This document was prepared in conjunction with work accomplished under Contract No.
DE-AC09-76SR00001 with the U.S. Department of Energy.

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DPST-84-554

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STRONTIUM SORPTION ONTO SRP SOILS

SUMMARY

The effect of water and soil quality variables on the sorption of strontium onto SRP soils was evaluated. The variables cover the range of conditions observed in the low-level waste burial ground. The sorption coefficient of strontium is a direct function of clay content. Trench water concentrations of the divalent cations Mg^{2+}, and Ca^{2+} as well as the monovalent K^+ decrease the strontium K_d by a factor of three. The ions Na^+, Cl^- and NO_3^- have no effect. The pH of the water in contact with the soil is a major factor influencing K_d. In fact, the decrease in K_d caused by the cations Mg^{2+}, Ca^{2+}, and K^+, and the decrease caused by lower clay content of the soil can be explained completely by the observed pH changes.

The low Eh and high dissolved iron found in some trench waters was simulated by adding Fe^{2+} to the groundwater/soil system. A drastic drop in pH and Sr K_d occurred. Adjustment of the pH to the near neutral pH's of 6 to 7 observed for most trenchwaters resulted in a strontium K_d (approximately 1200 mL/g) lower than the pH 6.5 groundwater/soil K_d (approximately 2200 mL/g) but much higher than the actual trench water K_d's (7 to 300 mL/g). The limited strontium sorption may be a result of organics, iron, or the high ionic strength present in some of the trench waters. How organics might decrease strontium sorption is unknown at this

point, since strontium is only weakly complexed by most organic chelaters.

INTRODUCTION

To verify satisfactory containment performance of the SRP low-level radioactive waste burial site, the migration of radionuclides from the waste has been monitored since 3 years after the site's inception.[1] An understanding of the migration process and the potential concerns associated with it is being obtained from lysimeter and laboratory studies and from transport modeling.[2]

The laboratory studies provide much useful information.[2] The distribution coefficients obtained from these studies can be used in transport models such as the Dose-to-Man code to predict doses to individuals and to populations based on various scenarios.[3] The sorption data can be compared with results obtained from the lysimeters and monitoring to establish the validity of the laboratory data. The flexibility available in the laboratory allows one to cover the conditions expected or known in the burial ground. Finally, laboratory studies provide a means to understand the fundamental chemical process of the soil/waste/water system of the SRP burial ground. And to a limited extent, generic technology on the migration of radionuclides from an operating shallow land burial site in a humid region may be developed.

The radionuclides Sr-90, Tc-99, Cs-137, I-129, Pu-238, Pu-239, Ru-106, and Sb-125 are considered important because of their expected mobility, long lifetime, and/or toxicity.[4] The nuclide of interest in this report, strontium-90, has a half-life of 28.1 years, and has a slight potential for migration. The estimated inventory of strontium-90 in the burial ground is about 11,000 Ci.[5] If ingested by man, strontium substitutes for calcium in bone tissues making toxicity also an important concern.[6]

A large amount of laboratory research using SRP soil and groundwater is reported for strontium-90.[7,8,9] Ryan found that strontium and cesium sorption were strongly influenced by trench waters, i.e., perched waters that occur in the waste trenches.[9] K_d's significantly above and below that obtained for a typical groundwater were observed. A study was conducted by Oblath to determine what effect each of the cations present at elevated levels in the trench water have on the cesium K_d.[10] A similar study for strontium is the main thrust of this work. Previous SRP strontium work is summarized first. Data from well and trench water monitoring, lysimeters, and laboratory studies are included. The second section, on current work, addresses the following topics: comparison with previous work, effect of experimental conditions on strontium K_d, effect of ionic composition on strontium K_d, effect of iron on strontium K_d, effect of pH on strontium K_d, and comparison of data to trench water results.

Summary of Previous Strontium Work

Site monitoring, lysimeter studies, laboratory studies, and transport modeling are being used to better understand radionuclide migration from waste at the burial ground. The principal method used to monitor radionuclide migration is gross alpha and beta analysis of the groundwater. Insignificant amounts of nonvolatile beta emitters are in the groundwater, usually less than 50 pCi/L.[2] Aside from tritium, strontium-90 is one of the principal beta emitters in the groundwater beneath the burial ground,[2] although sometimes the activity can be attributed to natural uranium and its decay products or to previous contamination from spills of spent solvent rather than to the leaching of the trench wastes.[11] In a limited survey of 20 wells, five contained strontium-90 above the 6 pCi/L detection limit.[12,13] One well contained an anomalously high value of 1600 pCi/L, but the other four wells contained only 7 to 30 pCi/L of strontium-90.

