

Radius of Influence of Bio-Remediation treatment of soil and Groundwater using an Electro-kinetic cell system.

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Abstract

At pH between 5 and 9, mineralization of hydrocarbons in groundwater is highly dependent on oxygen availability. The ultimate oxygen demand, coupled with the rate of degradation, establishes the rate at which oxygen must be delivered.

Technologies based on oxygen release to the soil and/or the groundwater such as oxygen diffusion or oxygen.

Releasing chemicals (Regenesis ORC) have generally a radius of influence of few meters.

This fact has several consequences on the practice of in-situ hydrocarbons bio treatment:

1. It requires a large numbers of reactor installations resulting with a higher cost of the treatment.
2. It stops the regular activity of the pumping station or the site during the time of the treatment.

A reactor-cell that combined electro-bioremediation and electro kinetic systems has been developed by *Elie ELGRESSY*.

Because the **EBR™** cell rotates the groundwater and thereby mixes the dissolved oxygen throughout the treatment area, the uniform radial diffusion of oxygen in the soil and the use of electro-kinetics system increase the radius of influence of the treatment.

In this contribution, we first present a theoretical basis for the analysis of the radius of influence on the concentration of dissolved oxygen (D.O.) and the Redox potential (ORP) for a general electro kinetics bioremediation system.

The simulation runs show clearly that first the rotation of the groundwater near the cell creates an homogeneous boundary condition at the reactor cell resulting with an increase in the radius of influence of 2 to 3 compared with simple oxygen diffusion system.

Moreover, the electric field created by an anode and cathode system increases the radius of influence by creating an additional flux in the charged water.

In order to validate this new technology, a pilot study was performed in the Haifa region Israel.

The software was calibrated on data collected in a pilot study. An analysis of the radius of influence on data results shows the efficiency of the new technology.

Keywords: Hydrocarbon pollution, groundwater remediation, Dissolved Oxygen, Redox potential, chemical transport simulation.

1. Introduction

Soil and groundwater contamination is the presence of man-made chemicals or other alteration in the natural soil and groundwater environment. This type of contamination typically arises from the rupture of underground storage tanks, application of pesticides, and percolation of contaminated surface water to subsurface strata, leaching of wastes from landfills or direct discharge of industrial wastes to the soil and groundwater.

The most common chemicals involved are petroleum hydrocarbon, solvents, pesticides, lead and other heavy metals. This occurrence of this phenomenon is correlated with the degree of industrialization and intensity of chemical usage.

Economical restoration of contaminated sites to environmentally acceptable conditions is an important challenge facing the scientific and technical community. Current in situ soil remediation technologies depend on hydraulic and air flow for effective remediation of soils and are not as effective in the clean-up of lower hydraulic conductivity soils (less than 10⁻⁵ cm/sec) such as fine sands, silts and clays.

In situ bioremediation is an attractive and often cost-effective option to remediate soil and groundwater contaminated with organics (See Saichek et al. (2003), Mah Y.K. et al. (2005), Kristina G. et al.(2007)). Successful implementation of in situ bioremediation is dependent upon presence, or effective injection, of electron acceptors and nutrients into the porous medium. Microbial processes require an electron donor, macronutrients (e.g, nitrogen and phosphates), micronutrients, trace nutrients and an electron acceptor. The use of electric fields is an innovative method for *in situ* restoration of contaminated hazardous waste sites. Direct currents (DC) are applied across electrodes inserted in the soil to generate an electric field for mobilization and extraction of contaminants and for bio-geochemical modifications of polluted soils and slurries. The driving mechanisms for this technique, known as electrokinetic remediation, are transport under electric fields (in particular electro-osmosis and ionic migration) coupled with electrolysis and geochemical reactions.

Extraction and removal are generally achieved by electro-deposition, precipitation or ion exchange (for heavy metals) and collection and treatment of organics in external systems. Contaminants that could be treated by electric field applications include inorganic, organic, and radioactive compounds that are charged (ionic) or non-charged (polar and non-polar).

The major advantages of the technology include:

- it can be implemented *in situ* with minimal disruption,
- it is well suited for fine-grained, heterogeneous media, where other techniques such as pump-and-treat can be ineffective, and accelerated rates of contaminant transport and extraction may be obtained.

At pH between 5 and 9, mineralization of hydrocarbons in groundwater is highly dependent on oxygen availability. The ultimate oxygen demand, coupled with the rate of degradation, establishes the rate at which oxygen must be delivered. Technologies based on oxygen release to the soil and/or the groundwater such as oxygen diffusion or oxygen releasing chemicals (Regenesis ORC) have generally a radius of influence of few meters.

This fact has several consequences on the practice of in-situ hydrocarbons bio treatment:

1. It requires a large numbers of reactor installations resulting with a higher cost of the treatment.
2. It stops the regular activity of the pumping station or the site during the time of the treatment.

1.1 E.K™ Systems

An innovative reactor-cell and method that combined electro-bioremediation and electro kinetic systems has been developed by *Elie Elgressy*.

Three types of measurements are performed in the soil in order to check the efficiency of the treatment:

- a. The ORP measurement.
- b. The DO measurement
- c. Bacteria count

A useful concept for bioremediation is that the ORP is a measurement of electrons that are available to reduce contaminants in the same way that pH is the measurement of available protons.

Oxygen is the preferred electron acceptor in aerobic respiration followed by nitrates, iron(+3), and sulfates. Therefore, the rate of hydrocarbon degradation increases with increasing ORP (Riser-Roberts, 1992). At a contaminated site where the dissolved oxygen level in groundwater is low (0.5 ppm) and the ORP is low (50 mV), degradation is occurring very slowly with the conversion of iron(+3) to iron(+2) is the dominant reduction reaction.

Dissolved oxygen is still the preferred (and fastest) electron acceptor, but, there isn't enough oxygen to make a difference. As the dissolved oxygen level is increased and more oxygen is available to accept electrons, the potential for the oxygen to be reduced increases, and the ORP increases.

The initial dissolved oxygen will be used to oxidize other materials before it's available to the bacteria because of a greater difference in potential energy yield of the reaction.

The preferred order of dissolved oxygen usage is decomposing organic matter (ORP-750 to -240 mV), oxidation of sulfides (ORP 600- to -150 mV),

oxidation (rusting) of iron (ORP-550 to +50 mV), oxidation of nitrates (ORP 150- to +400 mV), and finally aerobic respiration (bacteria, ORP +100 to +800 mV).

