



Acid rock drainage in the uranium mining and milling site of Poços de Caldas, Brazil — duration assessment, pollutant generation modelling and remediation strategies

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Abstract. This geochemical modeling work was carried out to simulate the acid drainage generation from one of the waste-rock piles at the Poços de Caldas uranium mining site. The mathematical code STEADQYL was used. The estimated results were in good agreement for sulphate and uranium concentrations and the duration of the acid water generation was estimated to be about 500 years. The effect of covering the dump with a material that minimized oxygen diffusion was assessed. Projections indicated that covering the dump with a 1.0 m thickness of a material (like clay), which had an oxygen diffusion coefficient of $10^{-9}\text{m}^2.\text{s}^{-1}$, would reduce the pollutant concentrations to acceptable values. The estimated cost, when using this strategy, would be about US \$10 million.

1. INTRODUCTION

Open pit mines involve moving large amounts of overburden that are subsequently disposed in waste rock piles. If pyrite is present, acid drainage may result. The disposal site for the waste rock is typically determined by convenience and the accessibility of the site to the mining operation; usually the choice is not based on considerations that facilitate environmental management and remediation. If the mine operator has a fair understanding of how the design of the waste pile can mitigate acid drainage generation, improved layout and design of waste rock piles may be achieved.

The chemical composition of the acid drainage is controlled by the pyrite oxidation rate, which is a function of temperature, pH, oxygen concentration, composition and amount of infiltrating water and the population of micro-organisms [1]. The proportions of different oxidation and reaction zones in a waste rock pile will depend on the rate that the pile is being oxidized and also on the rate and pathways of water transport through the pile. The region under oxidation can encompass the whole dump or may embody a thickness of only few meters. Moreover, due to the heterogeneity of the material forming the pile, the oxidation regions may differ in size throughout the dump.

Oxygen transport into the dump is governed by three major processes:

- (1) Transport as dissolved O_2 in the infiltrating water;
- (2) Diffusion of O_2 gas through rock pore spaces and;
- (3) Advective transport of O_2 gas through rock pore spaces.

Because the leach solutions cannot carry significant oxidant with them as they move through the dump, air infiltration is the main source of oxidant. Diffusive transport is the major process of oxygen transport in the vast majority of cases. Oxygen removed from the dump pore space in the process of pyrite oxidation will establish a concentration gradient in the pore space. Despite the fact that gas removal will set up a pressure gradient, the gradient will be rapidly dissipated by mass transport of air.

Intrinsic Oxidation Rate (IOR) is a key parameter for understanding how oxidation is taking place in a dump. It is merely the rate of oxygen consumption by the material deposited in the dump under the conditions applicable to the particular case and is a function of a large number of variables including pore-gas oxygen concentration, particle-size distribution, mineral surface area, bacterial population, temperature, pH, and ferric ion concentration. If one considers the possibility of including the

functional dependence of the IOR on these variables, numerically intensive calculations will be generated.

The IOR in pyritic waste rock is a critical parameter for estimating the acid-drainage potential and its probable duration. The IOR parameter should be of a form that is physically reasonable and that allows the following goals to be achieved:

- (a) identification of the mechanisms controlling the global oxidation of the pyritic material in the pile;
- (b) planning the appropriate remedial actions;
- (c) assessing the rehabilitation schemes to be applied;
- (d) assessing the building construction techniques
- (e) quantification of pollutant loads, with emphasis on the peak values and the related time for its occurrence.

Values of IOR in the range of 10^{-8} to 10^{-9} $\text{kg (O}_2\text{)}\text{m}^{-3}\cdot\text{s}^{-1}$ are typical for a large number of waste rock piles [2]. Values greater than 10^{-7} can be considered extremely high, whereas values around 10^{-10} are related to marginal acid drainage environmental problems.

Temperature and oxygen profiles can be analysed both to define the oxidation rate in a region of pyritic material and to determine the dominant gas-transport mechanisms. With this information and data on bulk physical properties of the dump material, the primary pollution-production rate can be calculated; from this result an estimate can be made of the time dependence of pollution load in drainage from the base of the waste rock dump.

