

SIMULATING LANDFILL AERATION SYSTEMS USING COMPUTATIONAL FLUID DYNAMICS TECHNIQUES

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Abstract. *The present study is an application of Computational Fluid Dynamics (CFD) on the numerical modeling of aeration systems of uncontrolled landfills. More specifically, the CFD algorithms provided by the commercial solver ANSYS Fluent 14.5, combined with an in-house source code developed to complement the main solver was used. Coupling the multiphase flow with a biochemical algorithm, the complex biochemical process of organic solid waste stabilization using aeration was simulated. Air was considered to enter the solid waste region which was partially saturated, using aeration wells. Multiphase flow both for the air and liquid phase was simulated taking into consideration the oxygen (O₂) transfer across phases. Unsaturated flow effects, such as capillary pressure and unsaturated hydraulic conductivity, were taken into consideration during the simulation using proper functions that were included in the main solver. After the initial model set up, a sensitivity analysis for basic biochemical parameters was implemented, examining the effect of various kinetic parameters variation. Moreover, pilot scale simulations were performed. The results obtained were in good agreement with corresponding previous experimental and computational results of the literature. Finally, a case study was simulated for a hypothetical landfill aeration well.*

1 INTRODUCTION

Landfill aeration is the process in which air is inserted in the mass of solid waste, usually via passive or forced aeration wells, in order to reduce its biodegradable organic load. During last decades, aeration systems have been globally used as a quick, effective and low-emission technology for the aerobic biological stabilization of landfills [1]. Although many studies have reported the successful use of this method presenting the results of its application in the field [1-7], knowledge gaps for the effective design of these systems still remain due to the fact that its design is still based on preliminary pilot-scale field experiments and former practical experience [8]. Cossu et al. [7] tried to use flow modeling, combined with corresponding pilot experimental data, in order to define the radius of influence of aeration wells in the field.

Numerous models are available in the literature concerning the modeling of aerobic biodegradation processes of organic solid waste, however only few of them are time and space integrated by means of being able to capture the space and time variation of the examined quantities. Moreover, most of these studies do not refer to landfill aeration, investigating other aerobic processes, such as composting, which is a similar process for organic solid waste treatment and stabilization. Thus, previous experience can be used for the assessment of the method of landfill aeration. This is very important due to the lack of an extensive research on biodegradation process kinetics of landfill aeration in the literature.

Biochemical kinetics of aerobic biodegradation processes, such as composting and landfill aeration are affected by various environmental parameters as [9, 10]: (1) temperature, (2) moisture, (2) oxygen concentration, (3) free air space and (4) pH. Extensive reviews are available in the literature [9-11] concerning the used of different kinetics for the description of aerobic biodegradation of organic solid waste as well as for the correction factors used in order to take into consideration the effect of the aforementioned environmental parameters.

Only a few simulation models in the literature describe temporal and spatial variations of aerobic degradation of organic solid waste as well as the flow field properties. However, Bari and Koenig [12, 13], using a multi-layer (1-D) model and experimental investigations for a forced aeration composting process, proved that this kind of spatial variations for the temperature and biodegradable volatile solid fields really exist. El-Fadel et al. [14-16], in their work, presented one of the first temporally and spatially dynamic model for the simulation of the generation and transport of gas during the anaerobic biodegradation of organic solid waste in a landfill. The “Landfill Degradation And Transport” of the Waste Management Research Group of the University of Southampton (UK) [17] is one of the first models that have been used in order to simulate the biodegradation of organic solid waste for both anaerobic [18, 19] and aerobic [20, 21] conditions. The LDAT model combines a multi-component biodegradation model with a fluid mechanics solver for the flow field calculations.

The present study is an application of Computational Fluid Dynamics (CFD) on the simulation of organic solid waste stabilization using aeration systems. A multiphase model is used in order to simulate the flow of gaseous and liquid phases. Unsaturated flow effects, such as capillary suction and hydraulic conductivity variation due to moisture degree saturation, were also taken into consideration. Moreover, oxygen (O_2) transport in both liquid and gaseous phases was simulated using proper advection-dispersion-reaction equations whilst the oxygen mass transfer across the phases due to its concentration gradient was also simulated. Energy equations were solved in order to calculate the temperature field as well as the heat generation due to the organic solid waste biodegradation and the other biochemical processes, which were included as source/sink terms in the energy conservation equations. Finally, the concentrations of the biodegradable organic solid waste and biomass were calculated solving proper biochemical kinetics, which were coupled with the main fluid flow solver. For the solution of the aforementioned equations, the commercial

solver ANSYS Fluent 14.5 [22] was used combined with an in-house source code which includes the biochemical model to simulate the organic solid waste stabilization and biomass production, as well as some additional terms of the flow equations that alter the main commercial CFD solver.

2 MATHEMATICAL MODEL

For the simulation of multiphase flow during landfill aeration, the Control or Finite Volume approach was used which is included in the commercial solver ANSYS Fluent 14.5, combined with an in-house code which was developed in ANSI C programming language and the "User Defined Functions" (UDFs) macros, which are included in the main CFD solver. The developed model aims to: (1) simulate the air-water flow in the unsaturated porous media of solid waste, (2) simulate the oxygen transport in both the phases (air and water) modeling the oxygen mass transfer across the interphase of the phases, (3) simulate the temperature variations due to the heat production and consumption due to the biochemical processes of the model and (4) simulate concentration of the organic solid waste and of the biomass using proper biochemical kinetics combined with some correction factors which include the effect of the environmental parameters, such as the temperature and the oxygen concentration on the biochemical reaction rates.

More specifically, our in-house subroutines define the capillary pressure effect for the water momentum equation, calculate the hydraulic conductivity of each phase as a function of moisture saturation, define the transport mechanism for the oxygen mass transfer across phases as well as the sink/source term due to the biochemical reaction of the organic solid biodegradation, calculate the temporal and spatial variations of the biodegradable organic solid waste and biomass including the effect of environmental factors and define proper source/sink terms in the energy equations, in order to estimate the temperature field and variation for each phase.

