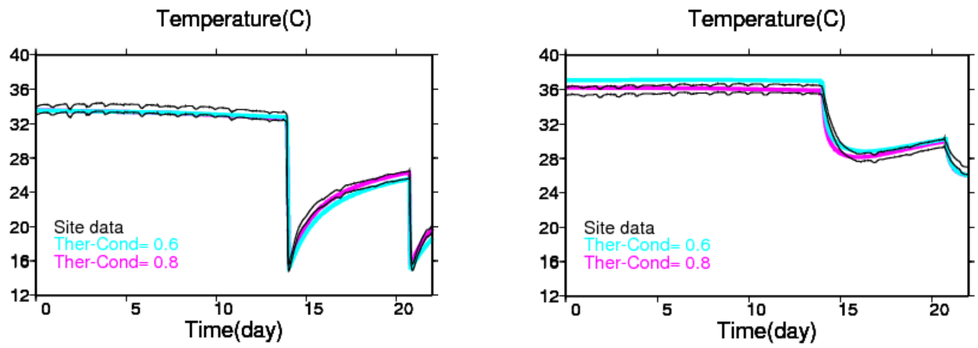

Fig. 3. Temperature changes during leachate recirculation at z = -1m (left) and z = -2m (right) compared to site data, using two values of thermal conductivity

As we can see in Fig. 3, the best value for this model is 0.8 W/m.K which matches the best to site data especially at depth -1m. Waste heat capacity for this landfill is estimated between 1000 and 1100 J/K.kg which is in the range of values proposed by different authors (Meima et al., 2007). As the model presented herein is a homogeneous model, it can not consider the effects of heterogeneity of thermal and hydraulic parameters which are believed to affect the measured site data.

CONCLUSION AND DISCUSSION

We have developed a numerical 3D model using finite volume method to simulate bioreactor landfills during leachate recirculation. A two dimensional problem of leachate injection and temperature variation is presented in this work. The model contains two parts which are coupled with each other: a two phase flow model based on Darcy's law and a biological model based on Monod's model (1949) which considers biogas production via hydrolysis of biodegradable solid waste and VFA production. Numerical simulations of a real bioreactor landfill have been taken out to study its thermal behavior during leachate recirculation. The results show that the stationary condition of temperature which is observed in the field data before leachate recirculation is a result of thermal exchanges with external temperature and also of bacterial activity which is in turn affected by temperature changes. Thermal conductivity and heat capacity of waste are very important parameters which influence temperature changes during and after leachate recirculation. Sensivity analysis leads to thermal conductivity ranging between 0.6 and 0.8 W/m.K and heat capacity ranging between 1000 and 1100 J/K.kg for this specific bioreactor landfill. The site temperature data are only available up to three meter deep. This makes it impossible to study the thermal behavior of deeper layers and especially the maximum temperature reached in depth. Knowledge of temperature in deeper layers would lead to a better understanding of thermal behavior of methanogenic bacteria, a better idea of their birth and decay curves, as well as thermal parameters of the top cover layer.
As a next step of this work, the model should be validated by more site data and experimental results. The biological behavior of a real bioreactor landfill should be discussed using this model. Both long term behavior and short term behavior during leachate recirculation episodes should be investigated.

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SIMULATION OF TWO-PHASE FLOW IN ANAEROBIC BIOREACTOR LANDFILLS

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SUMMARY: Early efforts to model the hydrodynamics of municipal solid waste landfills were based on Richards's law. These series of models consider a two-phase flow of gas and liquid in waste, considered as a porous medium. To simplify the model, gas pressure in the landfill is considered to be constant. These models can be used when dynamics of gas flow could be neglected. On the other hand, when we have leachate recirculation in a landfill, it is necessary to consider dynamics of gas flow to model the real behaviour of leachate flow. In this work we have developed a finite volume model of two-phase flow based on Darcy's law, considering biogas and heat production in anaerobic phase. This paper presents first the mathematical model of coupled flow and heat transport. The hydrodynamical model is then validated by the analytical solutions. It then discusses the selection of important parameters, using site results of electrical resistivity and back analysis. Finally it represents the results of numerical simulation of a real landfill and the strength and limitations of the model are discussed.

