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Organic Complexant- Enhanced Mobility of Toxic Elements in Low-Level Wastes

**Quarterly Progress Report
January-March 1984**

J. L. Swanson

June 1984

**Prepared for
the U.S. Nuclear Regulatory Commission
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PACIFIC NORTHWEST LABORATORY
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ABSTRACT

This report describes the results obtained during the third quarter's activities of a project whose objective is to determine how and to what extent organic complexants affect the mobility of toxic elements in subsurface groundwaters at commercial low-level waste disposal sites. This project will study nonradioactive toxic elements as well as elements having radioactive isotopes of importance (e.g. ^{63}Ni , ^{239}Pu , ^{241}Am). Organic complexants used in the nuclear industry are being emphasized, but we also examine others. We are employing generic soil components (e.g. hydrous oxides, silica, clays) so that the results will be broadly applicable.

Substantiation of the previously-indicated sorption of a Pu(IV)-EDTA complex by hydrous ferric oxide ($\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$) was obtained by comparing the sorption of EDTA in the presence and absence of Pu. Additional data on the sorption of a Ni-EDTA complex by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ were also obtained.

Preliminary Ni sorption data were obtained with other complexants (picolinic acid and citric acid), and another generic soil component (TiO_2).

Sorption of a Ni-EDTA complex by an anion exchange resin was observed. Complexed species are thus likely to be present in the resin wastes from certain reactor decontamination solution clean-up operations.

An experimental problem that caused some erroneous results for uncomplexed Ni was discovered and corrected. The filters being used to assure good separation of solid and liquid phases were removing Ni from solution, which skewed some earlier results.

INTRODUCTION

The primary mechanism of escape of toxic elements from commercial low-level waste disposal sites to the accessible environment, where they may endanger public health and safety, is migration with subsurface groundwaters. As the water passes through the soil, many of the toxic elements may be removed from solution by sorption or precipitation reactions, with the result that their migration rates (mobilities) are markedly retarded. The degree of retardation depends on the nature of the soil and on

the nature of the toxic element species present in the solution, and may be adversely affected by the presence of organic complexing agents.

Issues Addressed by This Research

This research addresses two key issues for licensing of low-level waste disposal sites: 1) What restrictions should be placed on the disposal of wastes containing different organic complexants? 2) How can the effects of organic complexants be included properly in predicting the release of toxic elements?

To properly address these key issues, the technical questions below must be examined.

1. What are the important gross mechanisms affecting the retardation of toxic element mobility in soil/groundwater systems? The relative importance of retardation phenomena such as precipitation, coprecipitation, surface adsorption, and diffusion of toxic elements into the bulk sorbent must be assessed.
2. What are the quantitative effects of specific organic complexants with specific soil components on these retardation mechanisms? Among such effects are extent of complex formation (with elements dissolved from the soil components as well as with toxic elements), rates of complex formation and dissociation, and sorption of the complexes themselves.
3. What data need to be obtained to provide the proper input to predictive models so that the toxic element migration rates obtained from such models will be meaningful? For this purpose it is of paramount importance to know whether the data represent a true equilibrium condition or are merely dictated by the kinetics of the system. Further, if changes with time are observed, it is necessary to know if these changes are caused by changes in the soil component properties (aging) or by changes in the extent of element complexation. It is also necessary to know the effects of element and complexant concentrations and the soil-to-solution ratio.
4. What are the basic mechanisms by which the important gross mechanisms operate? For example, if a complexed toxic element species appears to

be sorbed, how does this occur? Is the mechanism really sorption of a preformed complex, or does the apparent sorption involve first sorption of the complexant and then complexing of the toxic element by the sorbed complexant? Also, if a toxic element appears to be sorbed by a hydrous oxide or hydroxide, is sorption truly the mechanism or could the mechanism be precipitation caused by a localized high hydroxide ion concentration near the surface? While knowledge of such basic mechanisms may not be necessary for the development of adequate predictive capability, it would certainly lend confidence in the accuracy of such predictions.