2.1 Flow equations

Both phases (air and water) were assumed to follow the Navier-Stokes-Brinkman equations for the momentum conservation in each CV which is an extended Darcy's model for an "Euler-Euler" multiphase approach [22] used for unsaturated flow:

$$\frac{\partial \varepsilon \alpha_q \rho_q \bar{V}_q}{\partial t} + \nabla \varepsilon \alpha_q \rho_q \bar{V}_q \bar{V}_q = -\varepsilon \alpha_q \nabla P + \nabla \varepsilon \bar{\tau} + \varepsilon \alpha_q \rho_q \bar{g} - \alpha_q^2 \frac{\mu_q}{k k_r} \bar{V}_q - \varepsilon \alpha_q \nabla P_c \quad (1)$$

where ε is the total porosity of the medium [dimensionless], α_q is the volume fraction of q phase [m^3/m^3], ρ_q is the density of q phase [kg/m^3], \bar{V}_q is the Darcy velocity of q phase [m/sec], P is the pressure [Pa], τ shear stress [Pa], g is the acceleration of the gravity [m/sec^2], μ_q is the dynamic viscosity of q phase, k is the intrinsic or saturated permeability of q phase [m^2], k_r is the relative permeability [dimensionless] and P_c [Pa] is the capillary pressure. Relative permeability was calculated using the van Genuchten formulation [23]:

$$k_r = S_e^{0.5} (1 - (1 - S_e^{1/\gamma})^\gamma)^2 \quad (2)$$

for the aquatic phase and:

$$k_r = (1 - S_e)^{0.5} (1 - S_e^{1/\gamma})^{2\gamma} \quad (3)$$

for the gaseous phase, where S_e is the saturation calculated as:

$$S_e = \frac{a_w - a_{wr}}{a_{ws} - a_{wr}} \quad (4)$$

where a_w is the volume fraction of water, a_{wr} is the residual saturation, a_{ws} is the maximum saturation, and α [l/m], β and γ are the van Genuchten constants, where $\gamma = 1 - \frac{1}{\beta}$. The term

$\gamma a_q \nabla P_c$ for the capillary pressure effect was included only in the wetting phase and was calculated using the van Genuchten [23] – Mualem [24] formula:

$$P_c = -\frac{\rho_w g}{\alpha} \left(\left(\frac{1}{S_e} \right)^{1/\gamma} - 1 \right) \quad (5)$$

where ρ_w is the liquid phase density [kg/m^3].

A continuity equation was solved for the air phase:

$$\frac{\partial \varepsilon a_q \rho_q}{\partial t} + \nabla \varepsilon a_q \rho_q \vec{V}_q = \varepsilon S_q \quad (6)$$

where S_q is the mass source/sink term, whilst water volume fraction was calculated using the equation for the sum of volume fraction of all phases (that equals unit):

$$\sum_{i=gas,liquid} a_i = 1 \quad (7)$$

Oxygen mass transfer was simulated using a standard advection-dispersion-reaction equation for each phase [22]:

$$\frac{\partial \varepsilon a_q \rho_q C_p^q}{\partial t} + \nabla \varepsilon a_q \rho_q C_p^q \vec{V}_q = -\varepsilon a_q \rho_q \nabla \vec{J} + S_p \quad (8)$$

where C_p^q is the oxygen concentration in q phase [kg/kg_q], S_p is the sink/source term of oxygen mass due to its consumption due to organic solid waste biodegradation as well as due to its mass transfer across the phases' interface and \vec{J} the dispersion term calculated as:

$$\vec{J} = -D \times \nabla C_p^q \quad (9)$$

where D is the dispersion coefficient [m^2/sec].

The reason of the oxygen mass transfer across air and water is its concentration gradient across them, namely the difference between the real and the equilibrium oxygen concentration which was calculated, using Henry's Law:

$$H_c(T) = \frac{C_p^{eq_l}}{C_p^{eq_g}} \quad (10)$$

where $H_c(T)$ is the dimensionless form of Henry's constant, $C_p^{eq_l}$ is the equilibrium concentration of p species in water (*liquid* phase), $C_p^{eq_g}$ is the equilibrium concentration of p species in air (*gaseous* phase). Henry's constant can be calculated according to the following equation [25]:

$$H_H(T) = H_H^{T_o} \cdot \exp(-\alpha_H (\frac{1}{T_o} - \frac{1}{T})) \quad (11)$$

where $H_H^{T_o}$ is the dimensioned Henry's constant [M/atm] at T_o [$^{\circ}K$] temperature, α_H is the constant of exponential decrease [$^{\circ}K$] and T is the Temperature [$^{\circ}K$]. In the present study, α_H equals 1800 $^{\circ}K$ [25] while $H_H^{T_o}$ for O_2 equals 1.30×10^{-3} M/atm at 25 $^{\circ}C$ [25].

Thus, oxygen mass transfer rate between the phases' interphase was calculated as [26, 27]:

$$R_{pq_p} = a_p \rho_p K_{pq_p} (C_p^{eq_p} - C_p^p) \quad (12)$$

where R_{pq_p} is the mass flux of p species from phase q to phase p (with $R_{pq_p} = -R_{qp_p}$), a_p is the volume fraction of phase p [m^3/m^3], ρ_p is the density of p phase [kg/m^3], K_{pq_p} is the mass transfer rate of p species from phase q to phase p [1/sec], C_p^{eqp} is the equilibrium concentration of p species in phase p [kg_C/kg_p] and C_p^C is the concentration of p species in the phase p [kg_C/kg_p].

Finally, for each phase an energy equation was solved:

$$\frac{\partial \varepsilon a_q \rho_q h_q}{\partial t} + \nabla \varepsilon a_q \rho_q h_q \vec{V}_q = -\varepsilon a_q \rho_q \frac{\partial P}{\partial t} + \varepsilon \tau_q : \nabla \vec{V}_q - \nabla \varepsilon q_q + S_q \quad (13)$$

where h_q is the specific enthalpy of q phase [J/kg_q] and S_q is the sink/source term of energy which includes the energy production/consumption due to biochemical processes, calculated based on the enthalpies of Table 1 and the reaction rates of the biochemical reaction (see paragraph 2.2) and the heat transfer across phases which is using the Ranz and Marshal's model [28, 29] for the estimation of the volumetric heat transfer coefficient [W/m^2] between the phases.

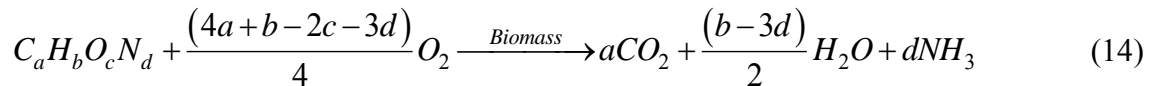
| a/a | Species | Symbol | Molecular Weight | Enthalpy of formation ¹ | Reference |
|-------|----------------|--------------------------|------------------|------------------------------------|-----------|
| 1 | Solid Waste | $C_6H_9O_{3.56}N_{0.32}$ | 142.44 | -15073.4 | [17] |
| 2 | Oxygen | O_2 | 32 | -365.94 | [30] |
| 3 | Carbon dioxide | CO_2 | 44 | -9528.64 | [31] |
| 4 | Water | H_2O | 18 | -15879.40 | [30] |
| 5 | Ammonia | NH_3 | 17 | -4722.94 | [30] |
| 6 | Ammonium ion | NH_4^+ | 18 | -7361.11 | [30] |
| 7 | Hydrogen ion | H^+ | 1 | 0.0 | [30] |
| 8 | Biomass | $C_5H_7NO_2$ | 99 | -13790.9 | [17] |
| 9 | Glucose | $C_6H_{12}O_6$ | 180 | -7061.11 | [31] |

¹ in kJ/kg

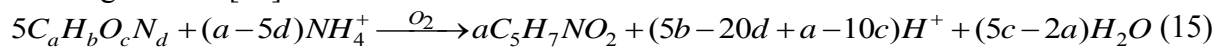
Table 1: Molecular Weight and enthalpy of formation for the used species.