1. INTRODUCTION

The main goal of bioreactor landfills is to accelerate the biological stabilization of the solid waste with the basic process of leachate recirculation. The need to manage landfill sites and to monitor biogas production and leachate injection makes it more and more necessary to be able to predict the behaviour of a bioreactor landfill during leachate recirculation. Mathematical models could be used to optimize the volumes of injected leachate, injection duration and discharge flow as well as the number of injection lines in a landfill site and their distances.

We have developed a multiphase flow model of a bioreactor landfill, using a finite volume method (Benard et al., 2005; Eymard et al., 2000), which considers heat and gas generation. Models for gas and heat generation in landfills are generally based on the classical Monod's (1949) model for bacterial growth, adapted later by Findikakis *et al.* (1979), Straub & Linch (1982) and Halvadakis (1983) to waste degradation. These models are generally functions of water content of each biodegradable component of the waste which is assumed to be constant in time. On the other hand, bioreactor landfills are usually based on leachate recirculation. Under this condition, water content of waste components changes. So it is necessary to introduce a function of saturation for gas and heat production terms, which therefore change during leachate recirculation.

Here we first develop the mathematical model of two-phase flow with biogas and heat production, introducing the numerical techniques to solve the conservation equations. The model

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is then validated by analytical solutions based on the Buckley-Leverett equations. Then we introduce the parameters which are chosen basically from bibliographical researches and are used for the bioreactor model. Finally numerical results are presented and compared with field results during and after periods of leachate injection. Measurements were made on site using a non-destructive non-intrusive method called "Electrical resistance Tomography" (ERT) which is an indirect mapping of saturation (Moreau *et al.*, 2003; Grellier *et al.*, 2003).

2. DESCRIPTION OF THE MODEL

2.1 Mass conservation and energy equations

The landfill is considered as a homogeneous and anisotropic three-phase porous medium. The three phases are solid waste, gas and leachate, where gas is considered as a mixture of CO2 and CH4 with equal percentages. The solid phase of the landfill is considered to be non-deformable with no consolidation which means that we just consider the effects of degradation in gas and heat production and not in deformation and settlement of the landfill. The gas and liquid phases are considered to be immiscible and Darcy's law is applicable for both fluid phases. We assume that there is a thermal equilibrium between the three phases and thermal radiation is neglected. Using Darcy's law, the mass conservation equation for each phase writes:

$$\frac{\partial m_P}{\partial t}+\nabla.\left[\rho_P \frac{k_{rP}k}{\mu_P}(-\nabla P_P+\rho_P g)\right]=\alpha_{bP}, \qquad (1)$$

where $m_l=\phi S \rho_l$, $m_g=\phi(1\text{-}S)\rho_g$ are the masses of phase $P=l,g$ per porous volume unit (kg/m^3), α_b is the production rate for each phase, S is the liquid saturation and ϕ and k are the porosity and absolute permeability of the porous medium (landfill waste). ρ_P, k_{rP} and P_P are the density, relative permeability and pressure for phase p. The gas density is defined by the perfect gas law, $\rho_g=\frac{M_g P_g}{RT}$, where M_g is the molecular mass of the mixture of methane and carbon dioxide and R is the universal gas constant. The gas pressure is defined as the sum of the liquid and capillary pressure.

For the capillary pressure and relative permeability, we have used the Van Genuchten's model (1980) with the parameters which are proposed for landfilled municipal waste (Aran, 2000; Kling *et al.,* 2006). The energy equation can be written as followed:

$$T\frac{d\eta}{dt}+\sum_{p=l,g} g_p \frac{dm_p}{dt}=-\nabla.(h_p\rho_p V_p)-\nabla.q+\alpha_q, \qquad (2)$$

V_p, $h_P(P_P,T)$, q, g, T and α_q are respectively the Darcy's velocity of phase p, the enthalpy per mass unit of phase p, the conductive heat flux, the gravity acceleration, the temperature and the heat source terms. g_p is defined as: $g_P(P_P,T)=h_P(P_P,T)-T\eta_P(P_P,T)$. $\eta_P(P_P,T)$ is the entropy per mass unit of phase p.