Background and Objectives

Organic complexing agents (e.g. EDTA, NTA, oxalate, citrate) are often used in the nuclear industry to remove radioactive materials from contaminated equipment and facilities. The wastes resulting from such activities therefore contain the organic complexants, unless they are destroyed before disposal of the wastes. These complexants affect the mobility of toxic elements. Toxic element mobilities can also be affected by other manufactured organic complexants that may enter the low-level waste disposal site by other means and by naturally occurring organic complexants. Determination of the potential importance of such complexants to the mobilities of toxic elements present in low-level waste is the objective of this research project.

There have been several reports of enhanced radionuclide mobilities resulting from the presence of organic complexants. The migration of a Co-EDTA complex from a low-level waste burial site was first reported by Means, Crerar, and Duguid (1978). Similar results have also been obtained at a different low-level waste disposal site (Toste, Kirby, and Pahl 1983) for both Co and alpha-emitting radionuclides. A Pu-EDTA complex was also found in leachates from the latter site (Cleveland and Rees 1981). These observations illustrate the potential importance of organic complexants to radionuclide mobilities and also provide a basis of the need for a broader project such as that planned here.

This project will build on our previous work in this area (Swanson 1981, 1982, 1983a). Three different soils were studied in that work. The elements

Co and Ni, which have important isotopes present in low-level wastes, were studied in the most detail. An important activity of this work was the development of procedures to determine whether a radionuclide sorption coefficient that appears to be stable with time is truly an equilibrium value or appears stable only because the rate of change is very slow. An understanding of this is necessary to accurately predict the rate of radionuclide migration.

The effect of organic complexants on the sorption of radionuclides was found in our previous work to vary widely among different complexants. This variation involves not only the concentration of complexant required to have an appreciable effect on the equilibrium sorption coefficient, but also involves how rapidly the equilibrium is attained. The latter factor is more important since, in the extreme case, kinetically inert complexes could migrate at the same speed as groundwater. Rapid dissociation of the complexes is desirable so that the uncomplexed radionuclide can be sorbed by the soil.

The comparison of complexing agents included study of the fate of some of the complexants themselves as well as study of their effects on radionuclide distribution. The results of these tests indicated that all organic complexing agents should not be considered to be equally hazardous from a waste disposal standpoint.

The earlier work also showed that sorption of some kinetically inert radionuclide complexes, as well as sorption of uncomplexed radionuclides, is important with some soils. In these cases, the radionuclide migration rates were lower than they would have been if the complexes had not been sorbed, but were higher than they would have been if the complexes were not kinetically inert, or if complexants were absent.

The three different soils studied affected organic complexant-enhanced radionuclide mobility differently. Two of them sorbed kinetically inert complexes and one did not. Radionuclide complexes dissociated more rapidly in some soil/groundwater systems than they did in others. The soils also differed in their leaching by complexant solutions; different elements were predominant in leachates of different soils, and rates of leaching were different. These differences also affected radionuclide behavior.

Scope of Research

The current project is continuing and expanding the investigations summarized above. It is employing generic soil constituents (e.g. hydrous oxides, clays, and silica) so that the results will be more broadly applicable. The number of toxic elements studied will be expanded by including transuranic (TRU) and nonradioactive toxic elements that may be present in low-level waste. We are continuing to emphasize study of the organic complexing agents used in the nuclear industry but will also examine the effects of some other manufactured complexants and some naturally occurring ones.

EXPERIMENTAL METHODS

The primary experimental method continues to be K&E experiments, in which both kinetic and equilibrium data are obtained by measuring the radionuclide distribution at various time intervals as the solution and soil component are shaken together. These experiments are being performed at ~22°C, generally in polypropylene tubes.

Solution Clarification

A problem that negated some of the earlier results became apparent during the course of recent experiments. The 2- μ m filters that were being used to assure that no finely divided solids were present in the analytical samples were removing uncomplexed Ni from solution, presumably by sorption on the filter medium. The earlier sorption data for uncomplexed Ni thus included sorption of both the filter medium and the added soil constituent. At low pH values the sorption by the filter medium was much greater than that by the soil constituent.

A similar problem with these filters had been observed earlier in similar systems (Swanson 1982) but was not apparent when the first tests were done in our current work. This was because the effect is much less pronounced in 0.1 M NaNO_3 than in 0.01 M NaNO_3 , and the initial checking was done in a 0.1 M NaNO_3 experiment.

