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WASTE PACKAGE MASS TRANSFER RATES IN A GEOLOGIC REPOSITORY

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ABSTRACT

The rate of dissolution of low-solubility species from a waste package surrounded by a porous medium can be predicted by calculating the diffusive-convective mass transfer from the surface liquid into the groundwater, if the boundary concentration of the species dissolved in the surface liquid is known. The rate of liquid-solid interaction is expressed in terms of this boundary concentration and experimentally determined rate constants for the liquid-surface reaction, and the derived rate of diffusive-convective mass transfer in the porous rock is expressed in terms of the boundary concentration gradient and diffusive properties. By equating these two rates, the boundary concentration can be determined. Equations for the time-dependent approach of the boundary concentration to the saturation concentration are developed. A dimensionless group specifies the conditions under which the mass transfer rate can be suitably estimated by assuming saturation concentration at the boundary, with sufficiently rapid solid-liquid interaction so that dissolution rate is controlled by diffusive transport in the porous medium surrounding the waste package. These conditions are predicted to occur for silica in borosilicate glass waste in a repository. They are expected to apply to the actinides and many of the long-lived fission products in glass and spent-fuel waste packages in a repository. For borosilicate glass in contact with saturated rock, surface concentrations near saturation are predicted to be reached in a few days or less for silica and within a few months for cesium. The resulting equations can be used to prescribe laboratory experiments to verify these predictions.

INTRODUCTION

Groundwater flow around a waste package

We have earlier presented analytical (nonempirical) equations that predict the rate at which radionuclides in liquid at the surface of a waste package can be transported into groundwater in the surrounding porous medium [1-4]. These analytical solutions were derived by Chambre' from the time-dependent equations of diffusive-convective mass transfer of dissolved species from the surface liquid into groundwater in the surrounding rock. The flow and concentration fields expected around a waste package are shown in Figure 1. The analytical solutions are exact for this flow field. We found that the expected groundwater velocities in some repositories are so low that the best estimate of the mass-transfer rate from the surface

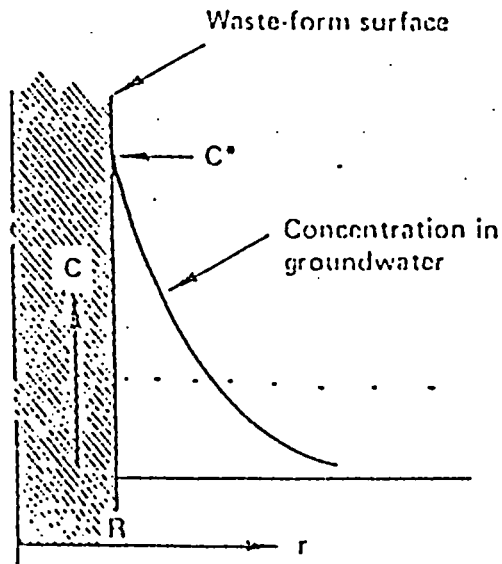
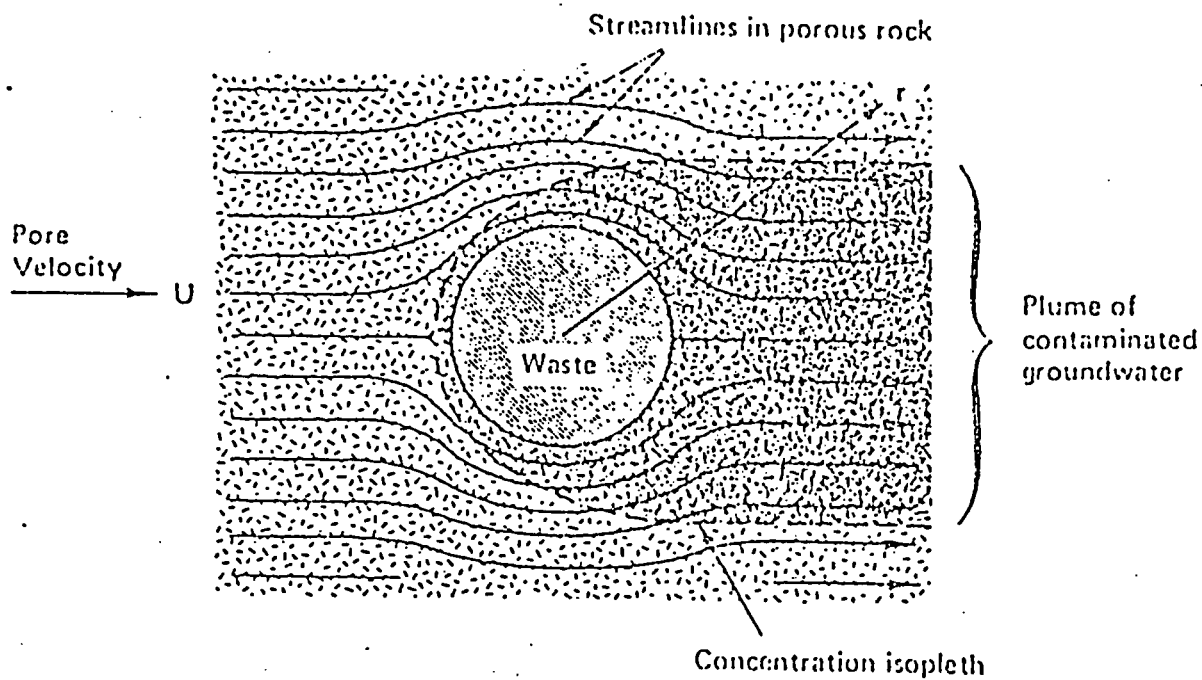


Fig. 1. Velocity and concentration profiles for groundwater flowing across a waste cylinder.

can be obtained from Chambre's diffusive-transport solutions that neglect groundwater motion. This means that to calculate rates of mass-transfer from the waste package at these low groundwater velocities, molecular diffusion is far more important than is convective transport. However, once the dissolved species have diffused into groundwater in the surrounding rock, convection remains important in determining the transport of dissolved species from the vicinity of the waste packages into the near and far fields.

Saturation concentration at the waste surface

The rate of mass transfer by diffusion from the surface liquid is emphasized in these analyses because it is found to be the phenomenon that controls the net rate at which most of the species in a waste solid dissolve. This was determined by conservatively assuming saturation concentration in the surface liquid to compute mass-transfer rates and by comparing those rates with the much larger rates deduced from laboratory experiments where solid-liquid reactions are controlling. The effect of the low rate of diffusion into the surrounding porous or fractured medium is to cause a rapid build-up of the concentration of the dissolved species in the surface liquid, and as saturation of each species is approached the net rate of solid-liquid reaction decreases to match the slow rate that the dissolved species can diffuse from the surface liquid.