The ORP looks at the total solutions potential and is a great indicator of oxygen being available or respiration. The ORP, however, cannot tell if the bacteria, contaminant, and oxygen are all in the same area so respiration occurs. Therefore, an abundance of centralized dissolved oxygen would generate a high ORP, even though the dissolved oxygen may be diffusing vertically upward to the vadose zone. An example would be oxygen diffusion or oxygen releasing chemicals (Regenesis RC) that rely on groundwater movement for transport farther than what occurs by diffusion.

Because the **EBR™** cell rotates, the groundwater and thereby mixes the dissolved oxygen throughout the treatment area, ORP is a more accurate indicator of respiration than in diffusion limited methods like oxygen diffusion or oxygen releasing chemicals (Regenesis ORC)

□ DO (Dissolved Oxygen)

At pH between 5 and 9, mineralization of hydrocarbons in groundwater is highly dependent on oxygen availability. The ultimate oxygen demand, coupled with the rate of degradation, establishes the rate at which oxygen must be delivered. Oxygen supply should be balanced with oxygen demand to optimize the cleanup time for bioremediation.

Too low an oxygen supply rate relative to the contaminant load results in extended remediation times. Too high a rate of oxygen supply results in elevated remedial cost and potential for soil gas binding. The theoretical oxygen requirements to remediate one gram of gasoline are 2.5 grams of dissolved oxygen to reduce the contaminant and 4.0 grams of oxygen for complete mineralization to carbon dioxide and water.

Dissolved oxygen should be maintained above the critical concentration for the promotion of aerobic activity, which ranges from 0.2 to 2.0 mg/l, with the most common being 0.5 mg/l (EPA, 1985).

For example, hydrocarbon degrading bacteria in well-oxygenated groundwater containing 4mg/l of molecular oxygen can degrade only 2 mg/l benzene.

□ **EKTM**- technology

EK (Electro-kinetics) in soil and ground water:

The Electrochemical remediation methods are based on the transport processes that occur when an electric current is passed through a soil or groundwater.

The movement of these particles is similar to the movement of ions. In the pore fluid of clay soils, the particles participate in the transfer of electrical charges and influence the electrical conductivity and the electro osmotic flow.

In the soil the current is carried by ions, and the anions will be transported toward the anode and the cations toward the cathode, this transport of ions in the electric field is electromigration.

Electro-osmosis is movement of pore water in the soil in the applied electric field.

The negative charge on the surface of most soil particles causes an accumulation of positively charged cations near the surface in the diffuse electric double layer.

Under the action of the electric field these actions will give a net flow of ions in the direction of the cathode, and by this water is forced to move towards the cathode.

□ **Electrolysis**

Application of direct electric current through electrodes immersed in water induces electrolysis reactions at the electrodes. Oxidation of water at the anode generates an acid front while reduction at the cathode produces a base front as described the classical electrolysis reactions.

The prevailing of electrolysis reactions at the electrodes depends on the availability of chemical species and the electrochemical potentials of these reactions. Although some secondary reactions might be favored at the cathode because of their lower electrochemical potential, the water reduction half reaction (H_2O/H_2) is dominant at early stages of the process. Within the first few days of processing, electrolysis reactions drops

the pH at the anode to below 2 and increase it at the cathode to above 10, depending upon the total current applied (Acar and Alshawabkeh 1993). Studies (Probstein and Hicks 1993; Eykholt and Daniel, 1994; Yeung and Datla, 1995) showed that while acid production enhances the process development of a high pH zone at the cathode adversely impacts extraction of heavy metals from soils.

2 Modeling approaches

Physical understanding and numerical simulation of the bio-remediation by electrokinetic treatment requires the mathematical formulation of transport processes, which are controlled by such variables as electrolysis reactions at the electrodes, pH and soil surface chemistry, equilibrium chemistry of the aqueous system, electrochemistry of the contaminants, and geotechnical/hydrological characteristics of the porous medium.

The complexity of transport processes necessitates working assumptions that simplify numerical simulation.

The following assumptions are employed in the theoretical development presented in this work:

- (1) The soil medium is isotropic,
- (2) The porous medium is a solid framework of cation exchange surfaces with the pore by chemically reactive species in aqueous solution,
- (3) All fluxes are linear homogeneous functions of all driving forces (or potential gradients),
- (4) Isothermal conditions prevail (coupled heat transfer is neglected),
- (5) All the applied voltage is effective in fluid and charge transport,
- (6) Electrophoresis is not present except near the bio-reactor,
- (7) The chemico-osmotic coupling is negligible,
- (8) Soil particles are treated as electrically nonconductive (insulators),
- (9) Surface conductance and streaming potential are negligible.

Based on the assumptions, theoretical formulations are provided for transport mechanisms under electric fields. Contaminant transport mechanisms include hydraulic or fluid flow, species or mass transport, and charge transport. When an electric gradient is imposed to a wet soil, cations (positive ions) move toward the cathode (negative electrode) and anions (negative ions) move toward the anode (positive electrode). Fine-grained soils, such as clay, possess a negative charge due to isomorphous substitution and the presence of broken bonds in the soil structure.

Therefore, excess mobile cations are required to balance the negative fixed charge on the soil solid particle surfaces. Mobile ions can drag water molecules when they are moved by an imposed electric field. As cations are in excess in the system, there is a net movement of pore water toward the cathode.

Several theoretical models have been proposed for soil decontamination using electrokinetic phenomena. Shapiro et al. (1993) developed a theoretical model for phenol and acetic acid extraction based on the advection-dispersion equation. Retardation of the contaminant was incorporated by using a linear sorption isotherm in their model.

Even though they reported that the model satisfactorily predicts transport of contaminants and pH distribution, no attempt was made to model nonlinear voltage distribution and the coupling effect between gradients.

Eykholt and Daniel (1994) conducted a series of electrokinetic extraction tests to evaluate the capability of the process to remove heavy metals. They developed a theoretical formulation to predict concentration distribution, pH distribution, amount of flow, and voltage distribution along the length of the specimen. However, the assumption that the voltage distribution follows a bilinear distribution has been argued by different researchers (West and Stewart 1996). Also, the buffer capacity of the soil for pH computations and surface complication was not considered.

Alshwabkeh and Acar (1996) developed general theoretical frameworks for general electro-kinetics remediation.

They apply it for the description of the transport of lead (Pb) and nitrate (NO_3^-) in a coupled system.

The model can predict acid transport, lead transport and electric potential, pore pressure distribution across the electrodes in electrokinetic remediation. The surface conductance of clay particles was not considered and only a few dominant contaminant ions in the pore fluid were taken into account. The concentration of NO_3^- is determined from the charge balance with Pb.