Low levels of beta-gamma emitters are also in the perched trench waters. Much of this activity can be due to strontium-90.[12] Overall, migration of the strontium in the burial ground is slight. It has been estimated that, in all of the 195-acre burial ground, less than 16 mCi of nonvolatile beta-gamma activity has migrated from the waste trenches to the groundwater.[2]

Lysimeters have been used for several years at SRL to study nuclide migration.[13] Five gamma emitters are in the lysimeter effluents so far: Mn-54, Co-60, Ru-106, Sb-125, and Cs-137.[2] Strontium-90 is also expected to be present but significant quantities of Co-60, Sb-125, and Ru-106 in the lysimeter effluent precludes the use of gross nonvolatile beta measurements as an estimate of the strontium-90 concentration. A method to separate and concentrate Sr-90 from these nuclides has been recently developed.[14] Using this method, strontium-90 concentrations greater than 1 pCi/L have been found in 11 of the lysimeter effluents and of these, three had greater than 100 pCi/L, the highest concentration being 775 pCi/L.

There are several SRL studies of strontium-90 sorption. An early work by Prout investigated the effects of pH, strontium concentration, NaOH, $Ca(OH)_2$, NH_4OH, and $NaNO_3$ on the sorption.[7] Some of the data are illustrated in Figure 1. Notice the strong pH and concentration dependence. Distilled water was used in these studies so their applicability to the burial site is somewhat uncertain, although the data are useful for showing relative effects. Batch K_d's were also compared to column retention factors, and overall agreement was good (see Figure 1). Ryan later repeated the comparison using undistrubed soil columns and a typical groundwater from the burial ground.[8] Agreement between batch and column results was good, although the batch K_d's were consistently somewhat higher. The most recent study by Ryan determined strontium and cesium batch K_d's in the presence of several trench waters. Strontium trench water K_d's ranged from 1/3 up to nearly 15 times the average groundwater K_d of 20 mL/g. The study indicates how important the groundwater can be in determining the sorption characteristics of a radionuclide.

Current Work

The first step was to verify that previously reported results could be reproduced. Once this foundation had been established, it then was appropriate to compare this work to earlier results. The second step was to study how strontium sorption is affected by the variation of experimental conditions (e.g., groundwater composition, clay content of the soil, etc.). This allowed all the SRP site-specific strontium data to be drawn into a cohesive body of data. The final step was to determine how strontium K_d varies with pH in the presence of groundwater, similar to the Prout study done in the presence of distilled water.[7] Standard batch techniques were used in all studies. Details of the experimental procedure are given in the appendix.

Comparison of Current Studies With Previous Work -

Distilled water and groundwater K_d's for strontium can be compared to previously reported results. Distilled water comparisons are advantageous because they eliminate the variable of groundwater composition. For distilled water/soil equilibrations, K_d's of 310 and 400 mL/g were reported by Ryan[9] and Prout[7], respectively. These compare favorably with 360 and 390 mL/g obtained in the current study.

Reported groundwater K_d's of 1.1 to 10.9 mL/g[8] are significantly lower than distilled water K_d's. The range of K_d's may be a result of the variations in the clay content of the soil. (The soils ranged from 4 to 14% clay). In current study, using a soil containing 9% clay, distribution coefficients of 1-5 mL/g were obtained. As a further comparison, a portion of a soil which remained from the earlier Ryan studies gave a strontium K_d of 20 mL/g. The original value was 10 mL/g.[8] The agreement is good considering that soil had been stored for over a year. Even a different soil with a similar clay content gave a K_d of 15 mL/g, but only if the groundwater was adjusted with sodium bicarbonate to give the same final pH of 6.7. Overall, then, the reproducibility of experimental results is good.

Effect of Experimental Conditions on K_d -

Although the procedure used to determine a K_d is generally simple and straightforward, the results obtained are very much dependent upon the conditions of the experiment.[15] Conditions considered worthy of investigation were the percent sand/silt/clay (% s/s/c) of the soil and the method of soil/water separation.

The % s/s/c of a soil can influence radionuclide sorption by the increase in total surface area which results from decreasing particle size.[16] Soils at SRP with near 0% clay, up to 60%, and possibly even as high as 70-80% clay (in some local areas) are known.[17] Soils of 89/2/9 % s/s/c and 72/5/23 % s/s/c were available for laboratory study. Results are shown in Table I. The strontium K_d increases roughly a factor of three for a similar increase in the % clay.

To understand what effect, if any, the method of soil/water separation has on the amount of radionuclide remaining in the aqueous phase, the results of at least two independent methods should be compared. The effects of filtration and centrifugation on strontium K_d were considered. Table II shows the various combinations investigated. Under all these conditions the strontium K_d was nearly constant, varying from 11.5 to 12.7 mL/g. This implies that either there are few particles in the micron-submicron size range or that only a small amount of strontium sorbs to these smaller particles. Regardless of the reason, the results indicate that, under these conditions, the method of soil/water separation has very little influence on the results obtained.