2.2 Biodegradation model

Municipal solid waste is composed of different types of materials characterized by different substrate utilization rates by the micro-organisms, so the total gas generation rate is estimated as the sum of the rates of gas generation from the individual refuse components. The biodegradation rate could be represented by Monod's classic model in which water content of each component is considered to be constant. We modified this model using an empirical function of saturation, $f(S)$ of bacterial activities. The rate of degradation can be written as: $\frac{dA_i}{dt} = A_i.\lambda_i(T).f(S)$, A_i is the fraction of each component (i=1: rapidly biodegradable, i=2: fairly biodegradable and i=3: slowly biodegradable) and $\lambda_i(T)$ is the degradation kinetics for each component A_i which is defined by Arrhenius Law: $\lambda_i(T) = \beta_i \exp(-Ea_i / RT)$. Here $\beta_i(\sec^{-1})$ is a constant and Ea_i is the activating energy of each component.

Biogas production is defined by the exponential law, proposed by Halvadakis (1983) which is principally defined for the anaerobic phase of degradation (Findikakis *et al.* 1987): $\alpha_b = C_{Tb}\frac{dA_i}{dt}$. C_{Tb} is the potential biogas production and α_b is the rate of biogas production. The heat production rate is obtained by the relation: $\alpha_q = H\frac{1}{2M_b}\alpha_b$. H is the energy released for each mole of methane which is produced during degradation and M_b is the molar mass of biogas.

2.3 Modelling parameters

Different values are proposed in the literature for the porosity and hydraulic conductivity of MSW. The choice of values of porosity and hydraulic conductivity is based on the estimations from the site data and resistivity charts, considering the range of values proposed by different authors (Korfiatis *et al.*, 1984; Oweis *et al.*, 1990; Bleiker *et al*, 1993; Beaven *et al.*, 1995; Lanini, 1998; McCreanor *et al.*, 2000; Hudson *et al.*, 2001 and Franck *et al.*, 2006). The hydraulic parameters are chosen as: a porosity of 15%, a hydraulic conductivity equal to 5.0E-5 (m/sec) and a residual saturation equal to 0.35. For Van Genuchten parameters we have $m = 1-(1/n) = 0.8718$ and $\alpha / \rho g = 10E-4(Pa)$.

2.3.1 Saturation function

The saturation function is introduced into degradation kinetics, $\lambda_i(T,S)$ to make it possible to consider the saturation changes during leachate recirculation in the biodegradation model. Definition of the saturation function is based on existing empirical knowledge: Below a minimum saturation which is essential for bacterial activities, degradation and biogas production are inhibited. Biogas production increases with saturation to reach its maximum value near the saturation at field capacity until the complete saturation. The saturation function which is used in our model is based on the earlier model of Gil Diaz *et al*. (1995).

3. NUMERICAL TECHNIQUES

The numerical solution of the mathematical model is obtained using a finite volume method, which consists in a set of nonlinear discrete balance equations in grid blocks, coupled with the conservation equations. This system is solved using Newton's method (Benard *et al.* 2005; Eymard *et al.* 2000). The advantage of this method over other methods for solving the conservation equations is that the non structured grid blocks could be solved as well as the structured ones and there is flux conservation between two control volumes K and L, which can be expressed by this relation: $F^n_{K,L} = -F^n_{L,K}$. Considering a finite volume mesh of the domain, consisting of N grid blocks indexed by (i =1, . . . ,N), the volume of the cell i is denoted by V_i. The subscript j stands for any cell having a common interface $i|j$ with the cell i. Equations 1 and 2 give, after time discretization and finite volume discretization, a set of coupled nonlinear equations (Eymard et al., 2000):