The modeling of chemical and electrical flows in low permeability under electrical gradient has also been formulated using a generalized coupled flow equation where anion transport and cation transport is separated (Kim 1999).

In the present work we are interested to model the change in concentration of the dissolved oxygen DO and the Redox potential (ORP). Also we considered the flow in the saturated zone and the vadose zone where the groundwater may be unsaturated (or saturated with negative water pressure).

2.1 Mathematical model

In the **EBR™** system, the oxygenated recharge water is taken from wells, from a water supply system or from reservoirs. The recharge water is saturated with dissolved oxygen using a patented rotation system that mixed the oxygen in the reactor cell.

The oxygenated water should be recharged into the aquifer through discharge wells as explained below. Thus the recharged water is distributed into the same layers of the aquifer and the same pores which are pumped subsequently. At least two wells are necessary for alternative recharge and pumping. If more than two wells are available a part of the pumped water can be used for continuous supply and another part can be abstracted, saturated with oxygen, and recharged via one of the wells.

In the context of design of a plant for subterranean groundwater treatment, the movement of oxidizing agents together with their reactions with other substances such as organic matter and specially iron and manganese within the porous medium must be considered.

The transport of oxygen in groundwater depends on the velocity of the groundwater and of the recharged water and on dispersion and sorption processes; as well.

2.2 Fluid Flux

Fluid flux results from application of a hydraulic gradient (Darcy's law), an electric gradient (electro-osmosis) and/or a chemical gradient (chemical-osmosis).

Chemical-osmosis could be neglected because it is significant only in the presence of large chain molecules and in very active clay deposits (activity describes plasticity of the soil and equals plasticity index divided by clay fraction in percentage).

Fluid flux per unit area of the porous medium due to hydraulic and electric gradients,

J_w ($L^3L^{-2}T^{-1}$), is given by,

$$J_w = -k_h \nabla(h) - k_e \nabla(\phi) \quad (1)$$

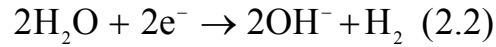
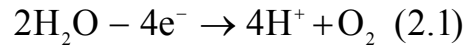
Where ∇ is the gradient operator, k_h (LT^{-1}) is the coefficient of hydraulic conductivity,

k_e ($L^2V^{-1}T^{-1}$) the coefficient of electro-osmotic permeability, h is the hydraulic head (L)

and ϕ is the electric potential (V). Equation (1) is valid for saturated conditions. For

unsaturated conditions the hydraulic conductivity is a function of the head $k_h(h)$ (LT^{-1}).

When inert electrodes are used in groundwater, oxidation of water at the anode generates an acid front, while reduction at the cathode produces a base front by the following electrolysis reactions:



The dissolved oxygen concentration (DO) corresponds to the measurement of O_2 concentration in the groundwater while the potential ORP measures the presence of e^- in the groundwater.

The transport and expansion of oxygenated water can be measured through experiments or by using a mathematical model, as was found by (Shapiro et al. (1993), Rutt (1990), Page (2002)).

The model is based on a diffusion-convection equation combined with a sorption term for both the dissolved oxygen and the redox potential.

In a general formulation we should also introduce an equation for the computation of the electric potential. We assume in this work that after the installation of the anode and cathode well, a static electric field has been created.

1.1 Modeling Transport for dissolved oxygen and the potential

Flow equation

Flow of any fluid is governed by mass and momentum conservation. In the case of groundwater, Darcy's law expresses the latter. Mass conservation, using the classical Boussinesq approximation (Bear 1979) can be written as

$$\frac{\partial(\rho\theta)}{\partial t} = -\rho\nabla \cdot \mathbf{q} + \rho Q_h$$

Where ρ the fluid density (kg/m³) is, θ is the porosity (volume of fluid per volume of soil), \mathbf{q} is the fluid flux (m³/s), and Q_h is the sink/source mass flux (m³/s).

As the state variable for the flow equation, both the pressure p (Pa) and the equivalent freshwater head h_f (m) can be used.

$$h_f = \frac{p}{\rho_0 g} + z$$

The flux \mathbf{q} is given by the general Darcy's law.

$$\mathbf{q} = -\mathbf{k}_h \nabla(h_f) - \mathbf{k}_e \nabla(\phi) \quad (1)$$

\mathbf{k} is the intrinsic permeability tensor (m²) and P the pressure.

In order to fully constrain the solution of the flow equation, spatial and temporal boundary conditions must be imposed. On all domain boundaries, boundary conditions must be unique and defined at all times. There are three types of flow boundary conditions:

- Dirichlet or fixed head boundary: $P(x,t) = f(x,t)$, which is usually adopted at the sea boundary.

- Neuman or constant flux boundary: $\frac{\rho_s}{\rho_f} \vec{q} \cdot \vec{n} = f(x,t)$

- Cauchy or mixed boundary: $\vec{q} \cdot \vec{n} + \alpha((\rho_s/\rho_f)z - h_f) = 0$

Transport equation

The mass balance of solutes can be expressed as the change in solute mass being equal to the divergence of mass fluxes (advective, diffusive and dispersive ones) plus sink/source contributions.

For the concentration of dissolved oxygen (DO), the basic equation is given by:

$$\frac{\partial \left(\rho \left(\omega + \frac{1-\theta}{\theta} L_H \right) \right)}{\partial t} = -\nabla \cdot (\mathbf{j}_{adv} + \mathbf{j}_{disp}) + \rho Q \xi^*$$

Where: ω is the concentration of the dissolved material after reaction with the dissolved oxygen, L_H is the concentration of the sorbed material after reaction with the dissolved oxygen.

The advective mass flux is straightforward to express and is equal to $\mathbf{j}_{adv} = \rho \mathbf{q} \omega$. The dispersive and diffusive fluxes are treated together as $\mathbf{j}_{dif} = -\rho \mathbf{D} \nabla \omega$ where \mathbf{D} is the diffusion and dispersion tensor. According to the Bear-Scheidegger dispersion relationship (Bear, 1979), it is equal to

$\mathbf{D} = \phi(\mathbf{D}_d + \mathbf{D}_m)$ in the case of saturated flow.

$$\mathbf{D}_d = D_d \mathbf{T}$$

$$\mathbf{D}_m = \alpha_T |q| \mathbf{I} + (\alpha_L - \alpha_T) \frac{\mathbf{q} \otimes \mathbf{q}}{|q|}$$

Where \mathbf{D}_d is the molecular diffusion tensor and \mathbf{T} the tortuosity tensor. α_L and α_T are called longitudinal and transversal dispersivity (m).

$\mathbf{D}_d = D_d \mathbf{T}$ will not be considered in this work.