A sequence of lined drill holes can be used to obtain pore-gas oxygen concentrations and temperature profiles within the dump. Enough probe holes should be installed to obtain profiles through the various types of material present in the pile. In situ measurement of oxygen concentrations and temperature profiles in a waste pile is the best way to assess the IOR and subsequently predict the potential for generating acid drainage and its duration. It should be pointed out, however, that other techniques are available to deal with the problem.

The waste rock pile (WRP-4) of Poços de Caldas mining site, the object of the present study, was chosen because most of the infiltrating water is collected in a single holding pond. This situation represents a particularly unique opportunity that is not generally available and has not been previously exploited. This allows for checking the accuracy and internal consistency of the geochemical modelling predictions.

The objective of the present work was to study and investigate the rate and degree of acid drainage generation in the WRP-4 waste pile by means of geochemical modelling and to recommend possible remediation strategies.

The results from this work are expected to contribute to the study of problems associated with acid mine drainage prediction and management and should also support the decisions concerning the remedial actions to be implemented by the mining operator.

2. DESCRIPTION OF THE STUDY SITE

The uranium mill site and mine are located on the Poços de Caldas plateau, in the Southeast region of Brazil. The alkaline complex corresponds to a circular volcanic structure that began forming in the upper Cretaceous (87 ma) and evolved in successive steps until 60 ma. This intrusion is rounded by the levelling of bed rocks consisting of granites and gneisses. These rocks are frequently cut by diabase dykes, amphibolites and gneisses.

These igneous-polycyclic activities, of alkaline nature, associated with intense metassomatic processes and strong weathering, gave rise to a variety of rock types belonging to the Nepheline-Syenite family and to uranium mineralization.

The uranium enrichment in Poços de Caldas mine is related to hydrothermal events (primary mineralization) and to latter weathering processes (secondary mineralization).

The mine covers an area of about 2.5 km² and is divided into three mineralized units designated as ore bodies A, B and E for mining purposes.

The mining and milling facilities began commercial operation in 1982. However, the original intended production of 500 ton of U₃O₈ per year was never reached. As of 1995, 1172 tons of U₃O₈ were produced. During development of the mine 44.8 × 10⁶ m³ of rock were removed. From this amount, 10 million ton was used as building material (roads, ponds, etc). The rest was deposited into two major rock piles, waste rock pile 8 (WRP-8) and 4 (WRP-4). In contrast to WRP-8, all the drainage from WRP-4 is collected into a single holding pond.

3. RESULTS AND DISCUSSION

Geochemical processes can be divided into three groups based on their rates of reaction. Fast processes are assumed to proceed to equilibrium and can be modelled using equilibrium chemistry. Slow processes are assumed to proceed at a rate, which is *quasi-steady-state*, and therefore, can be modelled using chemical rate equations. Very slow processes are assumed to be slow enough that they have a negligible effect on the geochemistry.

The main purpose for using the kinetic modelling code, STEADYQL [3, 4] was to assess the value of the Intrinsic Oxidation Rate. Secondly, we aimed to develop a feeling for the equilibrium pollutant concentrations in the drainage. The model combines slow reaction kinetics with empirical rate laws and depicts rapid reactions as speciation equilibrium. The dissolution kinetics of primary and sulphide minerals are treated as irreversible processes far from equilibrium, and secondary phases are maintained in equilibrium with the aqueous phase. Speciation equilibrium is defined by the mass action equation where C_(i) is molar concentration of species i; K_(i) is the conditional stability constant; X_(j) is free concentration of component j and a_(i,j) is the stoichiometry coefficient j in species i:

$$C(i) = K(i) \prod X(j)^{a(i,j)} \quad (1)$$

The equation for defining the kinetics of a process in terms of its constituent components has the general form:

$$R = \frac{\partial C_A}{\partial t} = k[C_A]^a [C_B]^b \dots [C_N]^n \quad (2)$$

