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A RISK ASSESSMENT APPROACH FOR EVALUATING THE ENVIRONMENTAL SIGNIFICANCE THE TRANSPORT OF RADIOACTIVE CONTAMINANTS FROM DISPOSAL SITE TO AQUATIC SYSTEM

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ABSTRACT

Before that hazard assessment approach can be applied for solid wastes, an evaluation must be made of the transport and transformation of the contaminants in the solid waste leachate en route to an aquatic system, in order to derive the expected concentrations and forms of the contaminants of interest entering the aquatic system.

Contaminants in aquatic systems may be removed from the water column by sorption into the sediment bed of the system, or they may be added to the water column by desorption from the sediments.

KEYWORDS:
sorption, radioactive contaminants, waste rock, groundwater, clay dispersion

1. INTRODUCTION

The existence of natural barriers that can isolate radionuclides from the environment and the log term stability of these barriers are the principal reasons for the geological disposal of radioactive waste.

The characterization of a potential disposal site is directed at establishing the favorable and potentially adverse aspects of the geological environment as they relate to the performance of the natural barriers.[1]

The key characteristics of the geological environment that need to be considered in determining the efficacy of natural barriers in isolating waste from the biosphere are as follows:

- physical isolation and stability;
- hydro geological transport processes;
- -geochemical conditions and processes.

Transport by groundwater is the principal mechanism for the migration of radionuclides to the biosphere, and therefore disposal

concepts generally rely on the existence of a geological environment in which the fluxes and velocities of groundwater in the vicinity are sufficiently small to provide adequate isolation. [2]

Contaminants in aquatic systems may be removed from the water column by sorption into the sediment bed of the system, or they may be added to the water column by desorption from the sediments. [3]

Thus, sediments can act to ameliorate a short-term problem by reducing aqueous concentrations but may increase the duration of aqueous contamination through the slow release of contaminants back into the water.

Sorption – desorption processes appear to be controlled by the affinity of the contaminant for the sediment (distribution coefficient, Kd), which varies as a function of the complex parameters, and by the rate of delivery of the contaminant to and away from the water sediment interface, which should vary as a function of water turbulence, velocity, contaminant concentration and diffusion of the contaminant through the sediment bed.

2. MATERIALS AND METHODS

We tried to study the extractive power of the natural water in the existing depositing sites, depending on the water amount collected by washing the samples and the samples weight until there are no mobilized radionuclides left.

We admit the hypothesis that in a percolation process in situ, there is always an equilibrium between the solubilized species and the species associated to an ore, both for radionuclides and the salt ensemble that make up the load of percolation water.

A dynamic experimental model was imagined to simulate the lixiviation phenomena, the extractive power of the natural water in the existing depositing sites, depending on the water amount collected by washing the samples and the samples weight until there are no mobilizable radionuclides left.

The technical method used, correspondingly for followed standard conditions:

- geological material: 900 g clay extracted from the stratum situated in contact with the free aquifer;
- radioactive material: 500 g of waste rock from the dump site in Ciudanovita area;
- the volume of elution water: 500 cm^3 – elution water / 1 month;
- time for elution: 3 years;
- were determined by measuring the alpha radioactivity of the Ra sulfate precipitate, separated radio-chemically, together with the Ba and Pb sulfates. The efficiency of the precipitation method and of the measurement assembly is 0, 1830 imp/dez.
- We highlighted the U_{nat}, after the chemical processing, with the color reaction, using a PAN indicator (1.2-pyridil-azo-2-naftol)

and afterwards we measured its radioactivity. The chemical efficiency of this method is 83%.

The alpha measurements were made with a counting assembly FHT 1100 with a proportional counter with gas circulation and relative efficiency of 26%.

3. RESULTS AND DISCUSSIONS

The obtained results, with the variations of different parameters are presented in figures 1 - 3.

1.The time of contact:

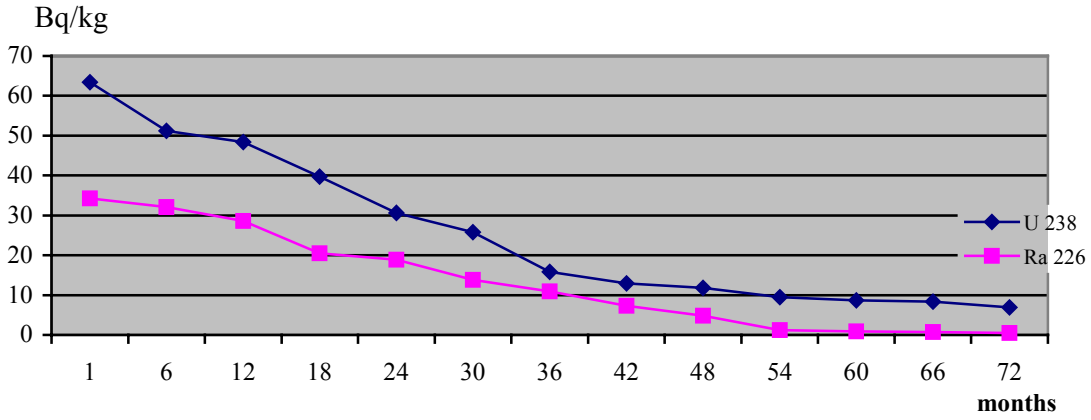

Fig. 1 – The concentration of ^{238}U and ^{226}Ra in elution water

The concentration of ^{226}Ra and ^{238}U in the solution of elution represented in fig.1 reach the highest values in the first year of the experiment for each other radionuclides but decrease in time until the approximate constant values.

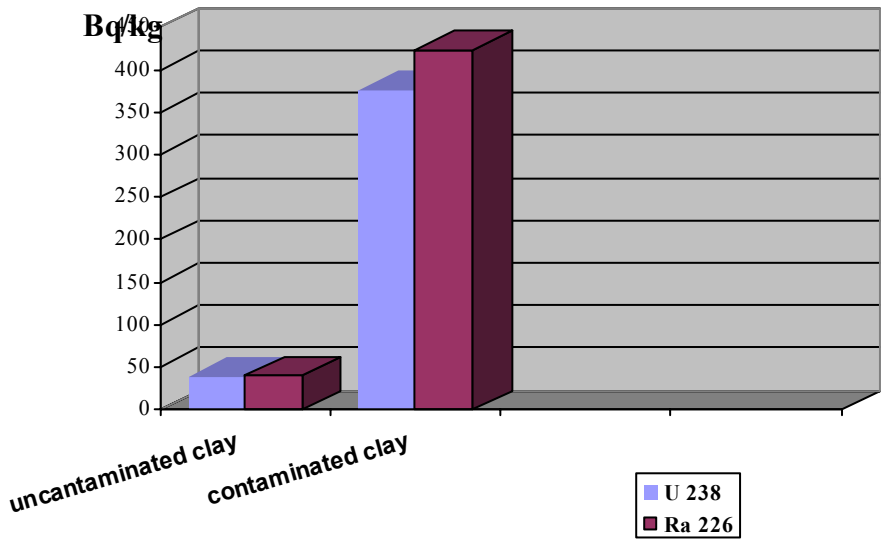

Fig. 2 - The concentration of ^{238}U and ^{226} Ra in the clay after 3 years

The concentration of ^{226}Ra and ^{238}U in clay represented in fig. 2, has a considerable growth in time.

2. The pH influence:

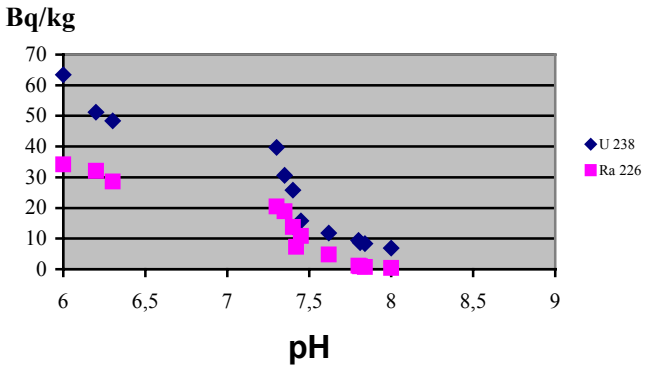

Fig. 3 – The concentration of ^{238}U and ^{226}Ra in elution water in function of pH

In fig.3 it can see how the desorption of the radionuclides from the waste rock in the elution water is higher, for a pH=6-7,5.

The literature contains much information on migration of radionuclides through the groundwater systems from various sources. Generally, these studies tend to be site specific and examine only a few radionuclides.[2]

In geological media, argillaceous sediments for example, the low hydraulic conductivity, lack of water-conducting features and immobility of colloids means that transport is dominated by diffusion of solutes, with retardation by sorption .

The use of the distribution coefficient (Kd) was considered the best approximation to the estimation of the degree of radionuclides sorption in geological material, taking into account the great number of its limitations, associated with it character of empirical parameter of a natural system and, therefore, with several factors that affect it. [4].

The distribution coefficient, definite as follows:

$$K_d = \frac{concentrationofradionuclide\sin solidephase}{concentrationofradionuclide\sin aquoesphase}$$

Represents a constant of equilibrium and includes a complex number of reactions or geochemical and biochemical processes, generally named sorption. [3]

Due to the large number of variables that affect the determination of the sorption of nuclides in geological materials, it is impossible to control all of them in a unique experience.