The mass balance equation may be written as:

$$\rho \theta \frac{\partial [\omega + (1-\theta)L_H]}{\partial t} = -\rho \mathbf{q} \cdot \nabla(\omega) + \nabla \cdot (\rho \mathbf{D} \nabla \omega) + \rho Q(\omega^* - \omega)$$

Similarly, for the ORP concentration the transport equation may be written as:

$$\rho \theta \frac{\partial [\omega + (1-\theta)F_H]}{\partial t} = -\rho \mathbf{q} \cdot \nabla(\omega) + \nabla \cdot (\rho \mathbf{D} \nabla \omega) + \rho Q(\omega^* - \omega)$$

F_H is the concentration of the sorbed material after REDOX reaction .

The equation for the rate of sorption was taken as linear.

1. For the REDOX reaction

$$\frac{\partial F_H}{\partial t} = a_1 \omega - b_1 F_H$$

where $a_1; b_1$ are rate coefficients.

2. For the DO concentration,

$$\frac{\partial L_H}{\partial t} = a_2 \xi - b_2 L_H$$

where $a_2; b_2$ are rate coefficients.

The rate of sorption for both, DO and ORP, depends on the -type of soil, the substances, and the concentration of the species in the dissolved as well as the sorbed phases. The simplest calculation leads to equation (2) assuming that the change of concentration is equal to the difference between the rates of adsorption and desorption. Assuming equilibrium between liquid and solid phase, the concentration of sorted material depends on the concentration of the solute only. The reaction between the dissolved state and solid state may be approximated by a linear Freundlich or Langmir isotherm.

2.2.1 Initial and boundary Conditions for the transport equations

Because the **EBR™** cell rotates the groundwater and thereby mixes the dissolved oxygen throughout the treatment area, the boundary conditions are different for the transport equations than for a classical pressure system:

a. Classic system

Just like in the flow equation initial conditions and boundary conditions are required.

The following boundary conditions are available:

For $z \leq 0$

Initial conditions:

$$\omega = \omega_0(x, 0); L_h(x, 0) = F_h(x, 0) = 0$$

Boundary conditions:

$$\frac{\partial \omega}{\partial z} = 0;$$

At the system:

$$\frac{\partial F_H}{\partial z} = \frac{\partial L_H}{\partial z} = 0$$

$$\omega \rightarrow 0;$$

Far from the system

$$F_H \rightarrow 0; L_H \rightarrow 0$$

EBR™ system

For $z \leq 0$

Initial conditions:

$$\omega = \omega_0(x, 0); L_h(x, 0) = F_h(x, 0) = 0$$

Boundary conditions:

$$\frac{\partial \omega}{\partial z} = 0;$$

At the system: $F_H(0, t) = F_0$

$$L_H(0, t) = L_0$$

$$\omega \rightarrow 0;$$

Far from the system

$$F_H \rightarrow 0; L_H \rightarrow 0$$

2.3 The FEFLOW software

In order to simulate the above equation for each vertical strip we use the FEFLOW numerical model and its interface. FEFLOW is a two- and three-dimensional finite element model for density-dependent flow, mass and heat transport processes, developed at WASY, Berlin. FEFLOW incorporates an advanced, hierarchically structured graphical working environment. The package is fully interactive, and contains graphical editors and 2D/3D finite-element mesh generators. FEFLOW also provides a data interface to GIS (Geographic Information System) and an interface manager for open programming.

The mass conservation equation is solved for a single component governed by advection, diffusion, mechanical dispersion, retardation (Henry, Freundlich or Langmuir isotherms), zero and first-order reaction terms. The model considers the advection-dispersion equation for component mass (salinity) and heat. Fluid density effects may be caused by simultaneous solute mass and temperature differences (thermohaline flow).

A Galerkin finite-element method for unstructured meshes is employed. For convection dominated problems, various types of up-winding techniques are provided to minimize numerical dispersion. An adaptive approach for mesh refinement and mesh coarsening is available in 2D. Time stepping schemes can be chosen from automatic predictor-corrector schemes of second order accuracy (Adams-Bashforth/ trapezoidal rule), or first order Euler or Crank-Nicolson schemes. Nonlinearities are solved either by the Newton or Picard iteration methods. Various equation solver and pre-conditioning algorithms are available for symmetric or non symmetric systems of equations.

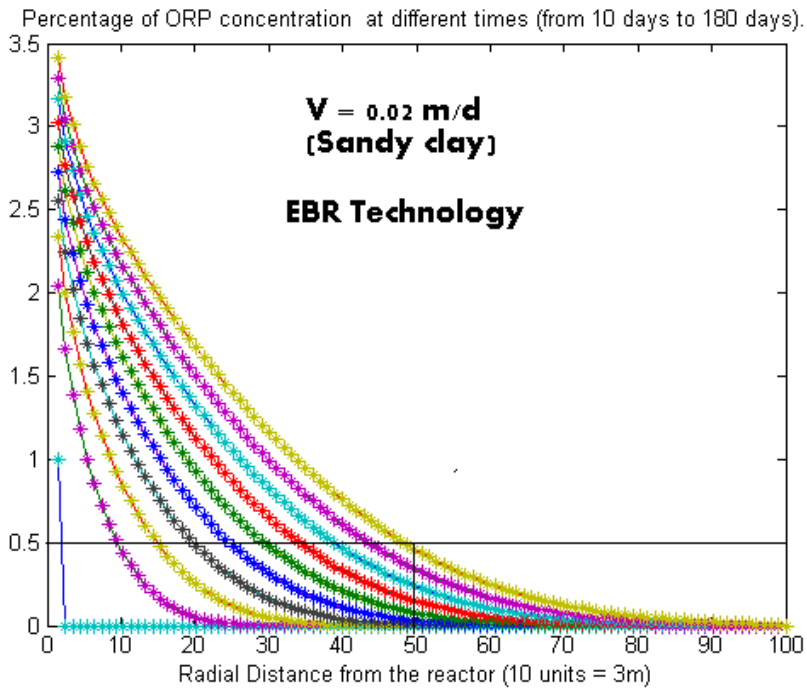
First-, second- and third-type boundary conditions are accommodated in FEFLOW that are specified for flow, mass, and heat. A specific boundary condition exists for singular pumping or

injection wells. Furthermore, constraint formulations can be combined for boundary conditions. The boundary conditions are node related, while material parameters are handled on an element-by-element basis. Boundary fluxes and transfer conditions can be assigned graphically.