Data illustrating the filter sorption problem are contained in Table 1, where results obtained after experimental suspensions were clarified by centrifugation or by filtration are presented. Filters having three different pore sizes were used; they were from three different sources and presumably had three different filter media. Comparable results were obtained with centrifugation or with filtration through the 0.1- or 0.2- μ filters. Only with the 2-nm filter, which is the one we had been using routinely, was the result different; in these cases the apparent Ni sorption was much higher. Based on the single-portion filtration data, it could be argued that the difference was due to the more efficient removal of finely-divided particulate species by the filter having the smallest pore size. However, data obtained when multiple portions of the suspension were clarified by the same filter show that this was not the case. As multiple portions were clarified by the 2-nm filter, the apparent sorption indicated by the Ni content of each filtrate portion decreased. This is not what would be expected if the 2-nm filter had removed finely-divided particulates more efficiently; if that had been the case, there should have been either no change in the Ni content of consecutive portions, or the Ni content would have decreased (giving an increasing apparent sorption) as the removed particulates decreased the

TABLE 1. Comparison of Apparent Uncomplexed Ni Sorption with Different Solution Clarification Techniques

Clarification Technique	% of Ni Sorbed(a)	
	Expt. 104(b)	Expt. 105(c)
Centrifugation	0 to 6	-1 to 1
Filtration through 0.2 μ	5	1
-second portion	6	2
Filtration through 0.1 μ	8	3
-second portion	11	1
Filtration through 0.002 μ (2 nm)	48	63
-second portion	31	26
-third portion	22	19

(a) Calculated from analysis of the initial solution and the solutions after clarification.

(b) pH = 5.9; 5×10^{-4} M $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ in 0.01 M NaNO_3 .

(c) pH = 4.2; 30 g/L SiO_2 in 0.01 M NaNO_3 .

effective pore size. Thus it appears that uncomplexed Ni was sorbed by the medium of the 2-nm filter, and that the sorption sites tended to become loaded as consecutive portions were passed through the same filter.

The different clarification techniques were also compared under a wide variety of other conditions. Close agreement persisted between uncomplexed Ni results obtained by centrifugation and by 0.2- μ filtration at all pH values examined. All four techniques gave comparable results in the presence of complexants, indicating that complexed Ni was not affected by any of the filter media.

Counting Procedures

Some experiments this quarter involved both ^{239}Pu and ^{14}C (as labeled EDTA). It was possible to measure both of these radioisotopes by liquid scintillation counting by careful selection of the detection ranges. There was no contribution of ^{14}C in the predominant ^{239}Pu range, but there was some ^{239}Pu contribution in the ^{14}C range that was easily corrected. The ^{239}Pu count in the ^{14}C range amounted to about 9% of the Pu count in its predominant range.

EXPERIMENTAL RESULTS

Experimental effort this quarter again centered on ^{63}Ni and ^{239}Pu radionuclides, and hydrous ferric oxide and silica soil components. More definitive evidence was obtained for the sorption of a Pu(IV)-EDTA complex by hydrous ferric oxide, corrected data were obtained for the sorption of uncomplexed Ni, preliminary data were obtained for Ni with other complexants and soil components, and the sorption of Ni complexes by ion exchange resins was examined.

Pu(IV)/EDTA $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ System

In the initial report of this series (Swanson 1983b) evidence was presented to indicate that a Pu-EDTA complex was sorbed by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$, which is not in agreement with the common hypothesis that Fe displaces Pu from the

complex, leaving uncomplexed Pu to be sorbed. This evidence was that higher Pu sorption was observed in the presence of EDTA than in its absence. Subsequent investigation (Swanson 1984) failed to provide additional evidence. Those previous investigations were hampered by the fact that Pu sorption was very high in all cases, so that changes caused by EDTA were hard to quantify. We have now obtained corroborative evidence by measuring the effect of Pu on the sorption of EDTA (^{14}C labeled) by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$.