$$\begin{cases} V_i \dfrac{m_{Pi}^{n+1} - m_{Pi}^{n}}{t^{n+1} - t^{n}} + \sum\limits_{i|j} F_{wij}^{n+1} = 0 \\ V_i \left[T_i^{n+1} \dfrac{\eta_{hi}^{n+1} - \eta_{hi}^{n}}{t^{n+1} - t^{n}} + \sum\limits_{P=l,g} g_{P_i}^{n+1} \dfrac{m_{Pi}^{n+1} - m_{Pi}^{n}}{t^{n+1} - t^{n}} \right] + \sum\limits_{i|j} F_{hij}^{n+1} = \overline{Q}_i^{n+1} \end{cases} \quad (3)$$

where $\overline{Q}_i^{n+1}$ is the heat source term in grid block i. The water flux F_{wij}^{n+1} and the energy flux F_{hij}^{n+1} across the interface $i|j$ are evaluated using an implicit finite difference scheme with respect to the pressures, the saturations and the temperatures of grid blocks i and j. For each time step, we applied an adapted Newton method to find an approximate value of thermodynamic and hydraulic unknowns of the whole system of equations. This method, classically used in the oil reservoir simulation setting (Aziz *et al.*, 1979), appears to be very stable and efficient for solving the conservation equations of mass and energy.

4. ANALYTICAL VALIDATION OF THE NUMERICAL METHOD

An analytical solution is used to validate the two-phase flow model. This analytical solution is based on a one dimensional linear fractional flow, described by Buckley-Leverett's one dimensional equations (Willhite, 1986). The important assumptions for the analytical solutions are: one dimensional linear flow, the fluids are incompressible and immiscible, the porous medium is not deformable, the displacements are based on Darcy equations and finally the capillary pressure is neglected. To validate our model we consider a vertical model of gas and water flow with gravity force. Applying Darcy's law for linear flow of water and gas with an injection flow rate q, the vertical fractional flow of water is expressed as:

$$f_w(S) = \frac{\left(\dfrac{k_{rl}(S)k}{\mu_l} . \dfrac{k_{rg}(S)k}{\mu_g} \Delta\rho g \right) + q \dfrac{k_{rl}(S)k}{\mu_l}}{\dfrac{k_{rl}(S)k}{\mu_l} + \dfrac{k_{rg}(S)k}{\mu_g}} . \quad (4)$$

A vertical one dimensional column is modelled to be compared with the analytical solution. Half of this column is fully saturated with water and the other half has saturation equal to 0.7 (see Figure 1). The porosity of the porous medium is 20%, the absolute permeability is $k = 1.0e^{-11}(m^2)$ and a residual saturation equal to 0.7 is considered to be used in Van Genuchten's equations for relative conductivities of water and gas. The maximum saturation is equal to 0.97, above which the mobility of gas is inhibited. Other Van Genuchten parameters are presented in section 2.3. An atmospheric pressure is applied at the left and there is no injection.

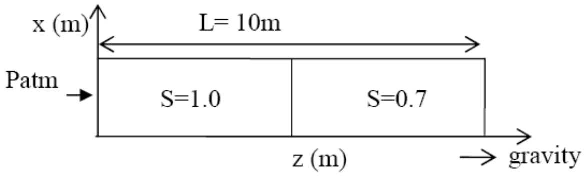

Figure 1. Dimensions and initial conditions for saturation in the one dimensional column

Using Equation 4, considering $q = 0$, the concave hull of fractional flow, $\hat{f}(S)$, could be plotted as presented in Figure 2 (left). The derivation of the concave hull is denoted as $\hat{f}'(S)$, which is presented in Figure 2 (right).