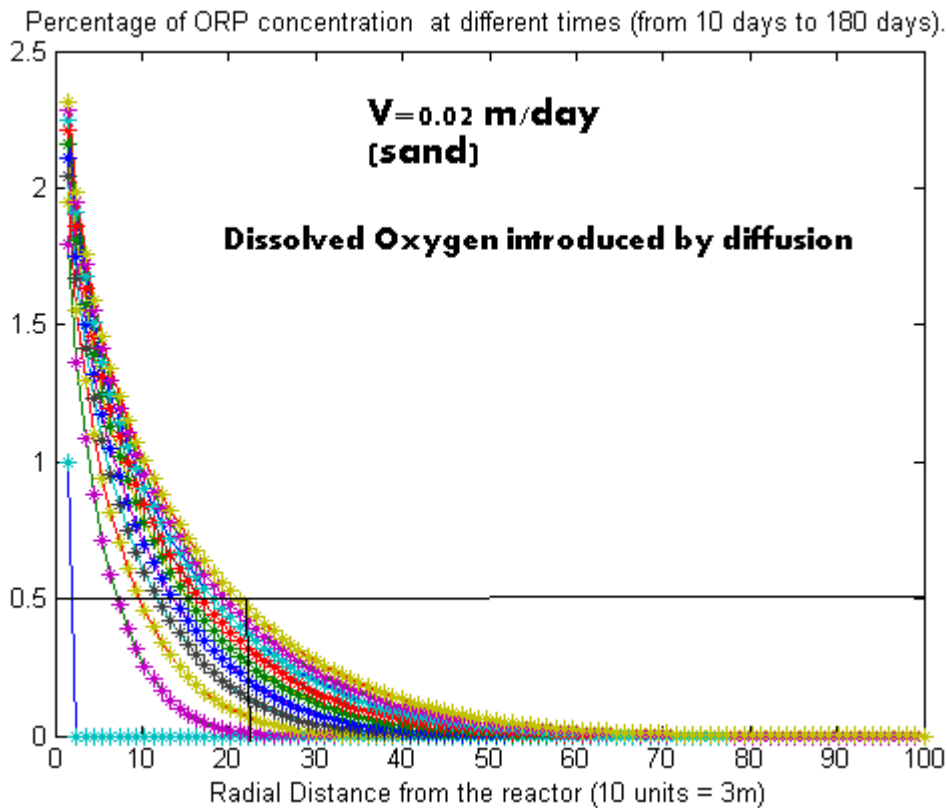
2. Modeling analysis of the radius of influence.

The mathematical model produces under horizontal and radial symmetrical screen conditions, concentration profiles of the oxygen in oxygenated water than had been recharged into a well and of solute iron in the aquifer as shown in Fig. 1a, 1b.

Figure 1a and 1b described the profiles of the ORP concentration at different times using the formulation for the **EBR™** system and for the classical system.



Figures 1a-1b: Percentage of ORP concentration at different times for the ERP system and the dissolved Oxygen system introduced by pressure



The results of the simulation show that the radius of influence is twice larger for the **EBR™** system.

5 Computation of the radius of influence by calibration of the numerical model

The analysis of the spatial effect of the **EBR™** reactor during a long term period (until 180 days) was performed using two different numerical models. The first model computes the spatial expansion of the dissolved oxygen into the groundwater and also into the soil in the unsaturated zone. While the second model, uses the REDOX process for the prediction of the spatial expansion of ORP concentration at the vicinity of the reactor.

The three dimensional coupled flow and transport of chemical species model described above was used for the computation. This numerical model FEFLOW developed in Germany was used for the solution of the couple problem for dissolved oxygen and potential ORP. This software is one of the reference models in hydro- geology. It is approved by the American EPA (Environmental Protection Agency) and the water commission of Israel as well. The geological data and hydrological parameters were collected from existing calibrated models of the Israel coastal aquifers. The dispersion coefficient and the retardation factors appearing in the transport equation were determined by calibrated the model to the results of the pilot tests.

The initial concentrations in the reactor were taken at 10 ppm of D.O and 670 mV for the ORP. We defined the radius of influence of the **EBR™** reactor for the ORP concentration as the spatial area where the ORP rises to a value that is more than 50 % of the ORP **EBR™** reactor concentration. In the same way, we defined the radius of influence of the **EBR™** reactor for the DO concentration as the spatial area where the DO decreases to a value that is more than 50 % of the DO **EBR™** reactor concentration.

The different simulation runs performed estimate the radius of influence of the **EBR™** system.

5.1 Simulation for the dissolved oxygen (DO)

5.1.1 Saturated zones

The reactor cell was placed at a depth of 4.3 m from the ground level. The concentration of dissolved oxygen (DO) was of 10 ppm in the reactor.

Figure 4 shows the contour lines of the DO concentration after 180 days obtained by simulation.

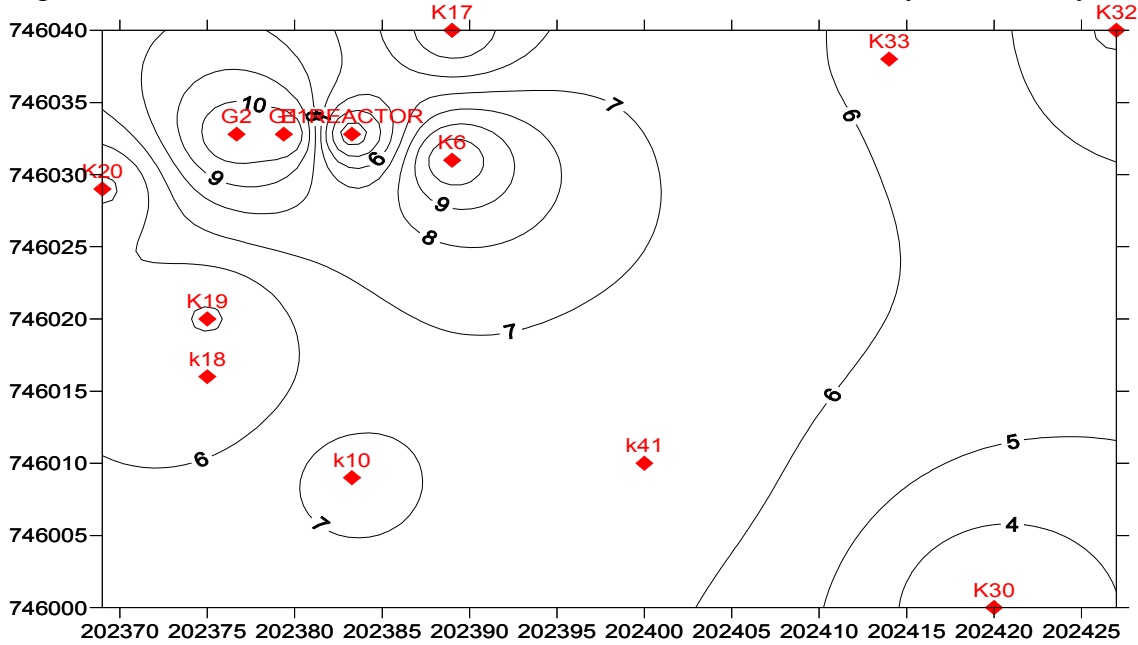


Figure 4: DO concentration in the groundwater at a depth of 4.3 m.