Table 2 presents the results of the experiments in which EDTA sorption was measured in the presence and absence of Pu(IV). In three experiments in the absence of Pu, ~17% of the EDTA was sorbed by $1.5 \times 10^{-4} \text{ M}$ $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ at pH ~7.4. When part of the EDTA was complex by Pu(IV) before exposure to $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$, a much larger portion of the EDTA was sorbed. This indicates that a Pu(IV)-EDTA complex is indeed sorbed by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$.

Calculation of the EDTA:Pu ratio in the sorbed complex gave values ranging from 0.25 to 0.5 (Table 2), assuming that all of the sorbed Pu was present as a complex. Because we don't know whether this assumption is correct, we apply no significance to these ratios. However, we do feel that the data clearly show that sorption of a Pu(IV)-EDTA complex by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ does indeed occur.

TABLE 2. Evidence for Sorption of Pu(IV)-EDTA Complex by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ ^(a)

Tube No.	Initial Composition, M		Initial EDTA/Pu	Time Days	pH	% of Total Sorbed		EDTA Sorbed as ^(b) Pu Complex, M	EDTA/Pu in ^(c) Sorbed Complex
	EDTA	Pu				EDTA	Pu		
130	3.8×10^{-6}	--	∞	1	7.4	19	--	--	--
				2	7.3	19	--	--	--
118	5.8×10^{-6}	--	∞	1	7.4	16	--	--	--
				2	7.5	13	--	--	--
121	5.8×10^{-6}	--	∞	1	7.4	17	--	--	--
129	3.5×10^{-6}	4.5×10^{-6}	0.8	1	7.3	49	100	1.1×10^{-6}	0.25
				2	7.2	46	100	1.0×10^{-6}	0.23
117	5.4×10^{-6}	4.5×10^{-6}	1.2	1	7.2	61	100	2.4×10^{-6}	0.53
				2	7.2	58	100	2.2×10^{-6}	0.49
120	5.4×10^{-6}	2.3×10^{-6}	2.4	1	7.2	38	100	1.1×10^{-6}	0.49

(a) $1.5 \times 10^{-4} \text{ M}$ $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$.

(b) [(Fraction of EDTA sorbed in presence of Pu)-0.17] (initial EDTA concentration).

(c) (Molarity of EDTA sorbed as Pu complex)/(molarity of Pu sorbed).

Sorption of Uncomplexed Ni by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ and by SiO_2

In the previous report (Swanson 1984) we presented several figures comparing Ni sorption by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ and SiO_2 in the presence and absence of complexants. We have since determined that most of the data obtained in the absence of complexants were incorrect because of sorption of uncomplexed Ni by the filter medium in the 2-nm filters that we had been using to assure the absence of particulate material in the samples that were analyzed.

Figure 1 shows corrected data on how the sorption of uncomplexed Ni by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ and by SiO_2 varies with pH. Centrifugation was used in these experiments to separate the solid and liquid phases, thus avoiding sorption by filter medium. The variation of Ni sorption with pH is much more abrupt with $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ than with SiO_2 , and Ni sorption by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ is stronger than by SiO_2 having approximately the same total surface area. The total surface areas were calculated from the specific surface area values of $200 \text{ m}^2/\text{g}$ for Fe_2O_3 and $3.3 \text{ m}^2/\text{g}$ for SiO_2 , as reported by Leckie et al. (1980) for similarly prepared materials.

Sorption of uncomplexed Ni by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ was found to be the same in 0.01 M NaNO_3 as in 0.1 M NaNO_3 . The apparent difference reported in these conditions (Swanson 1984) was caused by much greater sorption of Ni by the filter medium in 0.01 M NaNO_3 than in 0.1 M NaNO_3 .

Ni/EDTA/ $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ System

Very little, if any, sorption of complexed Ni by the filter medium used in the earlier tests (Swanson 1984) has been observed in the recent examinations of filter problems. Therefore, the earlier results obtained in the presence of EDTA are valid. However, additional data obtained recently in $1 \times 10^{-4} \text{ M EDTA}$ give a somewhat different picture of the abruptness of the pH dependence than that presented earlier. These data are compared in Figure 2.

The distribution coefficients (K_d 's) of complexed and uncomplexed Ni by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ calculated from the data of Figures 1 and 2 are compared in Figure 3. The slopes of both of these lines are greater than those presented earlier (Swanson 1984) because of the additional and more correct data recently obtained.