2.2 Biodegradation kinetics

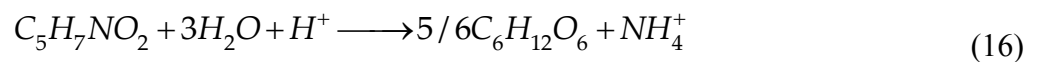
The system of biochemical equations taken into consideration in the model is presented below, starting with the chemical equation for the biodegradable solid waste [32]:



where a , b , c and d are constants referred to waste chemical composition. The chemical equation for bacteria growth is [20]:



The chemical equation for the biomass death is [20, 33]:



For the purpose of the present study, the chosen values for these constants were 6, 9, 3.56 and 0.32 [34].

Two kinds of biochemical kinetics were used in the present study in order to estimate the biochemical reaction rate of equation (14): (1) Monod kinetics and (2) first-order kinetics. The Monod equation is [10, 14-16, 20]:

$$S_s = \frac{dC_s}{dt} = -k_m \frac{C_s}{K_s + C_s} C_B \quad (17)$$

where S_s is the solid waste biodegradation rate [$kg/m^3 \cdot sec$], C_s is the concentration of the biodegradable solid waste [kg/m^3], t is the time [sec], k_m is the maximum biodegradation constant [sec^{-1}], K_s is the half saturation constant for the solid waste [kg/m^3] and C_B is the biomass concentration in the waste matrix [kg/m^3]. The first order kinetics is expressed as [9, 10]:

$$S_s = \frac{dC_s}{dt} = -k' C_s \quad (18)$$

where k' is the effective/corrected biodegradation rate [sec^{-1}].

The biochemical reaction rates of equations (14) and (15) were coupled using the concept of the yield coefficient according to which:

$$S_s = \frac{dC_s}{dt} = \frac{S_B}{Y_s} = \frac{1}{Y_s} \frac{dC_B}{dt} \quad (19)$$

where S_B is the biomass production rate [$kg/m^3 \cdot sec$] and Y_s is the yield coefficient which connects kinetics of biodegradable solid waste and biomass [$kg_B m^3 \cdot sec / kg_S m^3 \cdot sec$].

As it was mentioned in the introduction, temperature, moisture, oxygen concentration, free air space and pH are the main environmental parameters that alter the biochemical reaction rate of organic solid waste biodegradation. Thus, correction factors for temperature and oxygen were used for both the Monod and the first-order kinetics. Although correction factors for the free air space and the moisture were not applied, the volume fraction of the examined fluids (water and air) change during the simulation as the fluids move and thus alter the concentration of oxygen and the temperature field including that effect in the model. Finally, pH correction factor is not considered in the present study.

In order to simulate the effect of temperature on biochemical kinetics of the organic solid waste biodegradation, the cardinal temperatures correction factor of Rosso et al. [35] was used:

$$k_{temp} = \frac{(T - T_{max}) \times (T - T_{min})^2}{(T_{opt} - T_{min}) \times [(T_{opt} - T_{min}) \times (T - T_{opt}) - (T_{opt} - T_{max}) \times (T_{opt} + T_{min} - 2T)]} \quad (20)$$

where T_{min} , T_{max} and T_{opt} are the cardinal temperatures, namely the minimum, maximum and optimal temperatures for the aerobic biodegradation [$^{\circ}C$ or $^{\circ}K$] and T is the temperature [$^{\circ}C$ or $^{\circ}K$].

The corresponding correction factor for the oxygen concentration was assumed to be [10, 36]:

$$k_{O_2} = \frac{C_{O_2}}{K_{O_2} + C_{O_2}} \quad (21)$$

where C_{O_2} is the concentration of oxygen [kg/m^3] and K_{O_2} is the half saturation constant for oxygen [kg/m^3].

Finally, the following equation was used for the estimation of biomass decay rate, S_D , of equation (16):

$$S_D = k_d k_{temp2} C_B \quad (22)$$

where k_d is the bacterial decay rate in $[sec^{-1}]$ and k_{temp2} is the correction factor of temperature for the decay rate, calculated according to the following equation [37]:

$$k_{temp2} = 2.142 \times 10^{-4} T^2 - 2.356 \times 10^{-2} T + 1.348 \quad (5^\circ C < T < 75^\circ C) \quad (23)$$

2.3 Model assumptions and solution methods

All the above equations were solved with the assumptions that flow and mass transfer in the landfill can be described as a porous media flow, according to equations of paragraph 2.2. Solid waste porous media was considered to be rigid (not-deformable), flow was considered to be incompressible, fluids and porous media were assumed to be in thermal equilibrium, biomass and biodegradable solid waste was considered to be in fixed position and biodegradation was considered to take place in the liquid phase.

For the solution of the aforementioned equations, the commercial solver ANSYS Fluent 14.5 was used. More specifically, the used solver was "pressure based" with first-order numerical schemes for the spatial and temporal discretization while velocity-pressure coupling was achieved using the "phase coupled SIMPLE" algorithm [22]. All the equations for the biochemical process were solved implicitly, while the coupling with the flow equation was achieved using sink/source terms. The convergence criterion for the calculations was set at 10^{-12} .

3 RESULTS

The research group of the University of Southampton examined the solid waste aeration in a rectangular container (24 m long, 8 m wide and 3 m deep) [17, 20, 21]. This geometry is reproduced in two dimensions (Figure 1). Air was assumed to enter the container through a 2 m inlet at the bottom of the geometry (red line of Figure 1), which was simulated as a "pressure inlet" boundary condition whilst the top of the geometry was considered open ("pressure outlet", blue line of Figure 1). Finally walls of the container were considered impermeable ("non-slip wall" boundary condition, black line of Figure 1).

The chemical formula of the examined organic solid waste was assumed to be $C_6H_9O_{3.56}N_{0.32}$ [34]. Concerning the hydraulic properties of solid waste, the intrinsic permeability and porosity of solid waste were assumed to be $10.23 \times 10^{-11} m^2$ and 0.50, van Genuchten's constants α , β , and γ were assumed to be 0.5, 2 and 0.8 respectively [17] and the relative saturation of the waste was assumed to be 0.20 [38]. Moreover, the used values for the biochemical modeling are available at Table 2.