Where R is the rate of reaction, t is time and k is the rate constant C_N is the concentration of the n-th component and a,b,c...n are the stoichiometric coefficients. The rate of each process is converted to a flux, J, of material entering or leaving the system. The general form of the flux equation is given by:

$$J_{l,j} = s_{l,j} \prod_m P_m^{wl,m} \prod_i C_i^{mi,j} \quad (3)$$

Where J_{l,j} is the flux of component j due to process l (the units are mol per unit area per unit time), s_{l,j} is the stoichiometric coefficient of component j in process l,

$$\prod_m P_m^{wl,m}$$

Captures all modifying or normalising parameters for the J_{l,j} flux (these may include the rate constant, reactive area constraints, porosity and moisture content) and

$$\prod_i C_i^{m,j}$$

Captures all concentration terms related to the flux. Under the conditions of quasi-steady-state, the sum of influent and effluent fluxes of the system must balance.

The chemical equilibrium/steady state problem is solved iteratively as follows: 1) The free concentrations of the components at steady state are estimated; 2) The corresponding concentrations of all of the species are computed from the mass-action equation; 3) The fluxes are calculated taking into consideration the concentrations of the species; 4) The concentrations of the species and the fluxes are then substituted into the mole-balance equations; 5) If all of the difference functions for the components are equal to zero, the mole balance equations are satisfied, and the problem is considered to be solved; and 6) If some of the differences functions are not equal to zero, the value of the difference functions and their derivatives with respect to the free concentrations of the component, are used to calculate improved values for the free concentration of components with the Newton-Raphson method. The process is repeated with the improved values until the mole-balance equations are satisfied.

The reactive surface area of the waste rock, $A_{r,tot}$ [$dm^2 \cdot dm^{-3}$] was treated as a dependent variable. The reactive surface area for a specific mineral A_i , is assumed to be related to mineral volume abundance, v_i , and the calibrated total reactive surface area:

$$A_i = \chi_i A_{r,tot} [dm^2 / dm^3] \quad (4)$$

$$\chi_i = \frac{v_i}{\sum_{j=1}^n v_j} \quad (5)$$

$$A_r = \chi A_s 0.006 [dm^2 \cdot dm^{-3}] \quad (6)$$

Where the figure of 0.006 stands for the accessibility factor [5, 6]. Differences in grain sizes and surface roughness between different minerals that could effect the distribution of surface area are neglected.

Physical surface area is approximately constant per volume rock and independent of the particle size distribution [6]. This value would be approximately equal to $100 dm^2 g^{-1}$, with no dependence on the size of the fraction between 0.1 to 4.7 mm. One can consider that this range is greater than the geometric area of the particles.

At an estimated bulk density for the waste rock of $2.8 kg \cdot m^{-3}$ and an assumed porosity for the heap of 15% [7] the physical surface area is on the order of $2000 m^2 \cdot dm^{-3}$.

The infiltration rate [8] for the undisturbed rock at the mining area is $0.1 mm^3 m^{-1} year^{-1}$. This figure is 5% of the local precipitation rate. However, the dump surface is completely altered in relation to the original surface. It is suggested that 40 to 60% of the precipitation will infiltrate into the dump [9]. It is also reported [11] that infiltration in different dumps averages 50% of the total precipitation. Stromberg and Banwart [6] had reported an infiltration rate of 70% for a waste dump in Sweden. Since the infiltration rate for the current site is not known, we elected to use the value reported by these authors since the amount of water collected in the holding pond suggest the occurrence of high infiltration rates.

Based on the work of Cross et al. [7, 8] who used a kinetic approach to model the redox front movement in the Poços de Caldas mine and that of Nordstrom et al. [11] who modelled rock-water

interaction in the mine of Poços de Calda and Morro do Ferro, we estimated an average mineralogical composition for the WRP-4 waste pile (Table 1).

This mineralogical data could then be used, together with the kinetic rate and equilibrium constant data within the STEADYQL modelling code to estimate the composition of the drainage. The reactions considered in the modelling exercise and the associated expressions are shown in Tables 2 and 3 respectively.