FIGURE 1. Uncomplexed Ni Sorption by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ and by SiO_2

FIGURE 2. Sorption of Ni-EDTA by $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$

FIGURE 3. Nickel Distribution in Presence and Absence of EDTA

The Ni sorption data shown in Figures 1, 2, and 3 were obtained in experiments lasting only a few days. Data were previously reported (Swanson 1984) to show that marked changes occur over longer time periods. For example, the sorption of Ni in the presence of 1×10^{-4} M EDTA at pH=5 decreased from ~50% after a few days to ~5% after ~80 days. One phenomenon that could cause such a change is the conversion of EDTA. However, this phenomenon does not appear to have been responsible for the observed change; determination (by A. P. Toste of PNL) of the EDTA concentration of that solution after 85 days showed ~55% of the amount initially added. This change in EDTA concentration does not account for the observed change in Ni distribution over the long time period.

The marked change of Ni behavior with time in the presence of EDTA thus remains unexplained. Among the possible explanations are: a) changes in the sorptive properties of $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ with time, b) changes in the nature of the complex, and c) slow kinetics of dissociation of the sorbed complex. The earlier discussion (Swanson 1984) ruled out changes in sorptive properties, based on some results obtained after eight days; a reexamination of experiments with longer-aged sorbent may show different results. The slow kinetics dismissed earlier should also be reexamined.

Other Ni/Complexant/Soil Component Systems

The study was expanded this quarter to include several other complexants (picolinic acid and citric acid) and another generic soil component (TiO_2). These complexants are among those commonly used for decontamination of reactors and other nuclear facilities and equipment. Titanium dioxide is reported to be a strong sorbent; since it is present as an impurity in the clays that we will be examining, its effect is being assessed in systems of interest here.

Only preliminary results are available in these systems. They show that:

- 1) The sorption of precomplexed Ni by SiO_2 (30 g/L) was near zero with either 1×10^{-4} or 1×10^{-3} M picolinic acid at pH 7, compared to ~30% in the absence of complexant.
- 2) Citrate (1×10^{-4} M) had a short-lived effect on Ni sorption by SiO_2 (30 g/L) at pH 7. This result was first indicated in normal experiments and

was subsequently verified in an experiment in which light was excluded (and in which centrifugation was used to separate phases). In the absence of citrate, 27% of the Ni was sorbed. Citrate was then added, and Ni sorption was followed with time. The Ni sorption decreased initially (only 5% was sorbed after 1 or 2 days) but then returned to near the no-complexant value after several days (24% sorption was measured 6 days after citrate addition). Earlier work (Swanson 1982, 1983a) showed similar short-lived citrate effects in different soil systems, and it was noted that citrate would thus not be likely to cause excessive radionuclide migration problems.

- 3) The effect of EDTA on Ni sorption by TiO_2 is similar to that observed with SiO_2 rather than that observed with $\text{Fe}_2\text{O}_3 \cdot x\text{H}_2\text{O}$ (Swanson 1984); little or no Ni sorption was measured from 1×10^{-4} M EDTA onto 30 g/L TiO_2 at pH 5 or at pH 7.

Sorption of Ni Complexes by Ion Exchange Resins

An important trend in reactor decontamination operations is toward the use of dilute complexant solutions, with subsequent sorption of both nuclides and complexants onto ion exchange resins for disposal. We are especially interested in whether resins sorb radionuclide complexes, which are a major concern in radionuclide migration, and have performed some preliminary investigations in this regard.

In a typical reactor decontamination operation, the solution is recirculated around a loop containing a cation exchange resin to sorb the metal ions removed from the contaminated surfaces. At the completion of the decontamination, the solution is discharged through both cation and anion exchange resins to sorb the complexants as well as the dissolved metal ions. We have done experiments with both a strong acid cation exchange resin (Bio-Rad AG-50W) and a strong base anion exchange resin (Bio-Rad AG-1).