In the following table, we estimate from the simulation results the radius of influence for different time steps.

Radius/time (days)	30	60	120	180
R(9ppm)	4	7	12	15
R(5ppm)	5	10	16	20

Table 1: Radius of Influence for different time for DO simulation at 4.3m depth.

5.1.2. Influence of the position of the reactor

The reactor cell was placed at a depth of 1 m from the ground level. The concentration of dissolved oxygen (DO) was of 10 ppm in the reactor.

Figure 5 shows the contour lines of the DO concentration after 180 days obtained by simulation.

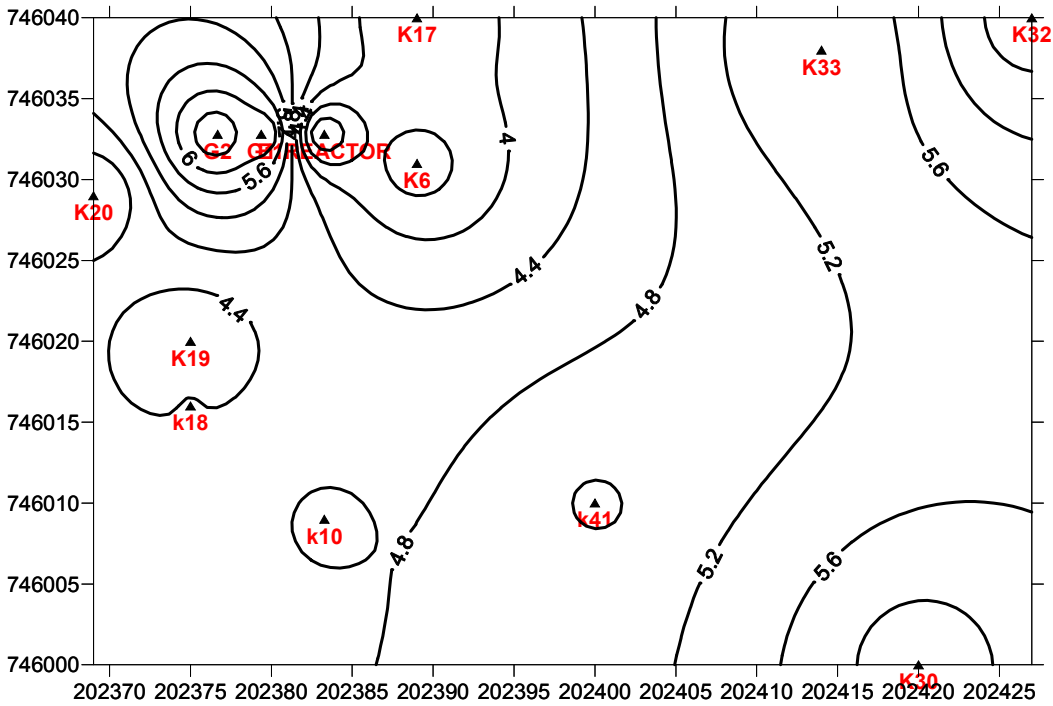


Figure 5: DO concentration in the groundwater at a depth of 1 m.

In the following table, we estimate from the simulation results the radius of influence for different time steps.

Radius/time (days)	30	60	120	180
R(9ppm)	3	5	8	12
R(5ppm)	4	6	9	14

Table 2: Radius of Influence for different time for DO simulation at 1m depth.

Figure 6 shows a three dimensional view of the radius of influence.