TABLE I. ESTIMATED AVERAGE MINERALOGICAL COMPOSITION OF WASTE ROCKS IN WRP-4

Mineral	Volume Fraction
K-Feldspar	50%
Kaolinite	20%
Muscovite	20%
Fluorite	0.42%
Uraninite	0.12%
Pyrite	2%
Mn-Oxides	0.17%
Secondary Fe-Oxides	2%

The reaction constants and parameter terms (Tables 4 and 5 respectively) have been corrected by the Arrhenius expression to the estimated average temperature of the dump (45°C). The estimated equilibrium concentrations are given in Table 6 along with the average concentrations in the drainage from WRP-4.

TABLE II. REACTIONS CONSIDERED IN THE PRESENT STUDY

Mineral	Dissolution Reaction
K-Feldspar	$KAlSi_3O_8 + H^+ + 9/2 H_2O \rightarrow K^+ + 2H_4SiO_2 + 1/2 Al_2Si_2O_5(OH)_4$
Kaolinite	$Al_2Si_2O_5(OH)_4 + 6H^+ \rightarrow 2Al^{3+} + 2H_4SiO_4 + H_2O$
Muscovite	$2KAl_2[AlSi_3O_{10}](OH)_2(s) + 2H^+ + 3H_2O \rightarrow K^+ + 3Al_2Si_2O_5(OH)_4$
	$3Al_2Si_2O_5(OH)_4 + 18H^+ \rightarrow 6Al^{3+} + 6H_4SiO_4 + H_2O$
Pyrite by O ₂	$2FeS_2 + 7O_2 + 2H_2O \rightarrow 2Fe^{2+} + 4SO_4^{2-} + 4H^+$
Pyrite by Fe ³⁺	$FeS_2 + 14 Fe^{3+} + 8H_2O \rightarrow 15 Fe^{2+} + 2SO_4^{2-} + 16 H^+$
Oxidation of Fe ²⁺	$4 Fe^{2+} + O_2 + 4H^+ \rightarrow 4Fe^{3+} + 2H_2O$
Hematite	$Fe_2O_3 + 3H_2O \rightarrow 2Fe^{3+} + 6OH^-$

TABLE III. EXPRESSIONS FOR PROCESSES

Reaction	Rate Equation
Kaolinite	$J_{ka} = K_{ka} P_{ka} \text{ (mol dm}^{-2} \text{ s}^{-1}\text{)}$
Pyrite by O ₂	$J_{PyO} = K_{PyO} P_{PyO} [O_2]^{0.5} \text{ (mol}^{0.5} \text{ dm}^{0.5} \text{ s}^{-1}\text{)}$
Pyrite by Fe ³⁺	$J_{PyF} = P_{PyF} [Fe^{3+}] \text{ (dm s}^{-1}\text{)}$
Oxidation of Fe ²⁺	$J_{FeO} = K_{FeO} P_{FeO} [Fe^{2+}][O_2][OH^-]^2 \text{ (mol}^2 \text{ dm}^{-7} \text{ atm}^{-1} \text{ s}^{-1}\text{)}$
K-Feldspar	$J_{Kf} = K_{Kf} P_{Kf} \text{ (mol dm}^{-2} \text{ s}^{-1}\text{)}$
Hematite	$J_{He} = K_{He} P_{He} \text{ (mol dm}^{-2} \text{ s}^{-1}\text{)}$
Muscovite	$J_{Musc} = K_{Musc} P_{Musc} [H^+]^{0.37}$
J _{in}	$J_{in} = vC_{in} \text{ (mol dm}^{-3} \text{ s}^{-1}\text{)}$
J _{out}	$J_{out} = vC_{out} \text{ (mol dm}^{-3} \text{ s}^{-1}\text{)}$