Ni sorption by the cation resin at pH 2.1 was quite high in the absence of complexant ($K_d \sim 10^5$ mL/g) but was markedly reduced by the presence of EDTA; K_d values of about 700 and 60 mL/g were measured with 1×10^{-4} and 1×10^{-3} M EDTA, respectively. To learn whether Ni-EDTA complexes were being sorbed, the sorption of ^{14}C -labeled EDTA was measured in the presence and the absence of

Ni. Essentially identical EDTA sorptions were measured with 0.01 M NaNO_3 solutions containing either only 1×10^{-4} M EDTA or 5×10^{-5} M Ni in addition to 1×10^{-4} M EDTA, indicating that Ni-EDTA complexes were not sorbed by the cation resin. The EDTA sorption was surprisingly high in these experiments, giving K_d values of about 100 mL/g.

Very little, if any, sorption of Ni by the anion resin occurred in the absence of complexants. However, Ni was strongly sorbed when 1×10^{-3} M EDTA was present ($K_d \sim 4 \times 10^3$ mL/g at pH 2.6), indicating that Ni-EDTA complexes are sorbed by the anion resin. If these complexes are eventually released by the resin into a soil system where they are kinetically inert and not sorbed by the soil, they could migrate at rates approaching the rate of groundwater.

OTHER RESULTS

The literature regarding the presence of nonradioactive toxic elements in commercially generated low-level wastes was surveyed to identify candidate elements to be included in this study. Of special interest were those whose mobility would be expected to be affected by organic complexants.

There appears to be very little valid information on the identities and the quantities of nonradioactive toxic elements in low-level wastes. A study of the chemical toxicity of low-level wastes was performed several years ago by General Research Corporation (1980), but that study emphasized toxic organic compounds. The report did say "...selected inorganics that are representative of metals found in LLW burial sites include barium, cadmium, chromium, copper, lead, and zinc," (p. 105) and "The three elements of greatest concern (in rank order) appear to be cadmium, copper, and chromium." (p. 140). It is to be stressed that the report did not say that these elements are present in LLW in sufficient quantities to be of concern; it said that if these elements are present in sufficient quantities, they could be of concern.

The question of chemical toxicity of low-level wastes has also been addressed by Wickham (1983), who is attempting to develop a waste classification system that evaluates the relative total hazard of chemical and radiological waste mixtures. Wickham provides evaluation of the total hazard of two types of low-level wastes; evaporator bottom concentrates, and reactor

control rods containing 80% silver, 15% indium, and 5% cadmium. The chemical hazard was judged to be insignificant relative to the radiological hazard with the evaporator concentrates, but the chemical hazard of the silver and the cadmium was judged to outweigh the radiological hazard with the control rods.

Because of the lack of valid information on the presence of nonradioactive toxic elements in low-level wastes, the selection of elements to be studied in this project is complicated. Cadmium appears to be a reasonable choice and will be studied first. However, we do feel that it would be prudent to give cadmium studies a lower priority than the studies with the radioactive toxic elements.

CONCLUSIONS

The previously indicated sorption of a Pu(IV)-EDTA complex by hydrous ferric oxide was substantiated by additional experiments that compared the sorption of EDTA in the presence and absence of Pu. Nickel-EDTA complexes are sorbed by anion exchange resins, and thus are likely to be present in the resin wastes from certain reactor decontamination solution clean-up operations. The question of kinetic inertness of Ni-EDTA complexes in the systems being studied here remains to be answered.

RECOMMENDATIONS

The results of the previous work lead to the recommendation that requirements for waste disposal should not be the same for all organic complexants used in waste treatment, since some complexants will have much less potential impact on toxic element mobilities than others. Work is continuing so that we expect to substantiate this recommendation in this project.

Efforts to obtain basic mechanistic information should be continued and expanded. Only in this way will it be possible to develop adequate and defensible predictive models of toxic element migration.

Another recommendation is that other investigators make use of the experimental procedures developed in our work to probe into the validity and meaning of toxic element distribution measures. Use of these methods would

reduce the number of incorrect values reported and would give increased confidence to using the correct values to predict mobilities in the environment.

PLANNED RESEARCH

Studies of the behavior of Ni and Pu with additional complexants and additional soil components (e.g. clays) will be expanded. The question of kinetically inert complexes will receive special attention since such complexes are of greatest concern to toxic element mobility. Studies will be initiated with cadmium, a nonradioactive toxic element that may be present in low-level wastes.

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