The relative pressure at the bottom of geometry (inlet) was gradually increased from 40 Pa at the beginning of the simulation to 500 Pa at $t = 0.4$ days. The simulation period was set at 1.5 days. Pressure field variation over time is presented in Figure 2 for $t = 1, 3, 6, 9$ and 12 h, while directions of velocity vectors at steady state (after $t = 0.4$ days) are shown in Figure 3. The ability of the present model to capture time-dependent boundary conditions can be useful during hybrid aeration process simulation, in which wells alter their operation during their use acting as positive/negative pressure wells.

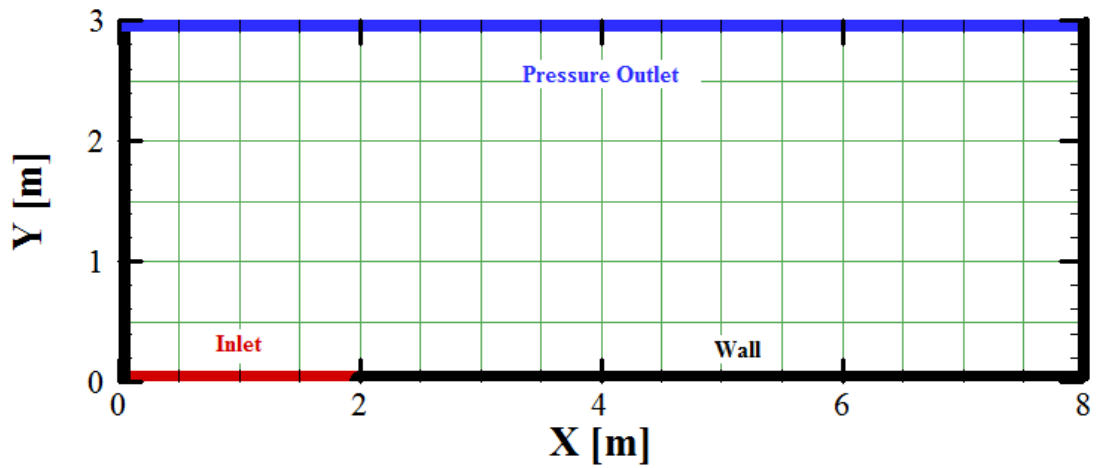


Figure 1. Examined two-dimensional geometry from [17, 20, 21]

| Parameter | Value | Reference |
|--|---------------------------------------|-----------------------------|
| Maximum biodegradation rate for Monod kinetics | 1 day^{-1} | [39] |
| Maximum biodegradation rate for 1st order kinetics | 0.05 day^{-1} | Estimated from data of [20] |
| Minimum temperature for biodegradation | $5 \text{ }^{\circ}\text{C}$ | [9] |
| Optimum temperature for biodegradation | $58.6 \text{ }^{\circ}\text{C}$ | [9] |
| Maximum temperature for biodegradation | $71.6 \text{ }^{\circ}\text{C}$ | [9] |
| Biodegradable solid waste half saturation constant | 1 kg/m^3 | [20] |
| Oxygen half saturation constant | $0.007 \times 10^{-3} \text{ kg/m}^3$ | [36] |
| Biomass decay rate | 0.05 day^{-1} | [39] |
| Yield Coefficient | 0.1 | [21] |
| | $\text{kg}_B \text{ m}^3\text{-sec/}$ | |
| | $\text{kg}_S \text{ m}^3\text{-sec}$ | |

Table 2: Values of biochemical parameters of the used kinetics in the present study.

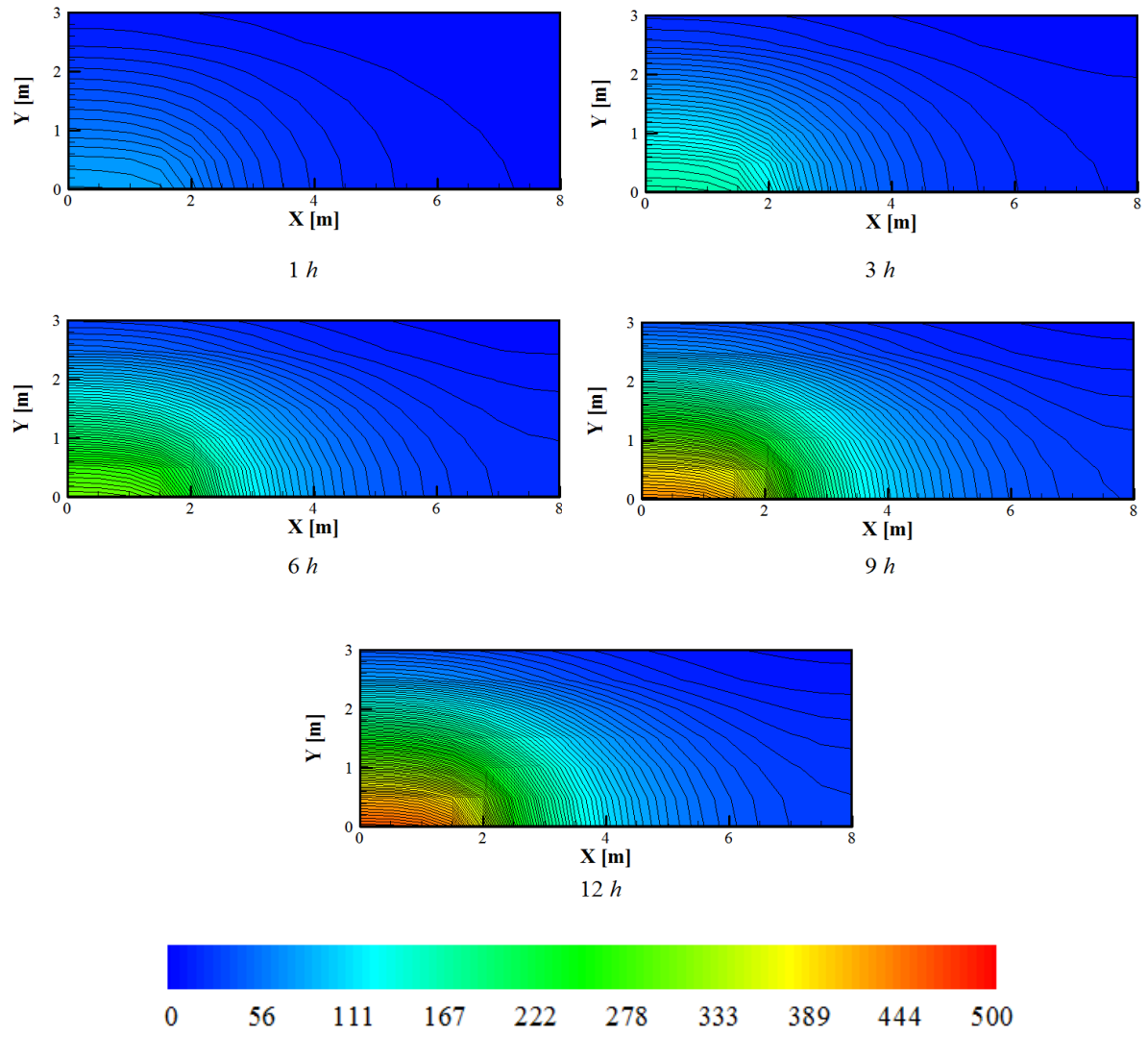


Figure 2. Relative pressure distribution [Pa]