From the concave hulls, the saturation of the shock fronts is obtained as $S_1^* \approx 0.9204$ and $S_2^* \approx 0.9060$, respectively for the first and second shock. The speed of propagation of the shocks is determined by Rankine- Hugoniot condition:

$$f'(S_1^*) = \frac{[f(1)=0] - f(S_1^*)}{1 - S_1^*} \quad \text{and} \quad f'(S_2^*) = \frac{f(S_2^*) - [f(0.7)=0]}{S_2^* - 0.7}.$$

The values are found as -1.4e^{-3} and 5.64e^{-4} (applying the porosity of the porous medium), respectively for the first and second shocks. Knowing the saturation of the two shocks the profile of the analytical solution is obtained, as presented in Figure 2 (right).

The displacement of the two-phase flow can be plotted in time using the speed of propagation of each shock. The numerical and analytical results are presented in Figure 3 for different steps of time. The analytical solution is plotted in full lines. Figure 3 shows that there is an agreement between numerical and analytical results for a vertical model of two-phase flow.

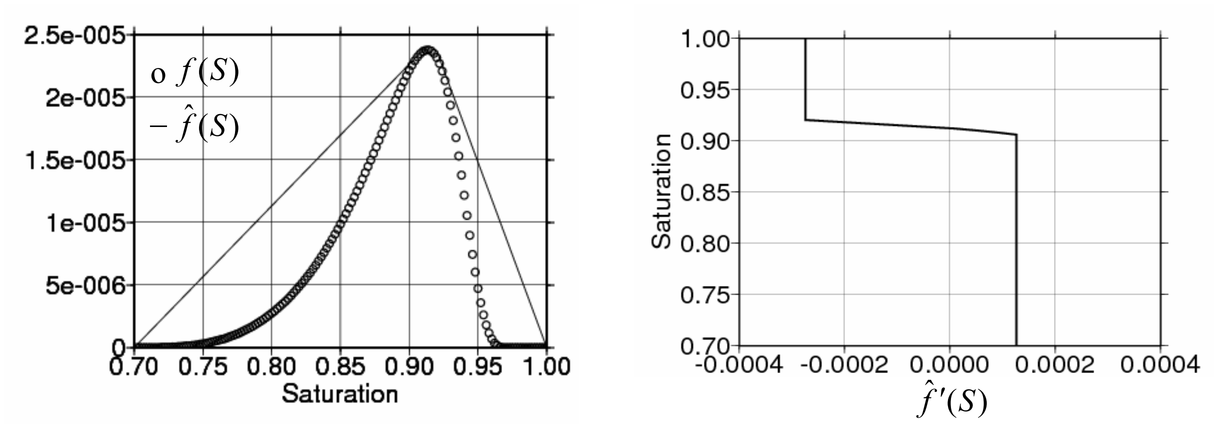

Figure 2. Concave hull of fractional flow (left) and Profile of the analytical solution (right)

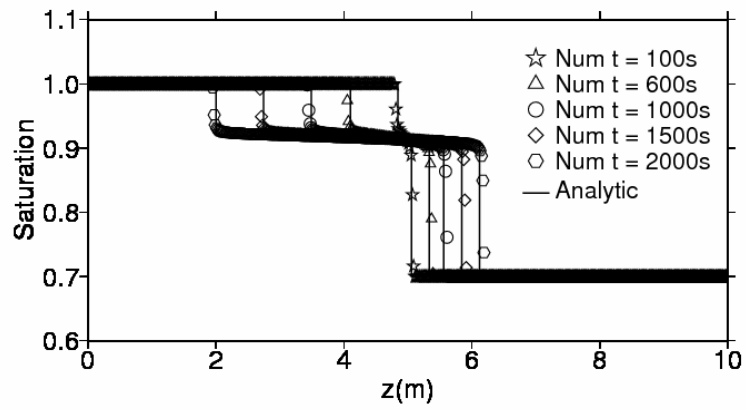

Figure 3. Numerical and analytical results for a two-phase flow

5. COMPARISION BETWEEN NUMERICAL AND IN-SITU RESULTS

5.1 Description of the site

The landfill site includes a 5000m^2 monitored bioreactor cell. A leachate recirculation network has been designed in order to have a regular leachate distribution through horizontal drains under the cell cover. The monitored cell is 6-12m in depth, with five leachate recirculation lines. The last injection operation was conducted with a one-week interval. The injection parameters are presented in Table 5.1 for two injection operations.