The diffusion process

The diffusion process that we are analyzing is not the diffusion through surface layers of precipitates and alteration products, as inferred by some [5] from laboratory leach experiments; we are analyzing the molecular diffusion of the dissolved species through the groundwater continuum in the tortuous pathways formed by pores and fractures in the surrounding rock. Chambre's first analytical solutions applied to a waste solid in contact with porous rock, but later solutions have included a backfill layer of crushed rock or clay between the waste solid and rock.

Processes affecting dissolution in a repository?

Extensive data are available on the rate of dissolution of waste-solid samples in laboratory experiments, where dissolution rate is controlled by the rate of solid-liquid reactions. Techniques have been proposed to extrapolate the laboratory leach data to predict dissolution rates in a repository environment, with the unjustified assumption that solid-liquid reactions will control the dissolution rate in a repository [6,7]. In the new analysis presented here we make no assumptions about which process is controlling. We solve the governing equations for the time-dependent mass transfer (dissolution) rate from the surface of a waste solid into surrounding saturated porous media, using the experimentally determined surface-liquid reaction rates as a boundary condition for predicting the actual concentration of the dissolved species in the exterior field and at the waste surface. This results in a more realistic estimate of the dissolution rate

than has been calculated heretofore from the mass-transfer equations by assuming saturation concentration in the surface liquid. The resulting analytical solutions can be used to determine the conditions under which either chemical reaction rate or exterior-field diffusion controls mass transfer (dissolution).

Topics to be Addressed

In the following sections the new equations for diffusional mass transfer with the reaction-rate boundary condition are derived, a general formulation of the results is presented, the results are illustrated for silica and other constituents in borosilicate glass, and the results are compared with observations on natural minerals.

DERIVATION OF THE THEORY

We assume that a spherical waste solid is surrounded by porous rock containing groundwater, with groundwater pore velocity low enough that mass transfer through the porous rock is controlled by molecular diffusion of dissolved species through the tortuous liquid pathways in the rock. Neglecting radioactive decay, the governing equation for diffusive mass transfer from the spherical waste solid is:

$$K \frac{\partial C}{\partial t} = D \frac{1}{r^2} \frac{\partial}{\partial r} \left(r^2 \frac{\partial C}{\partial r} \right), \quad t > 0, \quad r_0 < r < \infty \quad (1)$$

with the initial condition:

$$C(r, 0) = 0, \quad r_0 < r < \infty \quad (2)$$

where $C(r, t)$ is the concentration of dissolved species in groundwater, K is the retardation coefficient, D is the diffusion coefficient in the liquid, r is the radial distance from the waste center, r_0 is the radius of the waste form, and t is the time.

As a boundary condition at the waste-liquid interface, we specify that the diffusive current (mass-transfer rate) of dissolved species at the waste outer surface is equal to the net rate of dissolution by the solid-liquid reaction:

$$-\epsilon D \frac{C(r_0, t)}{\partial r} = j_0 \left[1 - \frac{C(r_0, t)}{C_s} \right], \quad t > 0 \quad (3)$$

Here ϵ is the porosity of the surrounding medium, C_s is the saturation concentration of the dissolved species, and j_0 is the experimental forward reaction rate of that species, per unit external surface area. The forward reaction rate is measured when the surface is in contact with a well-mixed liquid that contains none of the dissolved species being considered in Equation (1). The form of the right hand side of Equation (3), which meets the expected condition of zero dissolution rate when the solution is saturated, is suggested by experiments on dissolution of quartz [8-9] and implies a first-order solid-liquid reaction. As shown later, it is a good approximation for

the concentration-dependent dissolution of silica from borosilicate glass [12-14], and it is assumed to apply to other species in the waste.

The remaining boundary condition is:

$$C(\infty, t) = 0, \quad t > 0 \quad (4)$$

From the solution to the above set of equations we obtain the time-dependent concentration $C(r_o, \tau)$ in liquid adjacent to the outer surface of the waste:

$$C(r_o, \tau) = C_s \frac{R(1 - e^{-\tau} \operatorname{erfc}\sqrt{\tau})}{1 + R} \quad (5)$$

and the time-dependent dissolution rate $j_o(r, \tau)$ at r_o :

$$j(r_o, \tau) = j_o \frac{1 + R e^{-\tau} \operatorname{erfc}\sqrt{\tau}}{1 + R} \quad (6)$$

where the dimensionless time τ is defined as:

$$\tau \equiv \frac{(1 + R)^2 D t}{K r_o^2} \quad (7)$$

The dimensionless "flux ratio" R is defined as:

$$R = \frac{j_o r_o}{\epsilon D C_s} \quad (8)$$

and can be interpreted as:

$$R = \frac{\text{forward reaction rate per unit area at } r_o}{\text{steady-state diffusive mass transfer rate at } r_o} \quad (9)$$

We can interpret the forward reaction rate j_o in terms of a reaction-rate constant k :

$$j_o = k C_s \quad (10)$$

which results in:

$$R = \frac{k r_o}{\epsilon D} \quad (11)$$

Thus R has the significance of the square of a modified Thiele modulus.

At steady state the concentration and mass-transfer rate at the surface are:

$$C(r_o, \infty) = C_s \frac{R}{1 + R} \quad (12)$$

and

$$j(r_o, \infty) = \frac{j_o}{1 + R} \quad (13)$$

Equation (12) shows that when the flux ratio R is large, $C(r_o, \infty) \approx C_s$, and by Equation (13) $j(r_o, \infty) \approx \epsilon D C_s / r_o$. Under these conditions the net dissolution rate is controlled by molecular diffusion in the exterior field.

When $R \ll 1$, Equation (12) shows that the surface concentration $C(r_o, \infty)$ is much less than the saturation concentration, and Equation (13) shows that the net mass-transfer rate is controlled by the solid-liquid reaction and is equal to j_o . For intermediate values of R one must use Equations (12) and (13).

Another quantity of interest is the time t_{ss} necessary for the mass flux at the surface to reach within five percent of the steady-state value. From Equation (6) it can be determined that:

$$t_{ss} = \begin{cases} 0 & \text{for } R \leq 0.01 \\ \frac{10^4 K r_o^2}{\pi D} & \text{for } R \gg 1 \end{cases} \quad (14)$$

Chambre' has shown [15] that if radioactive decay is included in Equation (1), the above conclusions are affected by both decay and retardation. The analysis of the surface concentration and dissolution (mass transfer rate), which includes the effects of species decay within the waste form, has also been treated.

The normalized dissolution (mass-transfer) rate $j(r_o, \tau)/j_o$ and the normalized surface concentration $C(r_o, \tau)/C_s$ are shown in Figure 2 as a function of the dimensionless time τ and the dimensionless parameter R .