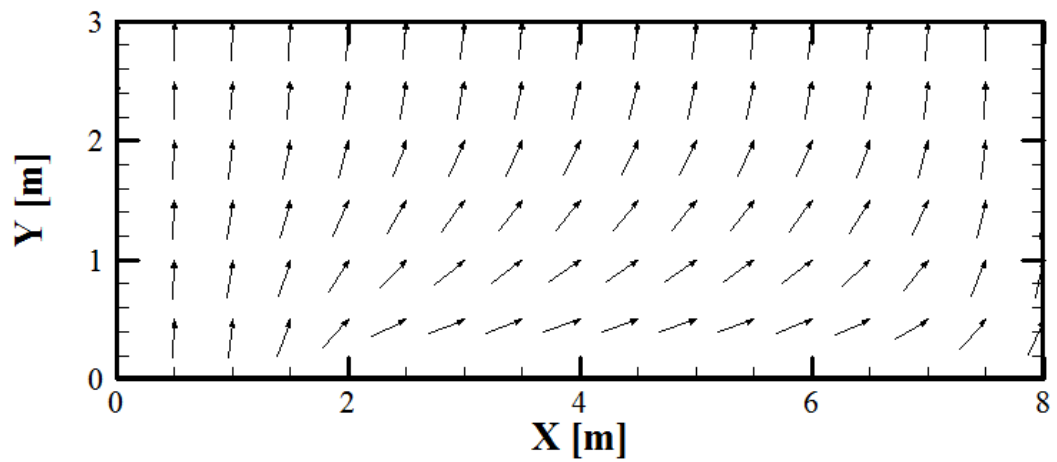


Figure 3. Directional velocity vectors

Rees-White et al. [20] used the "LDAT" model in order to simulate solid waste aeration in a 3m stack, presenting corresponding experimental investigations of the temperature field. In Figure 4, computational results of the present model for temperature versus time, using both Monod and first-order kinetics are shown and compared against reported data of previous investigations [20]. Hydraulic parameters were assumed to be the same as in the previous box aeration simulation and biochemical parameters were assumed to be those of Table 2. Initial biodegradable solid waste concentration was considered to be 200 kg/m^3 , while initial aerobic biomass concentration was set at 1 kg/m^3 , oxygen concentration in air was 0.177 kg/m^3 and temperature was 21.3°C . Simulation period was 2.5 days. It is shown that the present model's results are close to the previous experimental and computational results from Rees-White et al. [20].

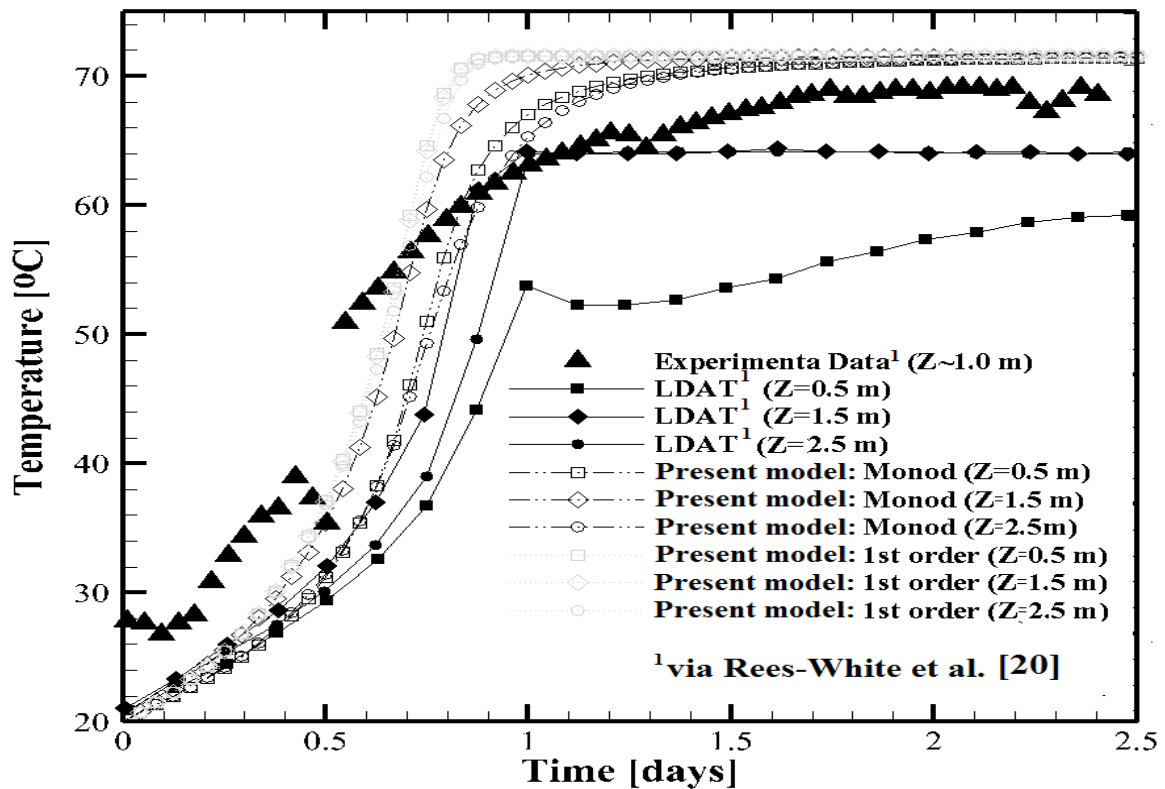


Figure 4. Temperature results of the present model versus experimental and computational results of Rees-White et al. [20]

Using the geometry of Figure 3, a sensitivity analysis for basic biochemical parameters was conducted using the same initial values. In order to compare the results a monitoring point at the center of the geometry was considered ($X = 4 \text{ m}$ and $Y = 1.5 \text{ m}$), while boundary conditions are again the same.

As it is mentioned in paragraph 2.2, the biodegradation rate of organic solid waste is affected by temperature according to equation (20). In Figure 5, the results of biodegradable solid waste concentration and temperature versus time are presented for various values of maximum biodegradation rate constant. Maximum biodegradation rate seems to affect the time instant, when the maximum temperature is reached. After the maximum temperature is reached, relatively slow biodegradation rates occur for all the k_m values. A 25% increase of k_m value results in a decrease of 0.20 days for the time needed to reach the maximum temperature, while a 50% decrease of its

value results in a 0.30-day earlier reach of the maximum temperature. On the other hand, when k_m decreases by 25 and 50% a delay of 0.5 and 1 day respectively is observed.