5.2 Simulation results and discussion

We have used a homogeneous two-dimensional 90m x 12m model to study the hydrodynamical behaviour of the bioreactor site during leachate injection and recirculation, with a zoom on the injection point. Boundary conditions are: zero-flux condition on top horizontal boundary except an injection point at the same location as on site; zero-flux condition on vertical boundaries and atmospheric pressure (perfect drainage) on bottom horizontal boundary. Discharge flow and injection duration are the same as the values in Table 5.1. Initial conditions are atmospheric pressure and a temperature of 33°C, according to the site results.

5.2.1 Initial saturation

As there is no way to measure the saturation of different parts and layers of a landfill without disturbing the site, we have decided to use electrical resistivity measurements which is a function of saturation in order to have an estimation of the initial saturation. The non-destructive method to measure the electrical resistivity is the « Electrical Resistance Tomography » (ERT) method which consists in carrying out series of measurements by commutating the electrodes, placed on the site preliminarily (Grellier 2005).

Table 5.1 Injection parameters

Parameters	First recirculation	Second recirculation
Injection duration	4h	3h10
Discharge (m^3/hr)	6.0	11.0
Outdoor Temperature (°C)	2.6	10.7

The results of the electrical resistivity measurements during leachate injection are presented in Figure 4, in terms of differential resistivity from a reference time before leachate injection. The negative values show the reduction of the values comparing to the reference time. This reduction can be explained by an increase of saturation. A number of numerical simulations were undertaken with different initial saturations and anisotropy coefficients between horizontal and vertical permeability in order to approximate to the same dimensions of the resistivity bulbs (about 10m wide and 9-10m long).

The results presented in Figure 4 are for an initial saturation of 45% and anisotropy of 0.02 (Horizontal/Vertical), which matches the best with electrical resistivity results. This estimation is based on the assumption that there is a direct and fairly linear relationship between electrical resistivity and saturation of the landfill which is not completely true. However this assumption makes it possible to have an estimation of the initial saturation value which can not be easily defined, without disturbing the site.

As our model is homogenous, we can not see the heterogeneity effect and preferential flows in the saturation results which are produced by the components with different permeability. This is why comparing to the resistivity results we have a complete gravitational flow without any inclination. Comparing the dissipation kinetics of resistivity and saturation bulbs between second leachate injection day and two days after, we observe 35% reduction in electrical resistivity from -50% to -15% which could be compared to 28% of saturation dissipation from 94% to 66%. Hereafter we will study the important role of porosity and hydraulic conductivity on the hydrodynamical results.

5.2.2 Porosity

The choice of porosity for the simulation is based on the pressure values of the injection point. Figure 5 shows the ratio of the pressure of the injection point to the atmospheric pressure during and after leachate injection. We can see that during the second injection the pressure of the injection point exceeds the atmospheric pressure for the smaller porosities, which means that a backpressure is needed to inject the leachate. On the other hand we know that the leachate injection on the landfill site is under gravitational forces and there is no backpressure for injection. So considering the pressure of the injection point and the dimensions of the resistivity bulbs a value of 15% is chosen for the bioreactor model.

5.2.3 Hydraulic conductivity

The numerical results for three values of hydraulic conductivity, 5.0e-4, 5.0e-5 and 5.0e-6, show that for the higher values of this parameter, most of the injected leachate moves rapidly downward the landfill, the injection point being under atmospheric pressure. On the other hand with the smaller values of hydraulic conductivity the injected water remains very near to the injection point even after two days of injection and water pressure is always higher than atmospheric pressure. The pressure at the injection point can reach more than twice the atmospheric pressure for a discharge of $11m^3$/hr.