APPLICATION TO SILICA AND CESIUM IN BOROSILICATE GLASS

To illustrate, we assume a waste glass cylinder of radius 0.15 m and length 2.4 m, resulting in an equivalent sphere of radius 0.44 m. From the laboratory leach data of Pederson et al. [12] for PNL 76-68 borosilicate glass we derive the following effective values for silica at 90°C:

$$j_o = 1.18 \text{ g SiO}_2/\text{m}^2\text{day}$$

$$C_s = 200 \text{ g SiO}_2/\text{m}^3$$

According to van Lier et al. [9], quoting Jander and Jahr [16], the room-temperature diffusion coefficient of the silicic-acid solute in water is $1.9 \times 10^{-2} \text{ m}^2/\text{yr}$. Correcting for temperature [17], we obtain $D = 7.7 \times 10^{-2} \text{ m}^2/\text{yr}$ at 90°C. The reduction of D because of tortuosity in the rock is conservatively neglected.

Assuming a rock porosity of 0.01, we estimate:

$$R_{\text{SiO}_2} = 1240$$

Assuming a retardation coefficient of unity for silica in the rock, a time to steady state is estimated:

$$t_{ss} = 320 \text{ years}$$

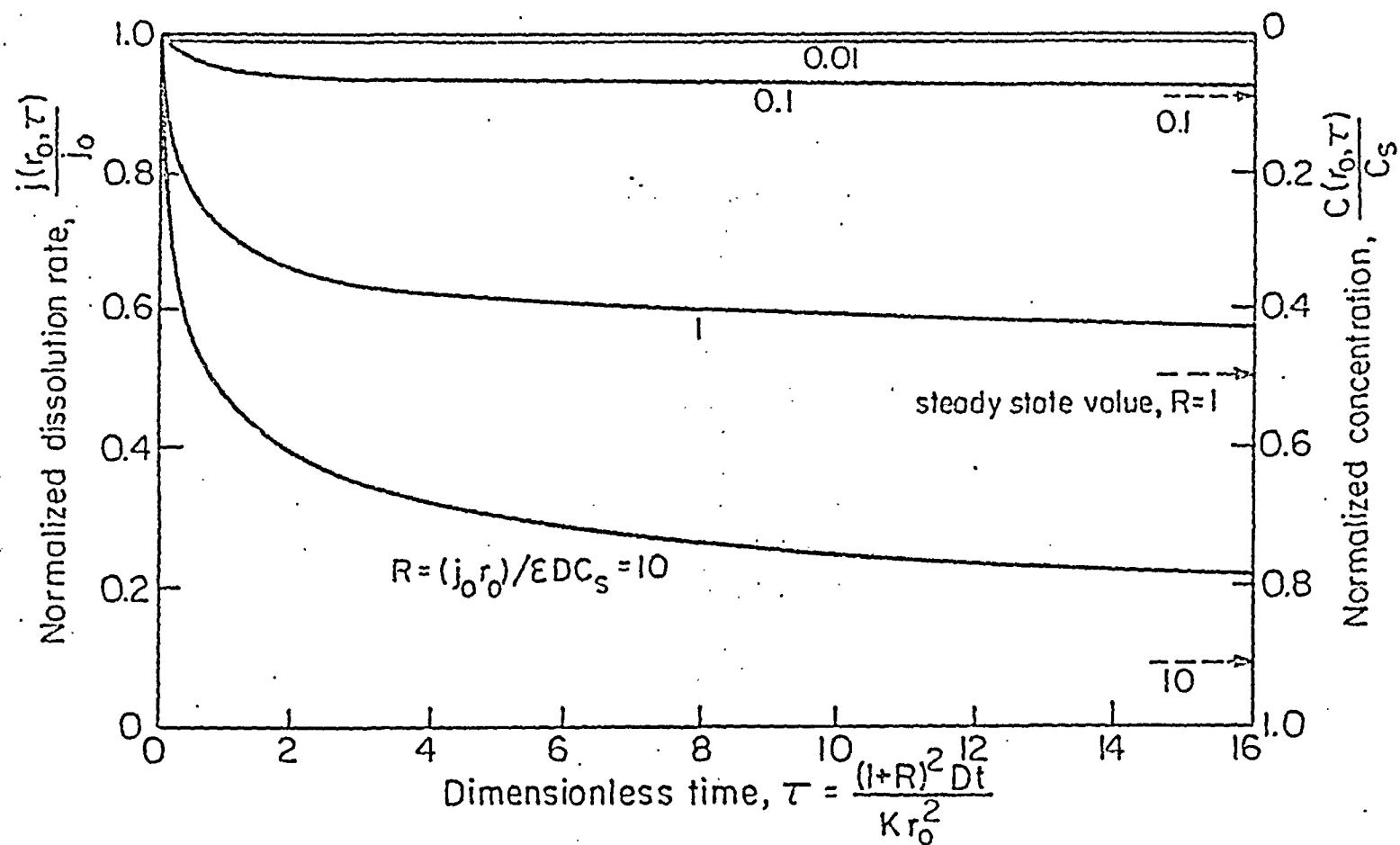


Fig. 2. Normalized dissolution rate and surface-liquid concentration as a function of dimensionless time for various values of the flux ratio R .

From Equation (12):

$$C(r_0, \infty) = 0.999 C_s, \text{ for SiO}_2$$

and the steady-state mass transfer is controlled by exterior-field diffusion.

Figure 3 shows the normalized surface mass flux of silica and the normalized surface concentration as a function of the dimensionless time and chronological time t . Also shown are the results predicted by assuming that saturation concentration always exists in the surface liquid, the assumption used in our previous conservative estimates of mass-transfer rate. For an assumed constant saturation the early mass-transfer rate is infinite, a result not physically reasonable. It is at these early times that the solid-liquid reaction rate is important for the dissolution of silica. The actual surface concentration is initially zero, and it grows with time at a finite rate determined by the solid-liquid reaction rate. Within about 7 minutes after a bare waste solid has been placed in contact with saturated rock ($\epsilon = 0.01$) the concentration of silica in the surface liquid will have reached 82 percent of saturation and the mass-transfer rate will be within 5 percent of that predicted for constant saturation at the surface, assuming no silica sorption in the rock. If silica sorbs with a retardation coefficient of 100, this time is increased to 700 minutes.

The time to reach near-saturation concentration will be longer if there is stagnant liquid between the waste surface and the exterior porous medium. This time delay is easily added to the theory because the intervening liquid is expected to be well mixed.