TABLE IV. REACTION CONSTANTS

Reaction	Constant
Kaolinite	$K_{ka} = 1.8 \times 10^{-15} \text{ (mol dm}^{-2} \text{ s}^{-1}\text{)}$
Pyrite by O_2	$K_{PyO} = 2.3 \times 10^{-2} \text{ (mol}^{0.5} \text{ dm}^{0.5} \text{ s}^{-1}\text{)}$
Pyrite by Fe^{3+}	$P_{PyF} = 5.5 \times 10^{-9} \text{ (dm s}^{-1}\text{)}$
Oxidation of Fe^{2+}	$K_{FeO} = 9.6 \times 10^{-13} \text{ (mol}^2 \text{ dm}^{-7} \text{ atm}^{-1} \text{ s}^{-1}\text{)}$
K-Feldspar	$K_{Kf} = 1.6 \times 10^{-15} \text{ (mol dm}^{-2} \text{ s}^{-1}\text{)}$
Hematite	$K_{He} = 5.0 \times 10^{-14} \text{ (mol dm}^{-2} \text{ s}^{-1}\text{)}$
Muscovite	$K_{Musc} = 3.24 \times 10^{-14} \text{ (mol dm}^{-2} \text{ s}^{-1}\text{)}$

TABLE V. PARAMETRIC TERMS

Reaction	Parameter Term
Kaolinite	$P_{ka} = 7.90 \times 10^4$
Pyrite by O_2	$P_{PyO} = 1.32 \times 10^4$
Pyrite by Fe^{3+}	$P_{PyF} = 1.10 \times 10^1$
K-Feldspar	$P_{Kf} = 1.78 \times 10^5$
Hematite	$P_{He} = 1.38 \times 10^4$
Muscovite	$P_{Musc} = 7.39 \times 10^4$

TABLE VI. ESTIMATED CONCENTRATIONS X OBSERVED CONCENTRATIONS OF CHEMICAL SPECIES IN THE WRP-4 DRAINAGE

Chemical Species	Estimated Concentration, (ppm)	Observed Concentration, (ppm)
SO_4	943	1010 ± 234
Al	53	118 ± 41
Fe	340	-
SiO_2	25	-
K(*)	63	7.73 ± 1.07
pH	4.34	3.3 ± 0.13

The model's prediction of sulphate concentration is very close to what is actually observed. This observation suggests that the derivation of the IOR by this method will be a valid approach. However, the model underestimates Al concentration by a factor of 1.8 and overestimates the pH value as well as K concentration by one order of magnitude. The aluminium concentration in the drainage results from the dissolution of muscovite, K-feldspar and kaolinite. The total Al production rate was predicted as being $2.25 \times 10^6 \text{ g.ha}^{-1} \text{ yr}^{-1}$ with 72% of the Al load in the drainage coming from K-feldspar dissolution, which also accounts for 90% of the produced silica. The production of K is mainly maintained by K-feldspar dissolution and to a lesser extent by muscovite dissolution. Both processes are acid consuming reactions along with kaolinite dissolution. K-feldspar dissolution alone accounts for 61% of H^+ consumption. The model overestimates the pH by one pH unit that represents one order of magnitude in terms of H^+ concentration. Thus the K-feldspar dissolution may be in charge both for the overestimation of K concentration and the pH. It is possible that the process of K-feldspar is not being well represented by the model and thus must be calibrated. It is important to observe, however, that the inconsistency between the predicted and observed values of pH and K concentration does not impact the estimation of the sulphate production rate.

It has been shown [12] that the uranium variations in the acid drainage follow the same pattern as sulphate. If the observed ratio between uranium and sulphate in the acid drainage were applied to the estimated sulphate concentration, a value of 160 Bq.L^{-1} would be obtained which is very close to the average uranium activity concentration of 175 Bq.L^{-1} in the drainage of WRP-4.

Pyrite oxidation by atmospheric oxygen is responsible for 99% of the generated sulphate, implying that oxidation by ferric iron is negligible. This finding supports the assumptions made earlier during

the mass balance calculations. The model also estimates an oxygen consumption of 7.2×10^5 moles ha^{-1} year^{-1} . If this value is converted to IOR units a figure of 3.3×10^{-9} $\text{kg} \cdot \text{m}^{-3} \cdot \text{s}^{-1}$ is obtained.

Finally, the estimated sulphate production is equal to 4.12×10^5 moles $\text{ha}^{-1} \cdot \text{a}^{-1}$. This corresponds to a total production of 1.98×10^6 kg per year. It can be estimated that the total content of pyrite in WRP-4 is 6.20×10^8 kg. If this amount of sulphide is converted to sulphate stoichiometrically an amount of 1.0×10^9 kg would be produced. As a result, approximately 500 years would be necessary to consume all the sulfidic material in the dump.