As the real landfill is an anisotropic medium in which the hydraulic conductivity and porosity changes with depth from the top layer, a numerical modelling was carried out to consider the changes of these values as a function of depth. The porosity and hydraulic conductivity change respectively from 0.2 and 5.0e-4 at the top to 0.1 and 5.0e-6 at the bottom of the model. Figure 6 shows that a model with variable hydraulic conductivity and porosity leads to less saturated bulbs which are dissipated faster than the model with average values. All the pressure ratios are smaller than one for the model with variable hydraulic conductivity and porosity.

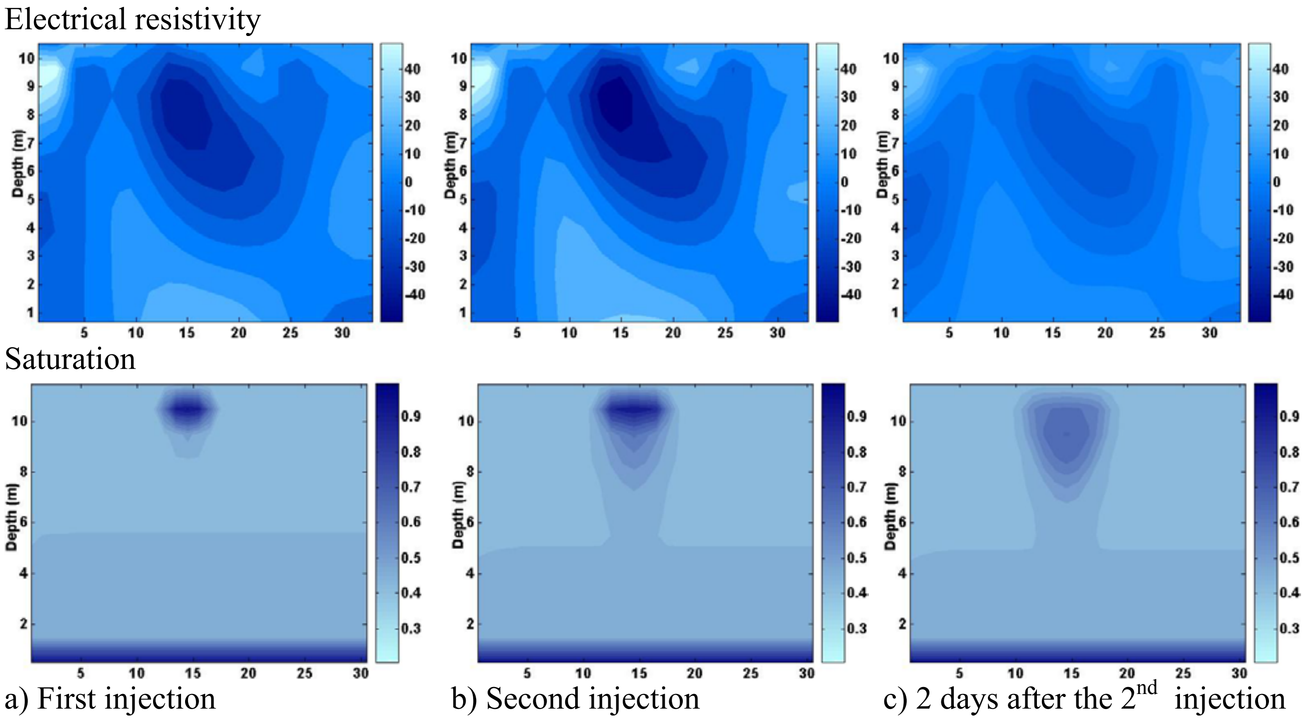

Figure 4. In-Situ and numerical modelling results for electrical resistivity and saturation during leachate recirculation