From the high-flow experiments of Strachan and Exahros, reported by Chick and Turcotte [14], we estimate the forward reaction rate of cesium dissolving from PNL 76-68 glass at 90°C:

$$j_0 = 2.0 \times 10^{-2} \text{ g/m}^2\text{day}$$

From [18]:

$$C_s = 5.97 \text{ g/m}^3$$

From [19], we adopt $K_{Cs} = 1000$

For a typical liquid diffusion coefficient $D = 1.2 \times 10^{-1} \text{ m}^2/\text{yr}$ at 90°C, which conservatively neglects the reduction in D because of tortuosity; the calculated concentration of cesium in the surface liquid as a function of time is shown in Figure 4. After 100 days the surface concentration is 90 percent of saturation. Steady-state mass transfer is achieved only after 2×10^5 years.

STEADY-STATE CONCENTRATIONS AND MASS-TRANSFER RATES FOR OTHER SPECIES

Table I presents values of the steady-state concentration ratio $C(r_0, \infty)/C_s$ and the flux ratio R for silica and several radioelements, calculated from Equations (12) and (13), together with the values of j_0 and C_s selected for these calculations. In all cases the steady-state surface concentration is within a fraction of a percent of saturation, the ratio R is large, and

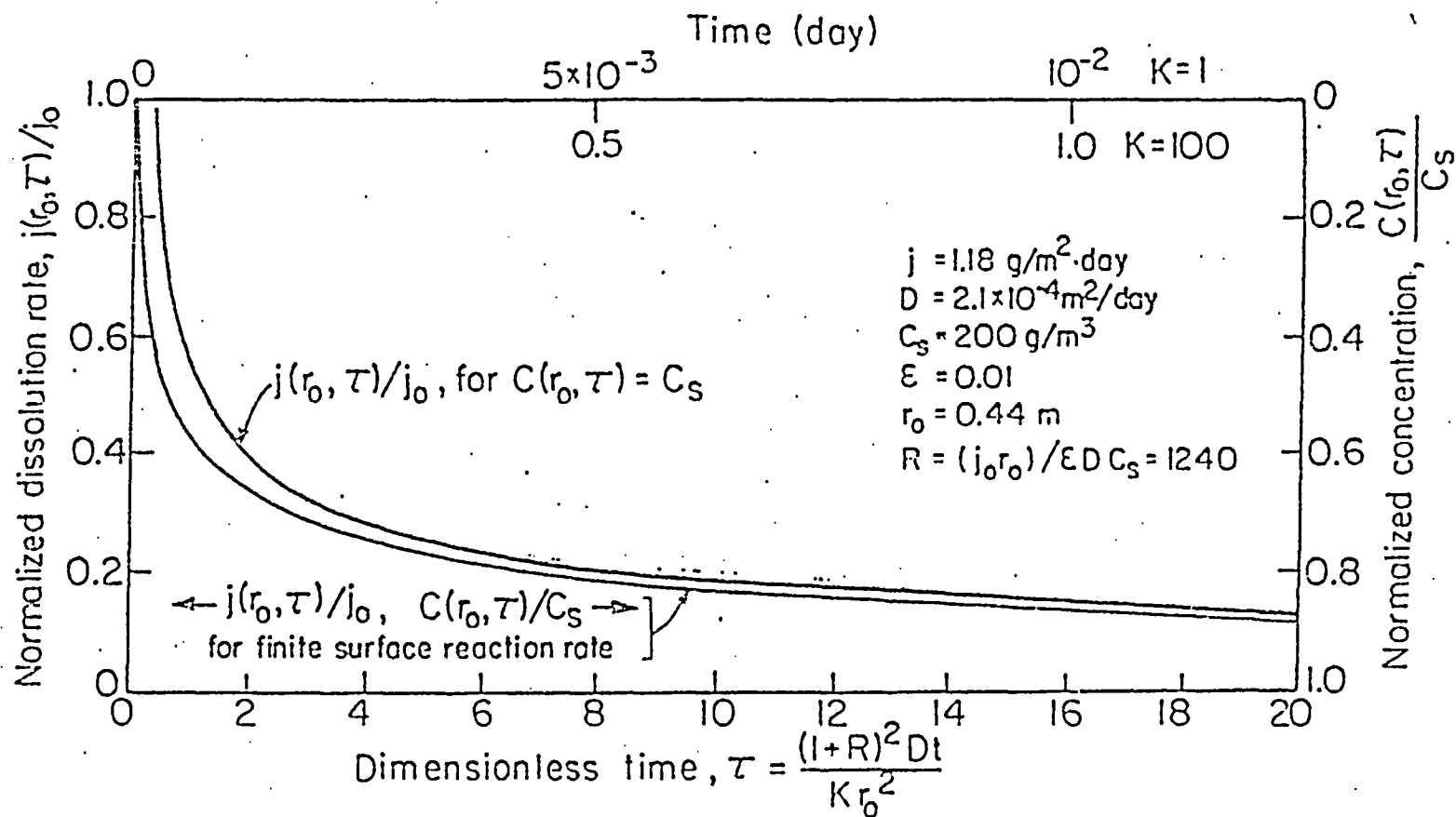


Fig. 3. Normalized dissolution rate and surface-liquid concentration of silica from borosilicate glass as a function of time, compared with dissolution rate for constant saturation concentration at the surface.