In Figure 6, results of biodegradable solid waste concentration and temperature are shown for various values of oxygen half saturation constant. The model is more sensitive to maximum biodegradation rate than to oxygen half saturation, thus, a 50% increase of K_{O_2} results in a 0.1 days delay to reach the maximum temperature, while a 50% decrease of K_{O_2} results in a relatively insignificant effect on the results.

In Figure 7, the effect of the value of biodegradable solid waste half saturation constant on biodegradable solid waste concentration and the temperature is shown. An 100% increase of the solid waste half saturation constant has almost no-effect on the results compared to the results of the default values. An 1000% increase ($10 \times K_s$) of solid waste half saturation constant has again small effect on the results presenting probably the need of more accurate experimental investigation of K_s , as small K_s values compared to biodegradable solid waste concentration C_s result in zero order kinetics.

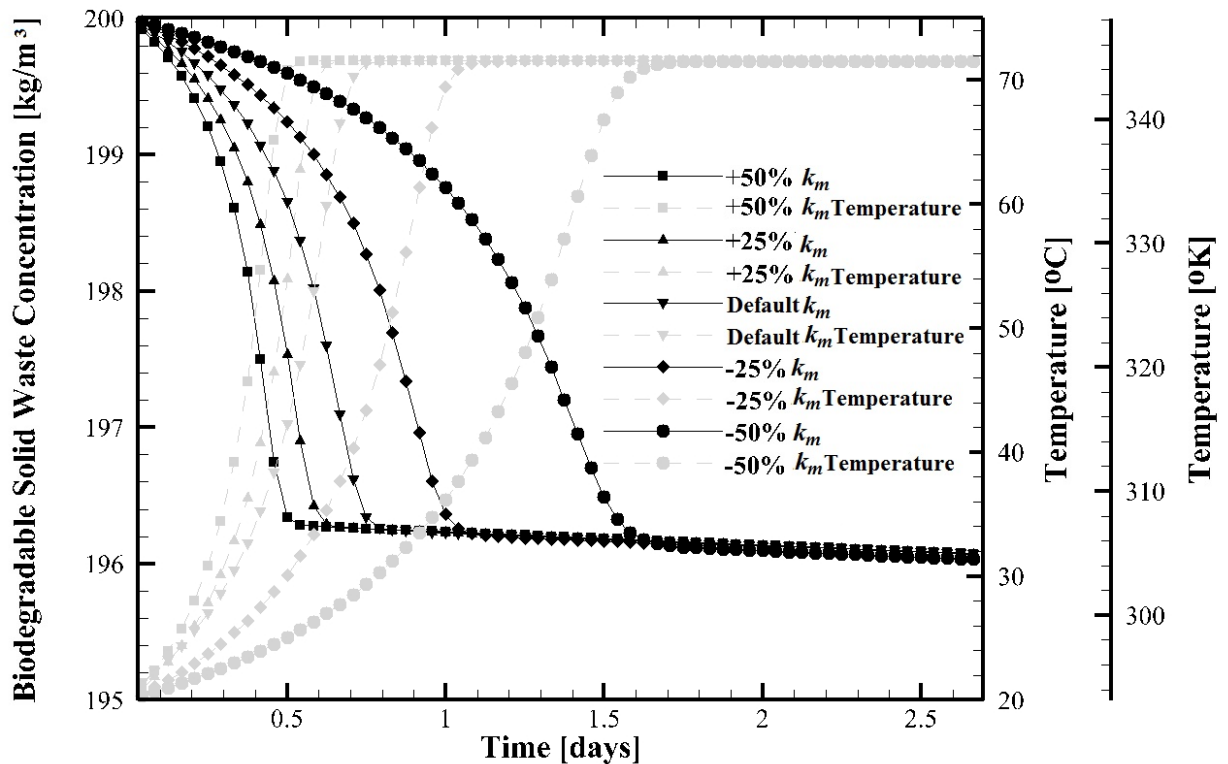


Figure 5. Biodegradable solid waste concentration (black lines) and temperature (gray lines) results for various maximum biodegradation rates

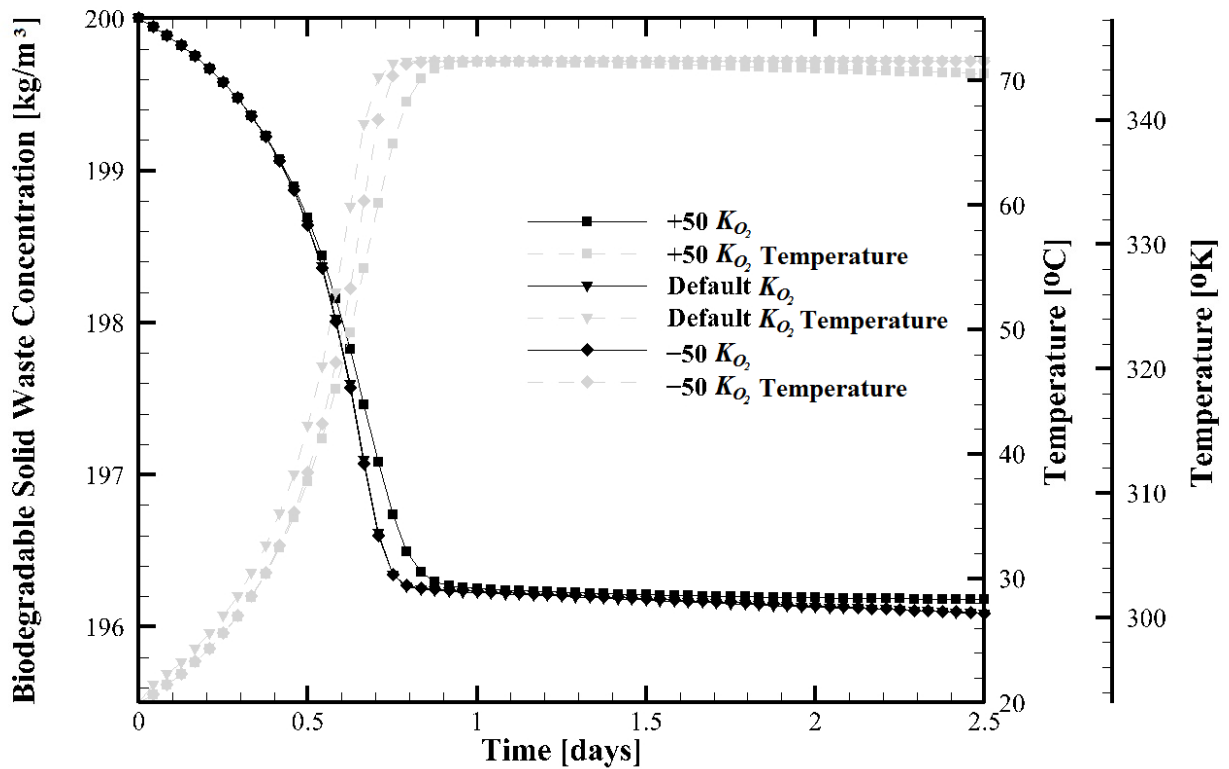


Figure 6. Biodegradable solid waste concentration (black lines) and temperature (gray lines) results for various oxygen half saturation constants