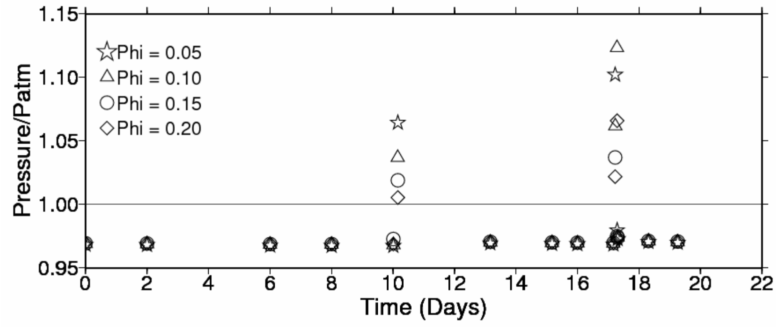

Figure 5. Ratios between pressure of the injection point and atmospheric pressure for different porosity values

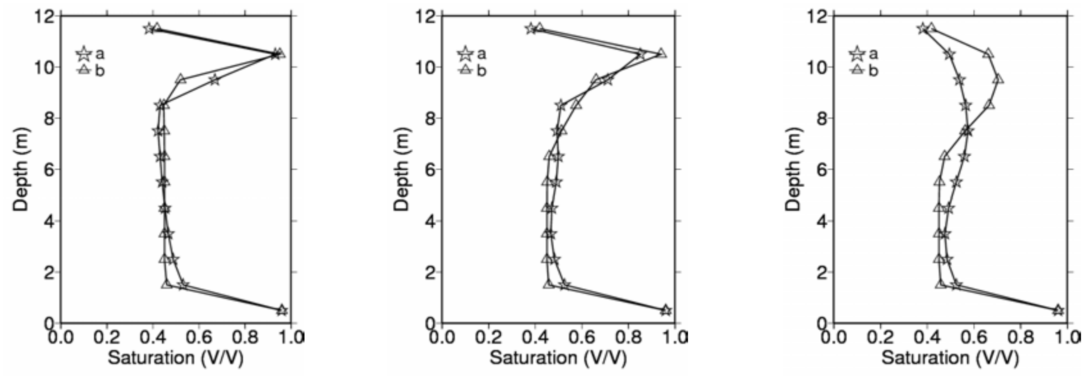

Figure 6. Comparison of saturation results of a) a model with variable hydraulic conductivity and porosity and b) homogeneous model with $\phi = 0.15$ and $Hyd - con = 5e - 5$

6. CONCLUSIONS

In this paper we have presented a numerical model of two-phase flow, applied to an anaerobic bioreactor landfill with leachate recirculation. This model is successfully validated by analytical solutions based on the Buckley-Leverett equations and Darcy's law, which is an advantage comparing to the models based on Richard's equations. The results of numerical simulation show that our model is able to reproduce changes and dissipations of saturation in a bioreactor landfill during leachate recirculation. Using this model to reproduce site results leads to find out that initial saturation is a very important parameter with a remarkable effect on the dimension of saturated zone around the injection point. Knowing this parameter in a bioreactor site leads to optimise the volume of injected leachate to reach the field capacity, where the biodegradation is optimal. Our results show that the injection process is more efficient in higher initial saturation values. This model is also very sensible to the changes of hydraulic conductivity and porosity, which are important parameters in hydrodynamical behaviour of a landfill. Changes of these values could lead to excessive pressure at the injection points. Considering that leachate injection on the real site was done under gravitational force without any backpressure, pressure values at the injection point lead to a better choice of porosity and hydraulic conductivity values. This model shows that hydraulic conductivity of the landfill has a very important effect on the behaviour of the saturation zone. The bigger values of this parameter lead to a faster drainage of the injected leachate and a smaller (in width) and less saturated bulb. In the other hand the smaller values of hydraulic conductivity lead to wider and more saturated bulbs. This is very important to estimate the influence zone of a volume of leachate which is injected into the landfill to optimize the location and number of the injection lines and their distances from each other. The next step of this research is to validate the model by additional field and laboratory experiments, especially for the temperature values and its changes during leachate recirculation. The production of biogas should be also compared to the field data and finally sensibility analyses are needed to define more precisely the hydrodynamical and biological parameters of heat and biogas production.

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