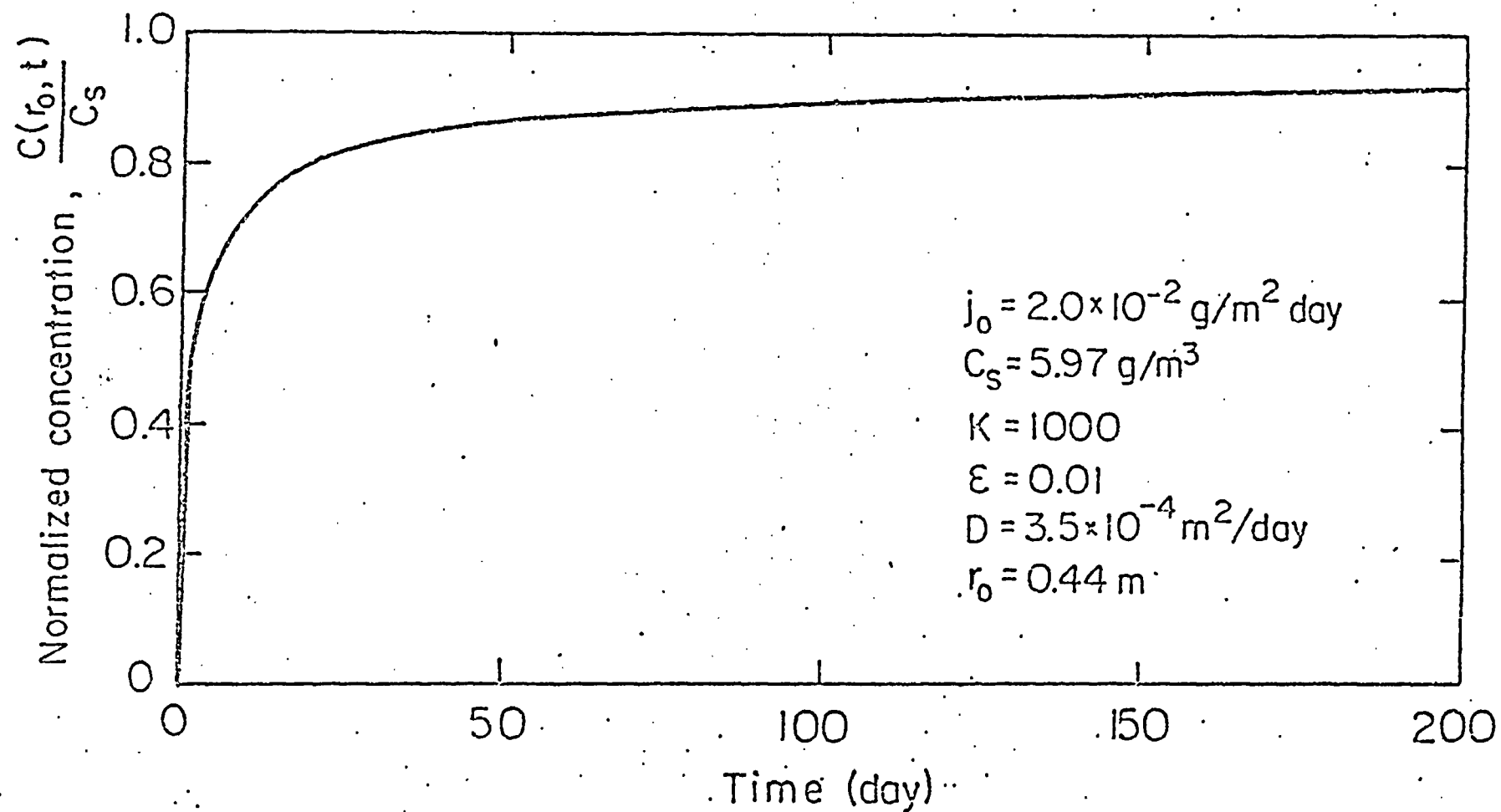


Fig. 4. Normalized surface-liquid concentration of cesium from borosilicate glass as a function of time.

Table I. Normalized surface concentration and flux ratio for borosilicate-glass constituents at steady state ($r_0 = 0.44$ m, $\epsilon = 0.01$)

Species	j_0 , g/m ² day	Solubility, g/m ³	Concentration ratio $d(r_0, \infty)/C_s$	Flux Ratio R
SiO ₂	8.0×10^{-1} <u>a/</u>	2.0×10^2 <u>c/</u>	0.999	5.1×10^2
U	9.84×10^{-5} <u>a/</u>	9.52×10^{-2} <u>c/</u>	0.998	5.3×10^2
Np	3.02×10^{-4} <u>a/</u>	2.37×10^{-2} <u>c/</u>	0.999	6.5×10^3
Pu	1.20×10^{-5} <u>a/</u>	1.43×10^{-5} <u>c/</u>	0.999	4.3×10^5
Am	6.29×10^{-5} <u>a/</u>	1.0×10^{-4} <u>d/</u>	0.999	3.2×10^5
Tc	2.02×10^{-1} <u>a/</u>	1.0×10^{-3} <u>d/</u>	0.999	1.0×10^8
Cm	2.57×10^{-7} <u>a/</u>	1.0×10^{-3} <u>d,e/</u>	0.992	1.3×10^2
Cs	2.0×10^{-2} <u>b/</u>	5.97 <u>f/</u>	0.998	4.3×10^2

a/ at 22°C, IAEA test, first-day leach data^[20].

b/ at 90°C, data of Strachan and Exarhos, as given by Chick and Turcotte^[14].

c/ at 25°C, concentration after 341 days in deaerated static leach experiment^[21].

d/ Estimates by Krauskopf^[17].

e/ Data subject to uncertainty.

f/ at 90°C, static leaching of borosilicate glass^[18], concentration in deionized water after 1 year, possible compound formation.

except for early times the dissolution rate is controlled by exterior-field mass transfer. The mass-transfer rates can be estimated accurately by assuming saturation concentration in the surface liquid, except during the first few days after the waste solid is first exposed to liquid.

COMPARISON WITH DATA ON NATURAL MINERALS

To illustrate the importance that solid-liquid reaction rate can have on dissolution rate, even for a dissolving solid surrounded by saturated rock, we consider the dissolution of natural quartz. The solubility of alpha silica is quoted to be 44 g/m^3 [22], over fourfold less than the apparent saturation concentration of silica in borosilicate glass. From the data of van Lier et al. [9], $j_0 = 3.1 \times 10^{-5} \text{ g/m}^2\text{day}$, over four orders of magnitude less than that deduced for silica in borosilicate glass. Thus, even for the same size of dissolving material the flux ratio R for quartz would be about four orders of magnitude less than that for silica in glass, and dissolution of quartz would be limited by solid-liquid reaction rates. Correcting further for the small size of quartz samples that have been studied as natural analogs, the flux ratio R is further decreased, and exterior-field diffusion is expected to be unimportant for quartz dissolution. This is consistent with the results of Wood and Walther [23].

Berner [24] has measured dissolution rates of various small grains of minerals in water-saturated rocks, solid, and sediments and observes that substances with solubilities greater than 10^{-4} moles/liter dissolve at a rate controlled by exterior-field diffusion, whereas substances with solubilities less than 10^{-5} moles/liter dissolve at rates controlled by surface reactions. Whether Berner's observations are consistent with the theory presented here is unclear. In the form of Equation (11), the flux ratio R is not dependent upon solubility, unless there is some connection between the reaction-rate constant k and C_s . However, Berner observes that the solid-liquid reaction is usually complex, and a higher-order reaction could result in the ranking observed by Berner. Without data on grain size r_0 , exterior-field porosity ϵ , and j_0 or k for Berner's experiments, we cannot test his observations with the theory presented herein.

Berner's observations with small grains cannot be generalized to the much larger sizes of solid wastes in a repository. The above theory shows that if the size of each separate dissolving grain is small enough (small r_0) a small flux ratio R can result and solid-liquid reaction rate can control dissolution, whereas for large r_0 exterior-field diffusion can control.

Although no generalization can be made on solubility alone, data in Table 1 show that for borosilicate glass waste the constituents with the lowest solubilities, plutonium and americium, have the greatest values of R , and their dissolution rate is controlled not by surface reaction rate but by exterior field mass transfer, assuming first-order solid-liquid reactions. This conclusion is consistent with that arrived at by comparing maximum exterior-field dissolution rates with those determined in laboratory leach experiments [19].