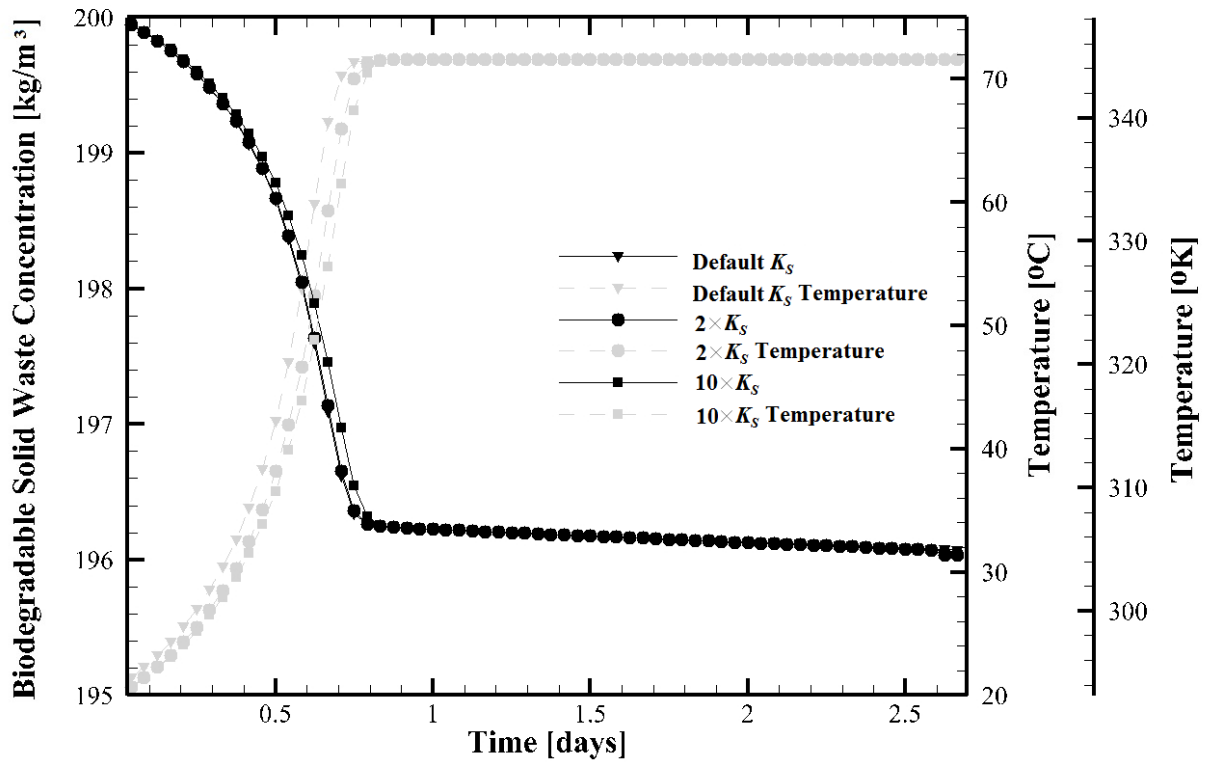


Figure 7. Biodegradable solid waste concentration (black lines) and temperature (gray lines) results for various biodegradable solid waste half saturation constants

Finally, a hypothetical case of an aeration well was examined in order to define its radius of influence. The aeration well with an internal diameter of 15 cm was assumed in the middle of a 20.0 m×20.0 m domain. The intrinsic permeability of the solid waste was assumed to be $10 \times 10^{-11} \text{ m}^2$, porosity 0.33 whilst van Genuchten's constants α , β , and γ were assumed to be 0.5, 2 and 0.8 [17] respectively. Initial free pore moisture was assumed to be 23%. The applied relative pressure difference at the inlet was 5000 Pa.

Cossu and Cestaro [40] defined the radius of influence of an aeration well, assuming a low limit of 2 cmH₂O pressure. Applying the present model with the same down radius pressure limit, a radius of influence of 7.5 m was determined (Figure 8). In addition to the pressure-based zone of influence, Cossu and Cestaro [40] defined "an alternative" well radius of influence based on a limit of 13-14% for oxygen concentration. In our case, an oxygen-based radius of influence of 11.2 m was determined, using the same limit of 13-14% ($\sim 0.186 \text{ kg/m}^3$) for oxygen concentration (Figure 9). This is a value in the range of 10-14 m that was investigated from previous researchers [40].