Owing to this circumstance, it appears as a useful parameter the Kd coefficient, measured in similar conditions to the media in study, varying successively the essay conditions. [5]

The experimental estimation of the Kd value is calculated as:

$$Kd = \frac{A_s / M}{A_l / V},$$

where: A_i =initial activity in the contact solution

A_f = final activity in the contact solution

V = volume of the contact solution

P = weight of the geological material

The results of measurements Kd taken in different conditions are showed in figures 4 and 5.

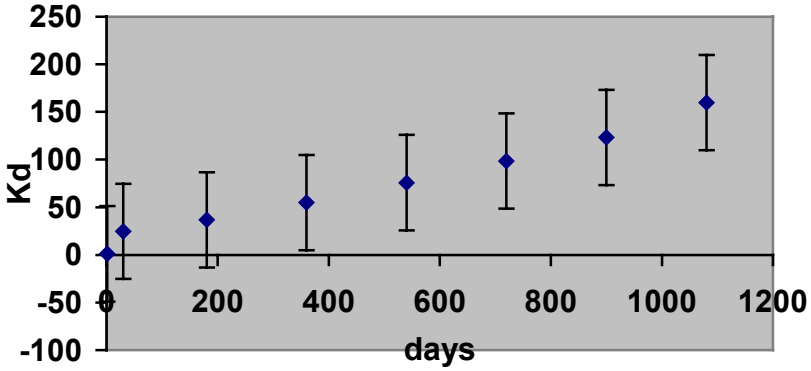

Fig.4. Variation of Kd the distribution coefficient for ^{238}U in function of time in the clay.

The obtained values of Kd are considerate higher when the ratio liquid/solid was decreased for each other radionuclides.

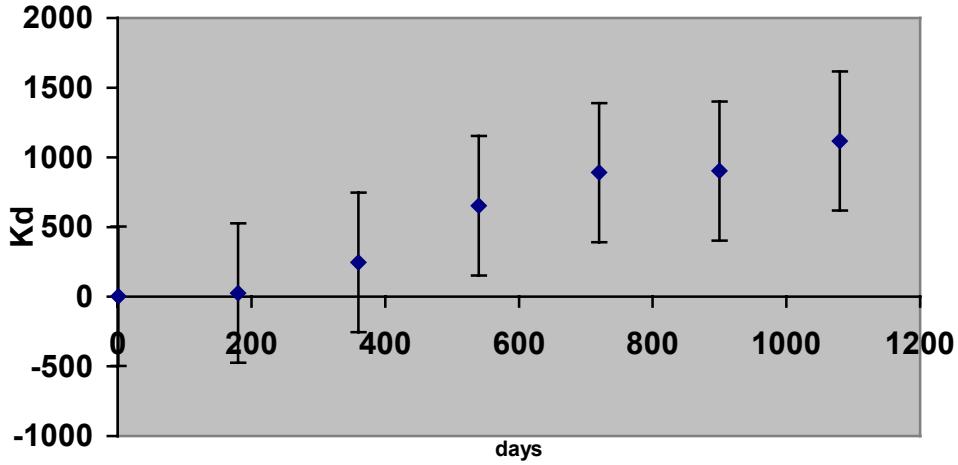

Fig.5- Variation of Kd the distribution coefficient for ^{226}Ra in function of the time in the clay

4. CONCLUSIONS

The soils with a higher level of clay have many places of cations exchange and have a better adsorption of radionuclides.

If the waste rock dump is used for the isolation of radioactive waste, intruding ground waters and rain waters could lead to dissolution of

radionuclides and their subsequent transport to the biosphere. Interaction with the geologic media is one of the most important factors retarding such transport. It is therefore necessary to understand the many interrelated factors and processes that govern radionuclide transport.

The clay has expansive area for cations exchange and so is adsorbing more efficient ^{226}Ra and ^{238}U which exist as tied cations in these places.

The dynamic moved pollutants by movement of water are generally slow. A little fraction of ^{226}Ra and ^{238}U from wasted dump is liable to be mobilized.

The slow movement of these phenomenons implies the fact that is very important to introduce a scale of time in the ratiocination about the evaluation of the impact of stock-piling of the radioactive waste rocks in the abandoned waste dump. [6]

If the observations actually made near to the places cannot give the direct extrapolated information on the long time term, they are though un avoidable to verify the level of actual damages.

Due to these considerations results that is not representative to inform a unique Kd value for a given media, and consequently, it has to be expressed as a range of value. This range is related with the uncertainty on the variation of physical, chemical and geological parameters that modify the results.

Results that the variation domain of Kd for clay is between 1-159 cm^3/g for ^{238}U and 3-1170 cm^3/g for ^{226}Ra. The same values could be used in the sensible analyses of mathematics models of migration because of its influence which is important when the nuclides have a long life term.

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