EFFECT OF INTERNAL CRACKS IN THE WASTE SOLID

The data in Table 1 predict that low solubilities will cause dissolution rate to be controlled by exterior-field diffusion, for the species considered in that table and for waste-solid sizes typical of those to be emplaced in a geologic repository. As has been discussed elsewhere [19], the waste solid is likely to be cracked and fragmented. When exterior-field mass transfer is controlling the dissolution rate and the flow of groundwater through the waste solid is negligible, the exterior surface area of the waste solid is the proper surface area on which to base the calculation of release rate, as has been done in the above analysis. If solid-liquid reaction rate is controlling, the availability of internal surface in cracks and fissures can increase the effective surface area available for solid-liquid reaction, and a highly fragmented solid will increase the effective value of j_0 used in the above analysis. The values of j_0 used in the numerical illustrations herein are based on solid samples with a reported effective surface area for solid-liquid reaction about fivefold greater than the outer geometric surface area of the sample. If the equivalent waste solid in the repository were to have no equivalent internal surface area for solid-liquid reaction, the effective j_0 would decrease by about fivefold, and R could decrease by this amount. If the other data are correct, the R values would still be large and our conclusions would remain the same.

If the waste solid were more highly fragmented than that implied in the laboratory experiments, the effective j_0 would increase and the time to reach near saturation in the surface liquid would become even shorter than estimated herein. Because this time is already short for the parameters selected for illustration, the conclusions will remain the same.

EFFECT OF TORTUOSITY

All of the above numerical illustrations conservatively assume a diffusion coefficient appropriate for a liquid continuum. Measured coefficients for molecular diffusion in granite [25,26] are 100- to 1000-fold less than the values used herein, whereas values tenfold lower are typical for other rocks. The lower diffusion coefficients increase the value of R , they further reduce the importance of solid-liquid reaction rates, and they increase the time to steady-state.

VERIFICATION OF FIRST-ORDER SOLID-LIQUID REACTION

The first-order rate of solid-liquid reaction implied in Equation (3) can be inferred from laboratory leach experiments in which a sample of surface area S is exposed for time t to a leachant of volume V . Because of the time scale of the experiments a spatially uniform concentration $C(t)$ of the dissolved species can be expected. For a first-order reaction and a leachant initially containing no solute (at $t = 0$), we expect:

$$C(t) = C_s \left[1 - \exp \left(- \frac{S j_0 t}{V C_s} \right) \right] \quad (15)$$

as suggested by the experiments on silica by O'Conner and Greenberg [8]. Equation (15) correlates the data on quartz and vitreous silica obtained by van Lier et al. [9] and Wirth et al. [10].

The values of j_0 and C_s determined from the data of Pederson et al. [12] on the 28-day leaching of silica from borosilicate glass at 90°C, as quoted earlier, are used in Equation (15) to predict the curve of $C(t)$ plotted against St/V in Figure 5 and the curve of cumulative release in Figure 6. The agreement with the 28-day experimental data in Figure 5 is excellent, and there is reasonable agreement with the longer-term data of Strachan [13] in Figure 6. The same values of j_0 and C_s are used in an expression similar to Equation (15) to predict $C(t)$ when leachant is withdrawn from the experiment at a volumetric rate Q and is replaced with fresh leachant, with the results shown in Figure 7. The theory agrees well with the flow leaching data reported by Chick and Turcotte [14] for the dissolution of silica from borosilicate glass at 90°C.

Based on these comparisons, the first-order reaction-rate boundary condition Equation (3) is reasonable when the resulting boundary concentration of silica reaches a nearly constant value within a year or less, as is predicted for the repository conditions considered herein.

CONCLUSIONS

The theory for predicting waste-solid dissolution rate on the basis of mass-transfer rate in surrounding groundwater has been extended to include solid-liquid chemical reaction rate as a boundary condition. A quantitative criterion is developed to predict under what conditions either of the rate processes controls dissolution rate.

From data deduced from laboratory experiments on PNL 76-68 borosilicate glass, and using Krauskopf's [19] solubility data for some species, the dissolution rates of silica, uranium, neptunium, plutonium, americium, technetium, curium and cesium are predicted to be controlled by exterior-field mass transfer, except at early times of the order of weeks or months after the beginning of exposure to groundwater.

Based on the data adopted for this calculation, the dissolution rates predicted for these species can be adequately estimated by assuming saturation concentration in the liquid at the waste surface. This justifies the use of the analyses developed earlier by Chambre' [1-4] that include the effects of backfill and radioactive decay and assume constant saturation concentration at the waste surface.

This theory is a mechanistic representation of processes expected to control or affect the rate of dissolution of waste solids in a saturated-rock repository. It does not require ad hoc assumptions of effective ratios of waste surface area to repository water volume and effective water-solid contact time that have been postulated by others [6,7] to predict waste dissolution in a repository.

The theory identifies the experimental data important for predicting waste performance in a repository: saturation