One of the recommended remediation strategies is to cover the dump with a layer of an appropriate material that has an oxygen diffusion coefficient lower than the assemblage of rocks forming the dump. This would allow a reduction in the global oxidation rate by a significant factor. The reduction factor will depend to a very large extent on the cover properties rather than on the waste rock pile properties themselves [14]. The practical problem is to design a cover that remains relatively intact over a long period of time. The Global Oxidation Rate of the entity may be related to the cover thickness X_c and oxygen diffusion coefficient of the cover D_c by equation (7):

$$GOR = \sqrt{2C_0DS^*} (\sqrt{\alpha + n} - \sqrt{\alpha + n - 1}) \quad (7)$$

where:

$$\alpha = \sqrt{\frac{X_c}{D_c}} \left(\frac{S^* D}{2C_0} \right)$$

If it is required that the cover show adequate efficiency, α will have to be greater than unity. With the aid of equation 7 the sulfate concentration and load arising from the dump drainage was simulated for different cover thickness and oxygen diffusion coefficients. The infiltration rate was assumed to be equal to 50% of the total precipitation rate. The results are shown in Table 7. It can be observed that from the coefficient option 4 on, the reductions in the sulphate and uranium concentrations begin to be negligible.

It has been mentioned that the long term integrity of the covering system must be assured. As a result, a three layers cover system is usually employed. The system comprises a lower granular layer, an intermediate one of low gas permeability and an external one of sand and gravel to avoid the erosion of the system by rain and wind. The costs of a system like that adapted to the dimensions of the WRP-4 are represented in Table 8.

TABLE VII. EFFICIENCY OF THE DIFFERENT REMEDIATIONS APPYED TO THE DUMP

Dif. Coef. (m)	1×10^{-8} (1)	1×10^{-8} (2)	1×10^{-8} (3)	1×10^{-9} (4)	1×10^{-9} (5)	1×10^{-9} (6)	1×10^{-10} (7)	1×10^{-10} (8)	1×10^{-10} (9)
Thick.s. (m)	0.5	1.0	2.0	0.5	1.0	2.0	0.5	1.0	2.0
GOR $\text{kg/m}^2\text{s}$	1.8×10^{-8}	2.2×10^{-10}	1.1×10^{-10}	2.0×10^{-9}	1×10^{-9}	2.0×10^{-11}	2.0×10^{-10}	1.0×10^{-10}	5.0×10^{-11}
Load t/ano	139	75	38	15	7.7	3.9	1.8	0.76	0.34
[SO ₄] mg/l	287	154	78	32	16	8.0	3.7	1.58	0.79
[²³⁸ U] Bq/L	49	26	13	5.4	2.7	1.3	0.63	0.27	0.13

TABLE VIII. COVERING SYSTEM ASSOCIATED COSTS

	US\$/m ²	m ²	Total (US\$)
Lower Layer	3.5	56.5 × 10 ⁴	1.98 × 10 ⁶
Intermediate Layer (low permeability)	11	56.5 × 10 ⁴	6.22 × 10 ⁶
External Layer (gravel and sand)	3.0	56.5 × 10 ⁴	1.70 × 10 ⁶
Total			9.9 × 10 ⁶

It can be seen that approximately US\$ 10 million would be necessary for the installation of a 3-layer covering system. Other possibilities were considered such as covering with a plastic material or the backfilling the open pit with the waste rock. The first option would cost US\$ 12.7 million while the second one would amount to about US\$ 70 million. It must be emphasized that these cost refer to only one of the existing waste rock piles at the site.

4. CONCLUSIONS

Acid drainage generation is a long term problem at mining sites where sulfidic material is present in the rock. We demonstrated that geochemical modelling could predict the rates of pyrite oxidation and concentrations of pollutants in the acid drainage. It was also demonstrated that pyrite oxidation is mostly through reaction with oxygen and that oxidation by Fe³⁺ is of minor importance.