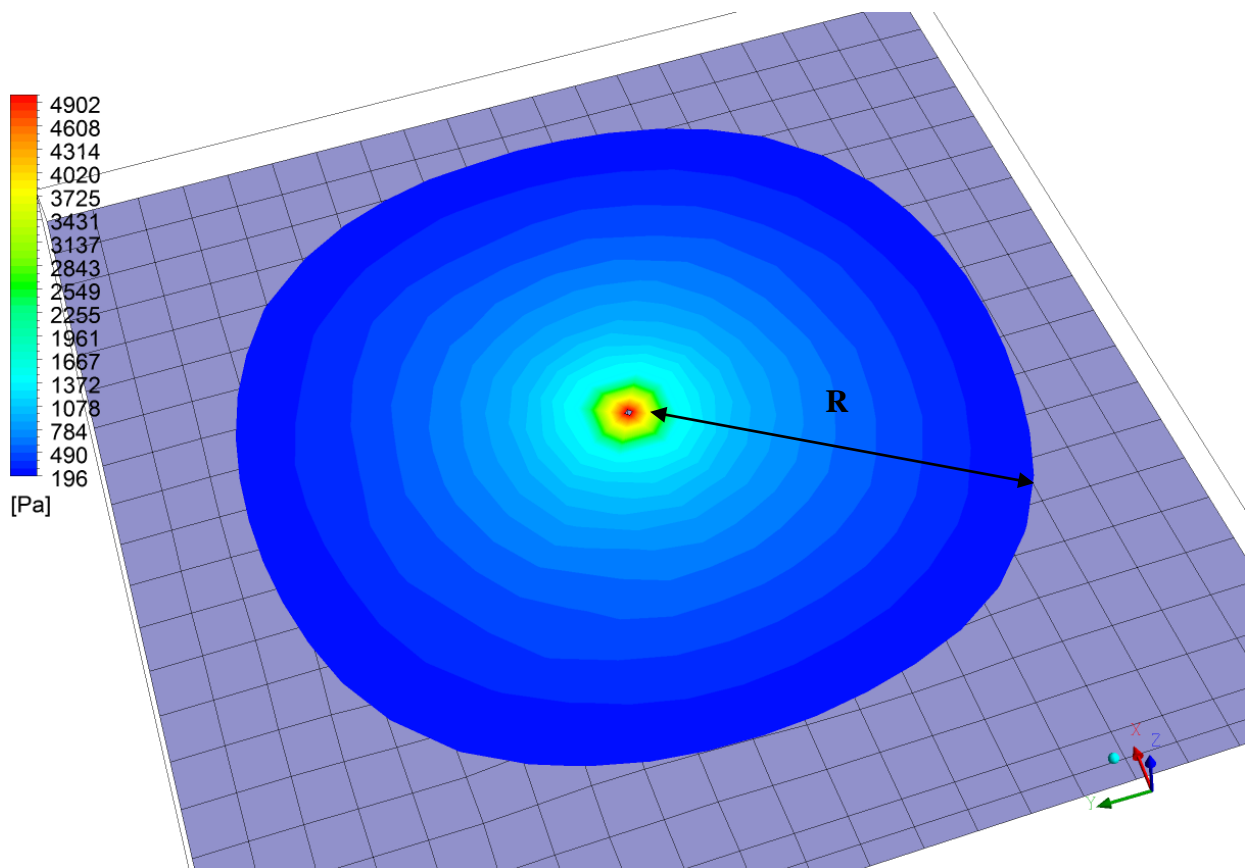


Figure 8. Relative pressure difference in a single well [Pa]

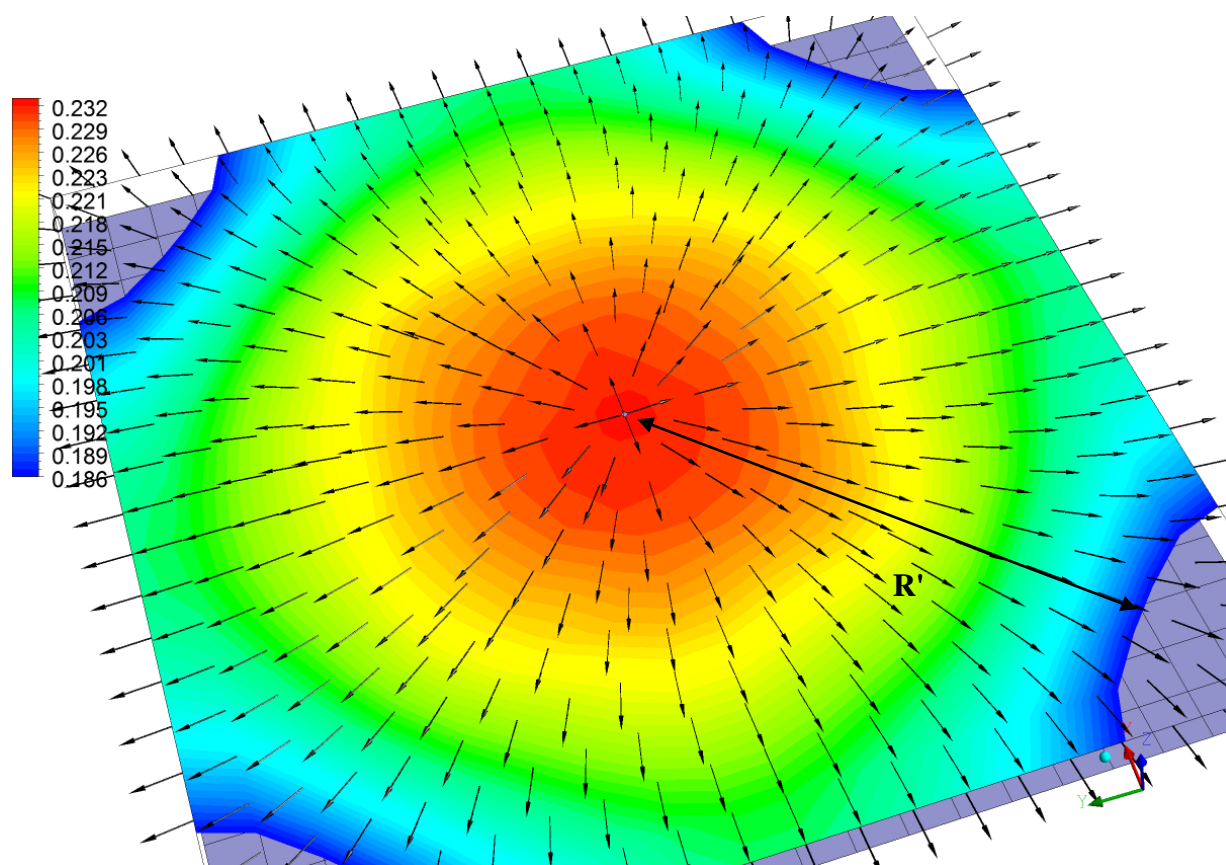


Figure 9. Oxygen concentration [kg/m^3] results and directional velocity vectors

4 CONCLUSIONS

In the present work, a computational model for the simulation of the aeration process of solid waste was developed. Computational Fluid Dynamics techniques and the algorithms included in the commercial solver ANSYS Fluent 14.5 were used. An in-house code was developed to alter the main solver and simulate the complex biochemical processes of landfill aeration. The computational investigation of some applications was conducted and the following conclusions were drawn:

- The present model uses a highly adaptable approach, which allows the use of multiple hydraulic properties for the porous media (including the capillary effect) and multiple biochemical kinetics (1st-order and Monod) and correction factor for the estimation of the effect of the environmental parameters (temperature, oxygen concentration etc).
- After a quantitative comparison, the model seems to predict accurately the temperature measured in previous works.
- The sensitivity analysis showed the high dependence of the biodegradation rate on temperature. The model is more sensitive to maximum biodegradation rate than to oxygen half saturation constant. Finally, the need for a more accurate experimental determination of the organic solid waste half saturation constant was emphasized.
- The model proved to be a useful tool for the estimation of the radius of influence for an aeration well using both pressure and oxygen concentration for its definition.

Concluding, it has to be noted that simulation of solid waste aeration is a relatively new research area, having most of the current modeling approaches to be "batch" or "point" models. Thus, the present study tried to focus on the development and application of a computational model that would combine a fluid flow model for the complex multiphase flow of water and air with a biochemical model for the biodegradation of the organic solid waste, aiming to bring the simulation of aeration systems from the area of research to engineering practice.

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