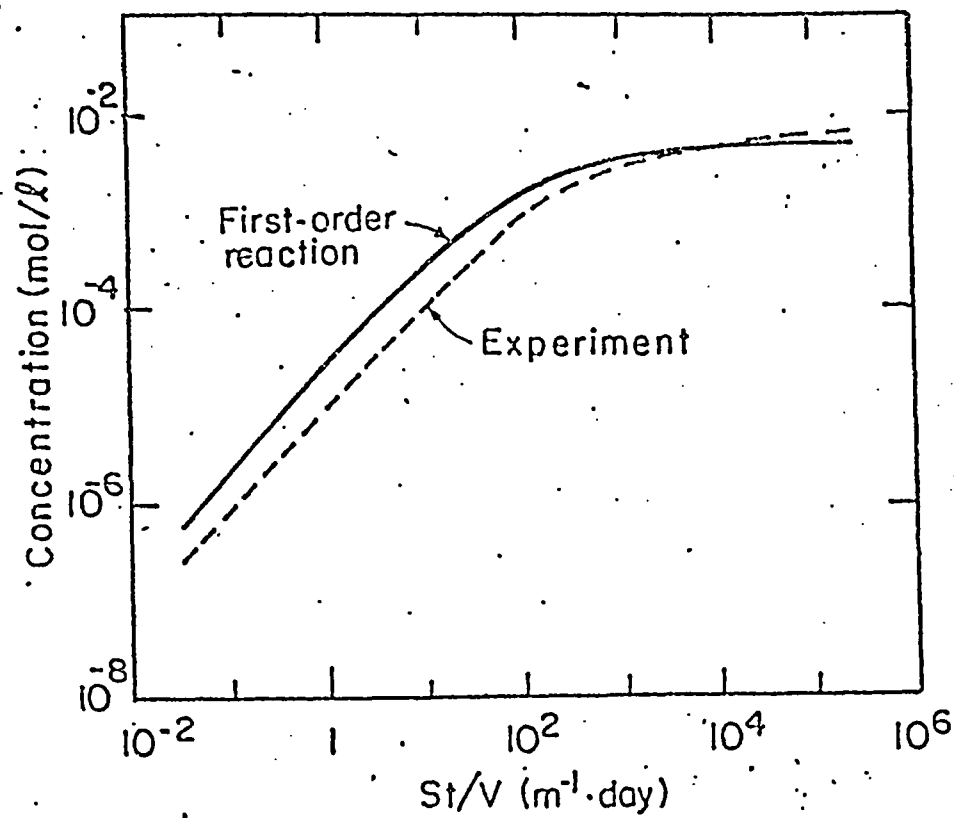


Fig. 5. Comparison of first-order solid-liquid reaction-rate equation with experimental data [12] for silica from borosilicate glass.

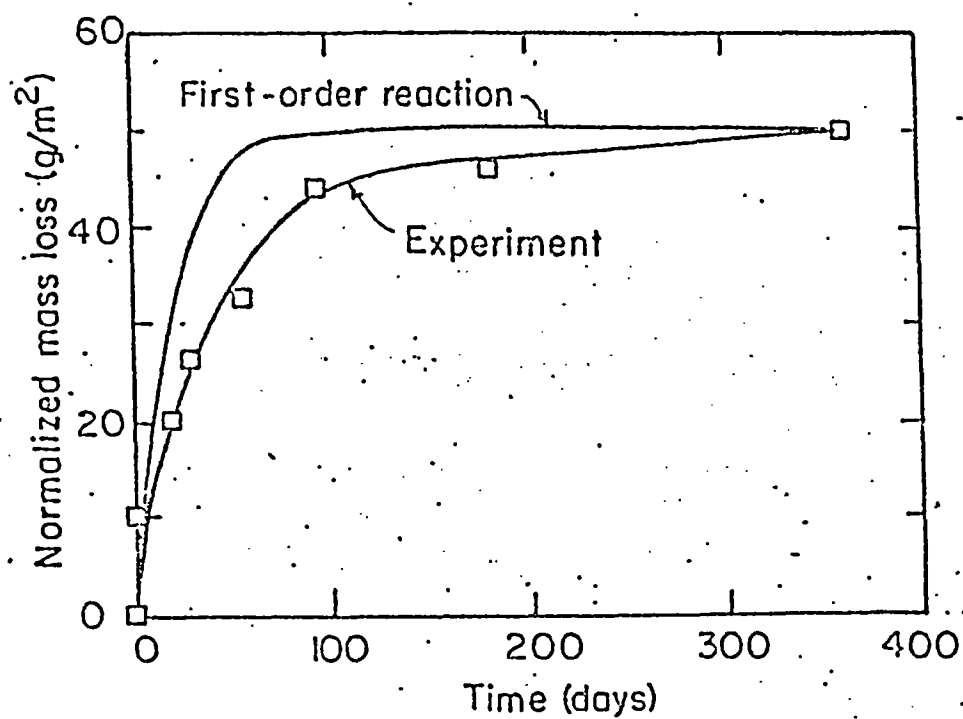


Fig. 6. Comparison of cumulative release from first-rate equation with one-year experimental data [13] for silica from borosilicate glass.

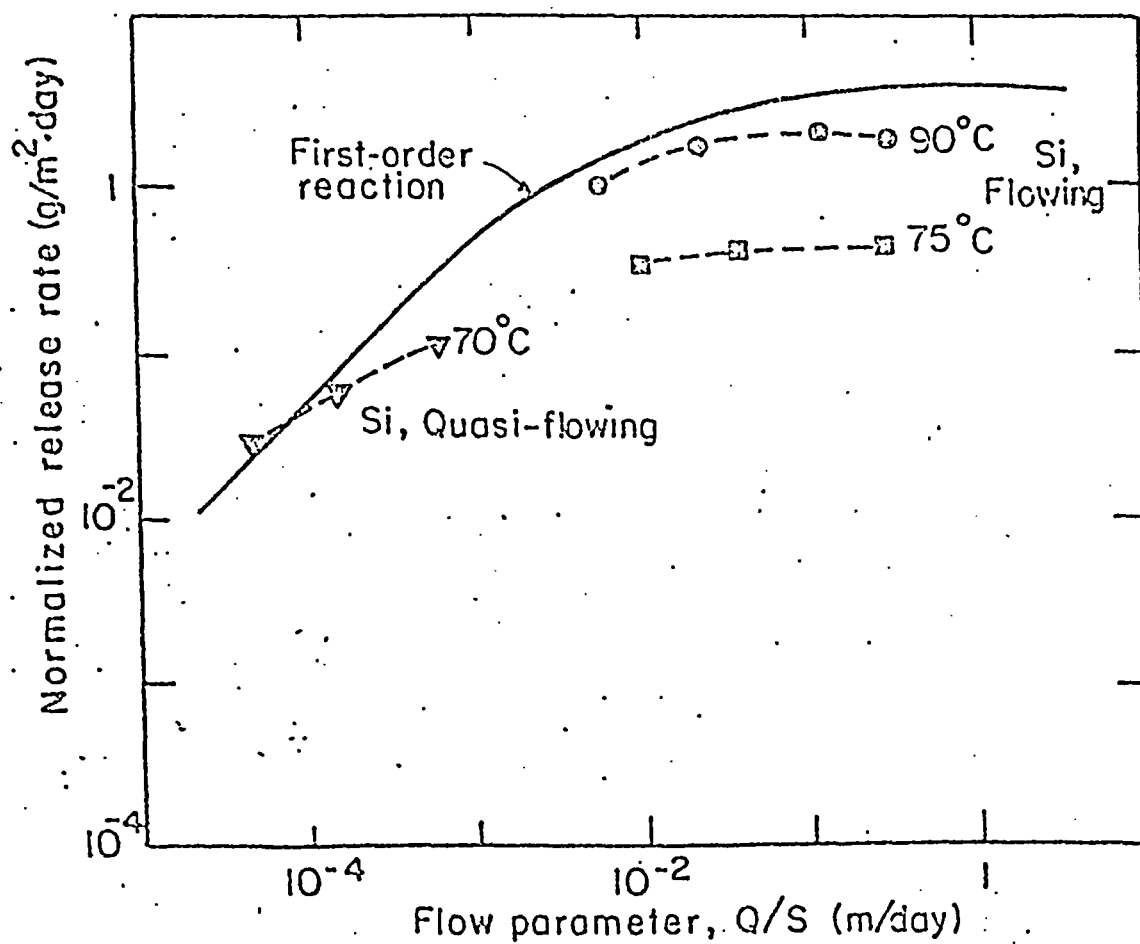


Fig. 7. Comparison of first-order solid-liquid reaction-rate equation with flow leaching data [14] for silica from borosilicate glass.

concentrations, diffusion coefficients, porosity, and retardation coefficients. Forward-reaction rate data are useful to confirm the importance of exterior-field mass transfer or to predict dissolution rates if solid-liquid reaction rate is important. Such data are needed for spent-fuel constituents such as carbon, iodine, cesium, technetium.