Remediation should concentrate on the reduction of oxygen diffusion into the dump. This may be attained by covering the dump with some sort of material (e.g. clay or compacted clay) that has a lower coefficient of diffusion for oxygen and lower permeability to water than assemblage of the rocks at the surface of the dump. Future monitoring of the dump should aim to improve understanding of the actual infiltration rates applicable to the site and of the distribution of oxygen in the dump. These data may be obtained by installation of lysimeters (work in progress) and installation of oxygen probes to measure oxygen concentration at different depths in the dump (work to be developed).

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REFERENCES

- [1] RITCHIE, A.I.M., The Waste-Rock Environment, In: Short Course Handbook on Environmental Geochemistry of Sulfide Mine-Wastes. Eds J.L.Jambor, D.W.Blowes, Mineralogical Association of Canada, Ontario, p. 133–161 (1994).
- [2] RITCHIE, A.I.M., Application of Oxidation Rates in Rehabilitation Design. In: Second Australian Acid Mine Drainage Workshop. Eds. N.J. Grundon, L.C. Bell, Charles Town, p. 101–116 (1995).
- [3] FURRER, G., WESTALL, J., The Study of Soil Chemistry Through Quasi-Steady-State Models: I. Mathematical Definition of the Model. *Geochimica Cosmochimica Acta*. Vol. 53, p. 595–601 (1989).
- [4] FURRER, G., SOLLINS, P., WESTALL, J.C., The Study of Soil Chemistry through Quasi-Steady-State Models: II. Acidity of Soil Solution. *Geochimica Cosmochimica Acta*. Vol 54, p. 2263–2374 (1990).

- [5] WHITE, A.F., PETERSON, M.L., Role of Reactive-Surface-Area-Characterization in Geochemical Kinetic Models. In: Chemical Modelling in Aqueous Systems II. Eds. D.C. Melchior, R.L. Bassett. 461–475 (1990).
- [6] STRÖMBERG, B., BANWART, S., Kinetic Modelling of Geochemical Processes at the Aitik Mining Waste Rock Site in Northern Sweden. *Applied Geochemistry* 9. p. 583–595 (1994).
- [7] CROSS, J.E., HAWORTH, A., NERETNIEKS, I., SHARLAND, S.M., TWEED, C.J., Modelling of Redox Front and Uranium Movement in a Uranium Mine at Poços de Caldas. *Radiochimica Acta*. 53/53, p. 445–451. (1991).
- [8] CROSS, J.E., HAWORTH, A., LICHTNER, P.C., MACKENZIE, A.B., MORENO, L., NERETNIEKS, I., NORDSTROM, D.K., READ, D., ROMERO, L., SCOTT, R.D., SHARLAND, S.M., TWEED, C.J., Testing Models of Redox Front Migration and Geochemistry at the Osamu Utsumi Mine and Morro do Ferro Analogue Study Sites, Poços de Caldas, Brazil. Swedish Nuclear Fuel and Waste Management Co. SKB Technical Report 90-21. p. 79 (1991).
- [9] MORIN, K., HUTT, N., An Empirical Technique for Predicting the Chemistry of Water Seeping from Mine-Rock Piles. In: International Land Reclamation and Mine Drainage Conference on the Abatement of Acid Drainage, Pittsburgh, (1994).
- [10] HARRIES, J.R., Acid Drainage from Waste Rock Dumps at Mine Sites. Technical Report ANSTO/E692, Australian Nuclear Science and Technology Organisation (1990).
- [11] NORDSTROM, D.K., PUIGDOMÈNECH, I., MCNUTT, R.H., Geochemical Modelling of Water-Rock Interactions at the Osamu Utsumi Mine and Morro do Ferro Analogue Study Sites, Poços de Caldas, Brazil Swedish Nuclear Fuel and Waste Management Co. SKB Technical Report 90-23. p. 733 (1990).
- [12] FERNANDES, H.M., Subsídios ao Descomissionamento ad Primeira Indústria de Mineração e Beneficiamento de Urânio no Brasil – O Caso do Complexo Mínero Industrial de Poços de Caldas – Niterói. (D.Sc. Thesis) Departamento de Geoquímica da Universidade Federal Fluminense. p. 250. (in Portuguese) (1997).