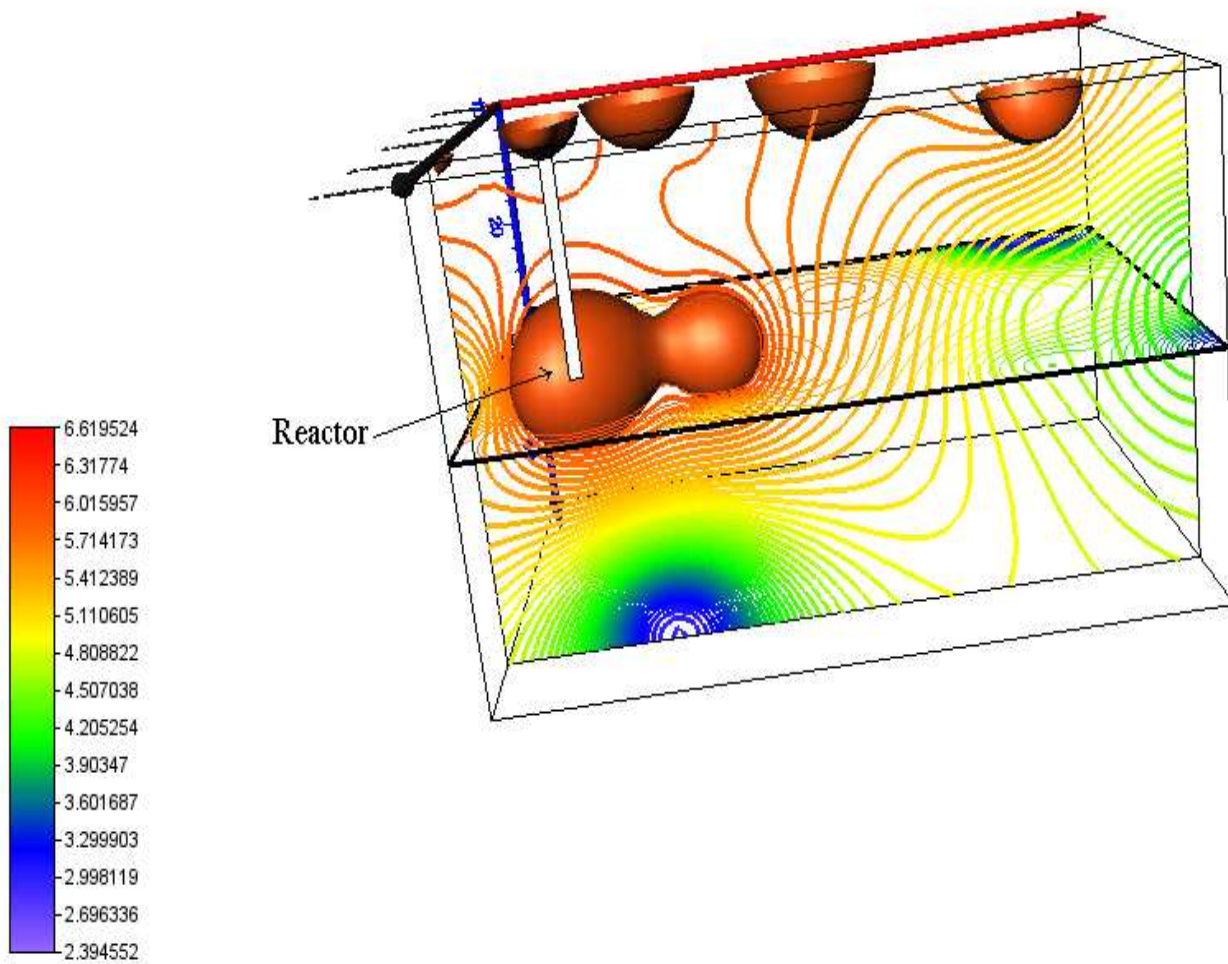


Figure 6: Three dimensional graph for the DO concentration

5.2 Simulation for the REDOX potential (ORP)

The reactor cell was placed at a depth of 4.3 m from the ground level. The concentration of dissolved oxygen (ORP) was of 670 mv in the reactor.

Figure 7 shows the contour lines of the DO concentration after 180 days obtained by simulation.

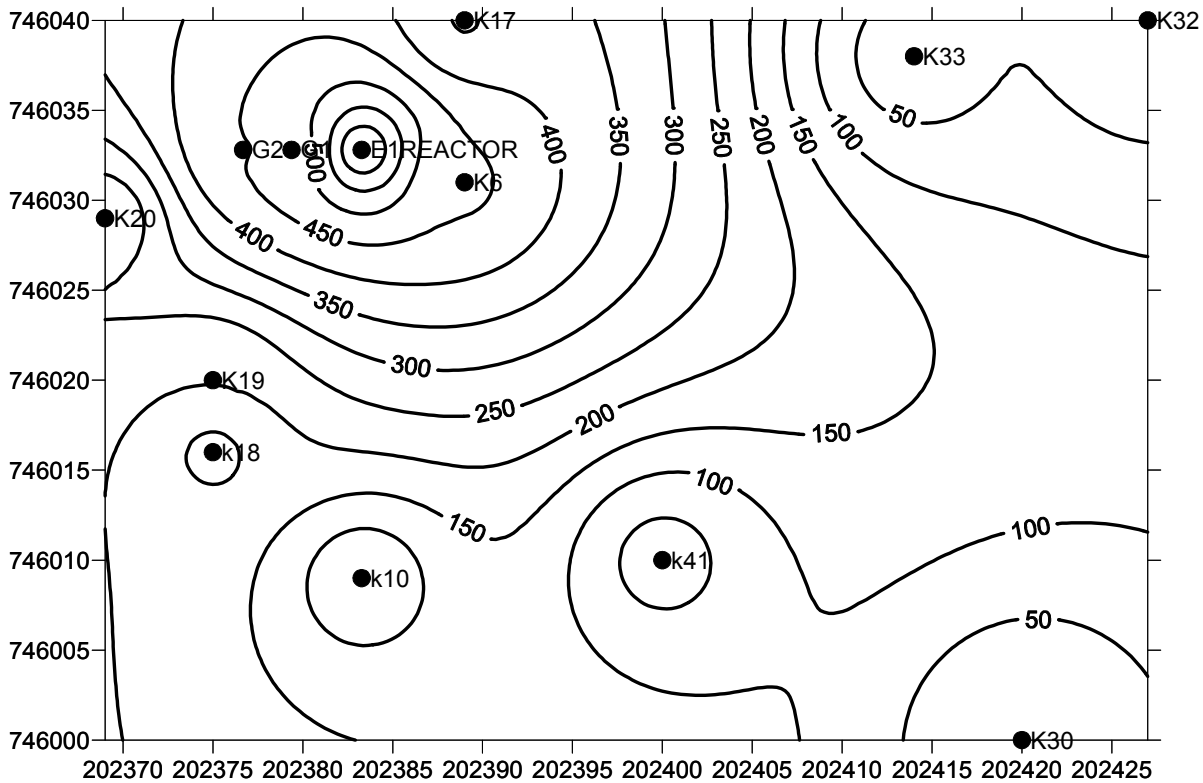


Figure 7: ORP concentration in the groundwater at a depth of 4.3 m.

In the following table, we estimate from the simulation results the radius of influence for different time steps.

Radius/time (days)	30	60	120	180
R(600)	4	7	16	20
R(360)	5	10	20	25

Table 3: Radius of Influence for different time for ORP simulation at 4.3m depth.

5.1.2. Influence of the position of the reactor

The reactor cell was placed at a depth of 1 m from the ground level. The concentration of dissolved oxygen (ORP) was of 670 mv in the reactor. Figure 8 shows the contour lines of the ORP concentration after 180 days obtained by simulation.