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Vol. 1, 2 MENTAL IMPACT STATEMENT: DISPOSAL OF HANFORD DEFENSE HIGH-LEVEL, TRANS-
& 3 URANIC AND TANK WASTES. March 1986. 3 vols. (A few extra copies available)
Available from Mr. Steve Leroy, U.S. Dept. of Energy, P.O. Box 550,
Richland, WA 99352 ph. (509) 376-7378.

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R Ba 1 "How can we tell if a Hanford Repository will be safe?". Olympia,
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RECENT REPORTS FROM U.S. DEPT. OF ENERGY, OFFICE OF CIVILIAN RADIOACTIVE WASTE MGMT.

DOE/RW-0004/2 OFFICE OF CIVILIAN RADIOACTIVE WASTE MANAGEMENT ANNUAL REPORT TO
CONGRESS. March 1986. 51 p.

"This is the third Annual Report on the activities and expenditures of the Office of Civilian Radioactive Waste Management (OCRWM)....for fiscal year 1985...from October 1, 1984 through September 30, 1985." (from Foreword)

DOE/RW-0041 OCRWM INFOLINK: AN ELECTRONIC BULLETIN BOARD. December 1985. 33 p.

"This Users Manual provides instructions for access to OCRWM INFOLINK. The Elec-Bulletin Board will provide ready access to texts of Press releases, Fact Sheets/Backgrounders, Congressional questions & answers, Congressional testimony, Speeches, Schedules, Miscellaneous announcements."

DOE/RW-0043 PROGRAM MANAGEMENT SYSTEM MANUAL. January 1986. 61 p.

"This PMS Manual describes the hierarchy of plans required to develop and maintain the cost, schedule, and technical baselines at the various organizational levels of the (CRWM) Program. It also establishes the management policies and procedures used in the implementation of the Program." (From Memorandum by Ben C. Rusche, Director, OCRWM)

DOE/RW-0044 REPORT OF THE TASK FORCE ON THE MRS/REPOSITORY INTERFACE. Feb. 1986.

"To analyze the design, licensing, cost, and schedule impacts of implementing an integrated waste-management system, the DOE established in April 1985 an MRS/repository interface task force and asked it to address...questions [on appearance of surface facilities, cost, licensing, schedule, and system functions of the integrated MRS/repository facility]" (From Exec. Summary)

DOE/RW-0052 NEAR-TERM PUBLIC INFORMATION PRODUCTS PROGRAM FY 86-87. Feb. 1986.
Rep. 3/86 52 p.

"This document contains a comprehensive listing of 65 OCRWM public information printed products and instructions for implementing a near-term, 18-month program... includes ... product requirements, methodology for timely product development and distribution, and a means to measure progress." (From memo from Ben C. Rusche)

DOE/RW-0063 WASTE MANAGEMENT SYSTEMS REQUIREMENTS AND DESCRIPTIONS (SRD).

"This document covers only the functional requirements of the system...[not] programmatic or procedural requirements pertaining to processes of designing, siting and licensing....a system description is provided for an 'improved-performance system' which would include a monitored retrievable storage (MRS) facility." (From Intro.)

DOE/RW-0065 TRANSPORTING SPENT NUCLEAR FUEL: AN OVERVIEW. March 1986. 23 p.

"This booklet focuses on various aspects of transporting commercial spent fuel, which accounts for the majority of the material to be shipped. [It] is intended to give reader a basic understanding of"...reasons for transportation of spent fuel, methods of shipping, safety and security precautions, emergency response procedures, the DOE program/uniquely appropriate to NWA transportation requirements. (From Intro.)

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PARISON STUDY (INTRACON) WORKSHOP AND COORDINATING COMMITTEE MEETING,
INTERLAKE, SWITZERLAND, FEBRUARY 15-20, 1982. January 1986.
121 p. + Appendices on microfiche.

BMI/ONWI-592 ERG REVIEW OF SALT CONSTITUTIVE LAW, SALT STRESS DETERMINATIONS, AND
SALT CORROSION AND MODELING STUDIES. Jo Ellen Balon. For ONWI.
March 1986. 29 p.

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DOE/EIS-0109F FINAL ENVIRONMENTAL IMPACT STATEMENT
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AT THE NIAGARA FALLS STORAGE SITE. Washington, D.C., U.S. Dept. of
Energy, April 1986. variable paging.

DOE/NE/44139-T2 ANNUAL REPORT, WEST VALLEY DEMONSTRATION PROJECT, WEST VALLEY,
NEW YORK. January 1986. Washington, D.C., USDOE Ass't. Secretary
for Nuclear Energy, January 1986. 13 p.

The Nuclear Waste Policy Act states in Section 2(a) "The Secretary [of DOE] shall solidify, in a form suitable for transportation and disposal, the high-level radioactive waste at the [Western N.Y. Nuclear Services Center, West Valley, N.Y.] Center by vitrification or by such other technology with the Secretary determines to be the most effective..." This project is being conducted at a former spent nuclear fuel reprocessing facility. Liquid waste is converted into a durable borosilicate glass and poured into stainless steel canisters which will be stored until placed deep in the earth in a federal repository.

DOE/NE/44139-T3 ANALYSIS OF THE TERMINAL WASTE FORM SELECTION FOR THE WEST VALLEY
DEMONSTRATION PROJECT. W. H. Hannum. By West Valley Nuclear Services
Company, Inc. for U.S. DOE. Jan. 1983. 23 p.

"This paper summarizes the environmental considerations associated with the selection of borosilicate glass as the waste form." (From Intro.)

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& K. Pruess, Lawrence Berkeley Laboratory. Prepared under auspices
of U.S. Nuclear Regulatory Comm. for U.S. Dept. of Energy. Dec. 1985.
19 p.

"In the context of nuclear waste isolation, heat pipe systems may evolve in the thermal regime near the waste packages. They may have very substantial effects on important aspects of repository performance, including peak temperatures, salinity, and gas content of the fluids near waste packages, and migration of chemical species." (From Intro.)

NUREG GENERIC TECHNICAL POSITION ON BOREHOLE AND SHAFT SEALING OF HIGH-LEVEL
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