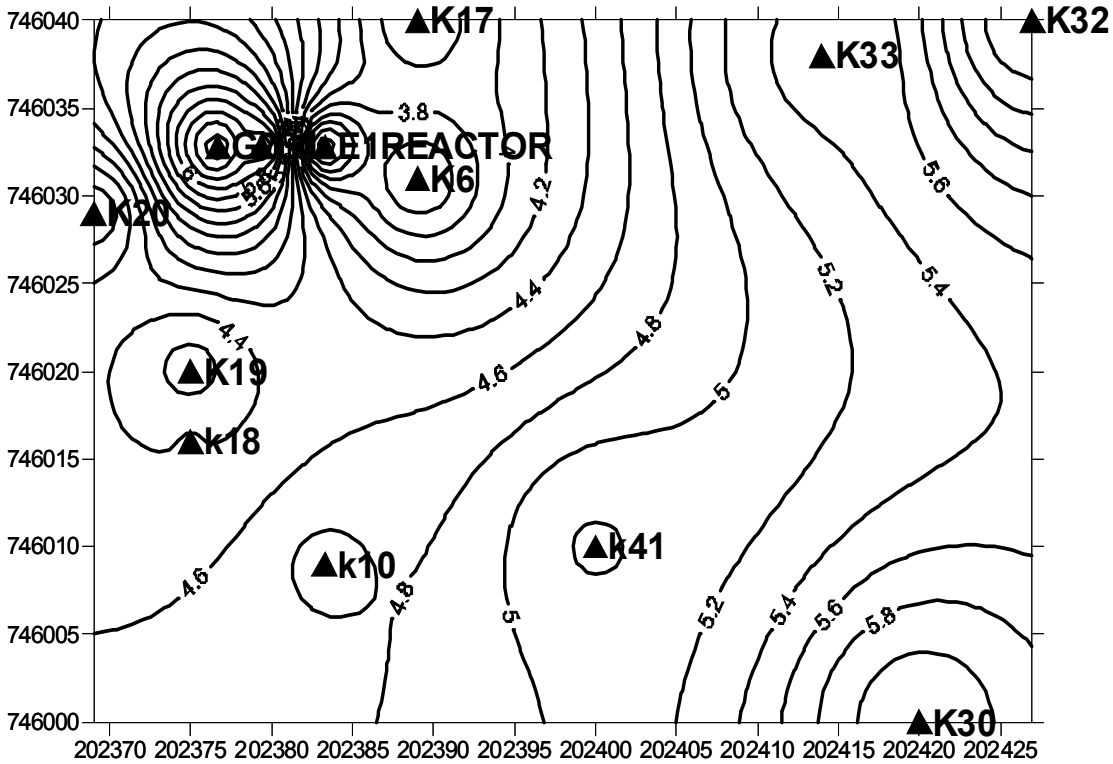


Figure 8: ORP concentration in the groundwater at a depth of 1 m.

In the following table, we estimate from the simulation results the radius of influence for different time steps.

Radius/time (days)	30	60	120	180
R(9ppm)	3	5	8	12
R(5ppm)	4	6	9	14

Table 4: Radius of Influence at different time for ORP simulation at 1m depth.

Figure 9 shows a three dimensional view of the radius of influence.

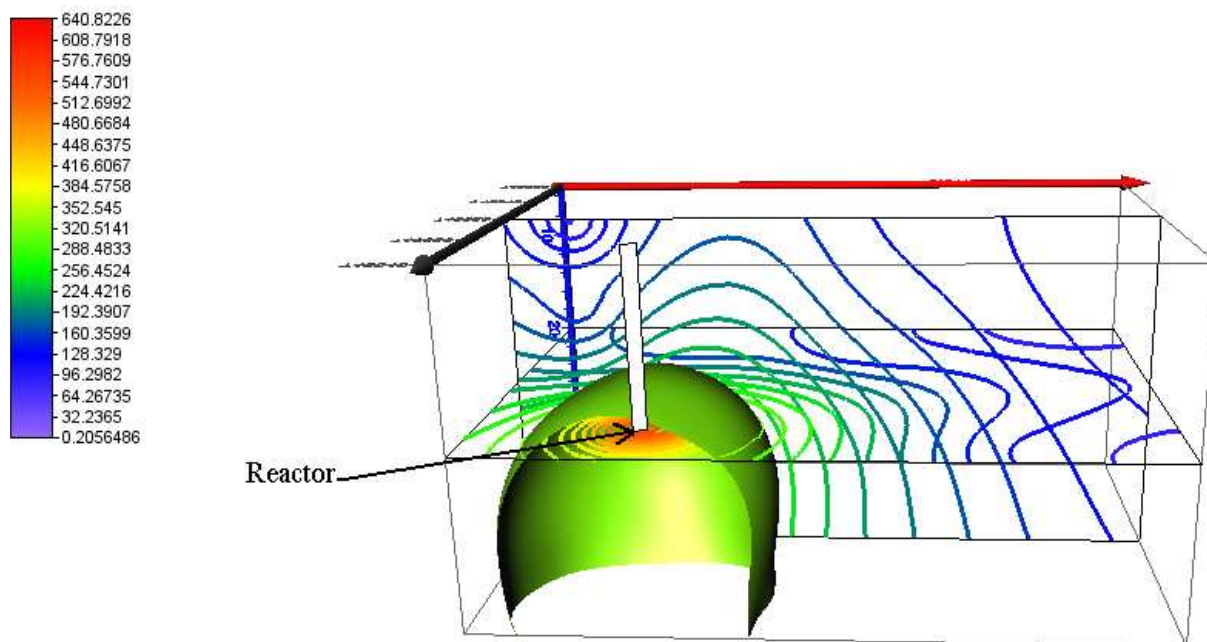


Figure 9: Three dimensional graph for the ORP concentration

6. Conclusion

In this contribution, we first present a theoretical basis for the analysis of the radius of influence on the concentration of dissolved oxygen (D.O.) and the Redox potential (ORP) for a general electro kinetics bioremediation system.

The simulation runs show clearly that first the rotation of the groundwater near the cell creates an homogeneous boundary condition at the reactor cell resulting with an increase in the radius of influence of 2 to 3 compared with simple oxygen diffusion system.

Moreover, the electric field created by an anode and cathode system increases the radius of influence by creating an additional flux in the charged water.

In order to validate this new technology, a pilot study was performed in the Haifa region Israel. The software was calibrated on data collected in a pilot study. An analysis of the radius of influence on data results shows the efficiency of the new technology.

In the saturated zone at a depth of 2.5 under the ground level, the different runs of the models show that the radius of influence for the ORP concentration in the water was around 30 m in the direction of the flow and 10 m in the other directions (See Figure 3).

For the DO concentration the radius of influence for the DO concentration in the water was around 15 m in the direction of the flow and 7 m in the other directions.

Moreover according to our calculation the EBR™ system enable us to clean the ground. In the unsaturated zone (or vadose zone) at a depth of 0.5 m under the ground level, the different runs of the models show that the radius of influence for the ORP concentration in the soil was around 13 m in the direction of the flow and 5 m in the other directions. For the DO concentration the radius of influence for the DO concentration in the water was around 7 m in the direction of the flow and 3 m in the other directions.

For initial concentrations of 12 ppm of DO and 850 mV of ORP, the radius of influence remains in the same order of magnitude in spite of the fact that the remediation is stronger in the vicinity of the EBR™ reactor. Also the position of the EBR™ reactor in depth, gave better numerical results when the EBR™ reactor is placed in the groundwater near the water level surface (around 1 m below the ground).

The results obtained in this pilot study, are relative to the type of soil in the studied area. The sandy clay found in the Haifa area seems to be appropriated for the EBR™ technology. However, this pilot study demonstrates the high level quality of the treatment done by the EBR™ technology. This technology performs ideal conditions for the attenuation of the biological and organic substances in the vadose zone and the aquifer.

4.

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