Effect of Ionic Composition on Strontium K_d -

No systematic study of the effect of ionic composition on strontium sorption has been reported. However, the influence of ionic composition is evident from studies such as Ryan's.[9] He found that strontium and cesium K_d's changed radically with water composition. A later study by Oblath clearly delineated which ions influenced cesium sorption.[10] A similar study has now been performed for strontium and the results are reported below.

The effects of four cations and three anions on strontium K_d are summarized in Table III. Conclusions drawn from the cation and anion data are:

1. Two anions, Cl^- and NO_3^-, have little or no effect on strontium sorption. (Compare NaCl to $NaNO_3$, KCl to

KNO_3, and $MgCl_2$ to $Mg(NO_3)_2$). The increased K_d in the presence of Na_2SO_4 may be a result of the increase in pH.

2. Sodium concentrations up to 2.5×10^{-3} $\underline{M}$ have little or no effect on strontium sorption.

3. The cations Mg^{2+}, Ca^{2+}, and K^+ cause similar decreases in strontium sorption with increasing cation concentration.

Effect of Iron on Strontium K_d -

Large amounts of iron are present in several of the trench waters. There are several pathways by which radionuclide sorption may be influenced by iron. As Fe^{2+} it may compete with other cations for ion exchange sites. (This is somewhat speculative as there is little reported about such behavior for Fe^{2+} in aqueous solutions.) Fe^{2+} also provides a low-Eh environment which becomes important if radionuclides capable of more than one oxidation state are present. If the water becomes oxidizing, Fe^{3+} forms and is rapidly hydrolyzed, resulting in an increase in the acidity of the solution:

$$Fe^{3+} + H_2O \rightarrow Fe(OH)_3 + 3H^+$$

As the iron precipitates, co-precipitation of other cations, including radionuclides, is possible. Such a process could occur if the reduced trench waters migrated from the trenches into the more oxidized groundwaters.

The complexity of iron chemistry in groundwater systems necessitates a rather thorough investigation. Variables considered here are 1) use of quinhydrone to maintain constant reducing conditions, 2) the form of iron salt used in stock solutions, 3) iron concentration, 4) time dependence of iron systems, 5) effect of Eh and pH on the iron systems.

These studies were conducted under oxic conditions. No attempt was made to exclude oxygen from the system, e.g., by using an inert-atmosphere glove box. As such, not all trench water conditions are reproduced. There are several reasons why such an approach was taken:

1. All trench water samples are exposed to air during sampling.

2. A large amount of iron precipitation occurs in trench water samples collected in glass BOD bottles, usually

within a few hours of sampling. This indicates that, in spite of the fact that air exposure has been kept to a minimum, there is extensive oxidation of the trench water samples.

3. When trench water samples are collected in glass BOD bottles, results agree well with Ryan's anoxic studies.[9]

4. This method is simple.

5. Oxidation of trench water simulates the process that occurs when anoxic trench waters migrate into the more oxic surrounding groundwaters.

Quinhydrone, an Eh/pH buffer composed of equimolar amounts of hydroquinone and quinone,[18] had little or no observable effect on strontium sorption (see Table IV A).

Initial studies were done using ferrous ammonium sulfate, $(NH_4)_2 SO_4 \cdot FeSO_4 \cdot 6H_2O$, also known as Mohr's salt, which when dissolved in water forms a relatively stable solution of Fe^{2+}.[19] Later studies were done using the more rapidly oxidized ferrous salt $FeCl_2 \cdot nH_2O$. No consistent differences were observed (see Table IV B).

Several iron concentrations were considered. The lowest concentration of iron, 20 ppm, had essentially the same effect as the higher concentrations (see Table IV B).

The effect of time on Eh, pH, and K_d is summarized in Table IV C. Long-term studies allow time for a substantial fraction of the Fe^{2+} to be oxidized to Fe^{3+}. This more closely simulates trench waters where, after the water is removed from the trench, large amounts of iron hydroxides precipitate out. Table IV C shows there may be a slight trend towards increased sorption at longer times. Results imply that little or no coprecipitation of radionuclides occur with the precipitation of iron. A similar independent study also found that coprecipitation of Co-60, Sr-85, and Cs-137 with ferric oxyhydroxide was minimal.[20]

Effect of pH on Strontium K_d -

The importance of pH on strontium sorption was shown long ago by Prout, but he had used distilled water rather than a groundwater.[7] The pH was a major factor influencing strontium sorption. As such, much more research was done in this area.

Both OH^- and HCO_3^- can be important in controlling the pH of natural waters. Their effect on strontium sorption when used to adjust the pH is shown in Figure 2. There is no significant difference between the two. Surprisingly, using distilled water rather than well I-17 groundwater caused little change in the K_d/pH curve (Figure 3).

To determine the effect of the soil on strontium sorption, K_d/pH curves were generated for a soil with 9% clay and another with 23% clay. The results, shown in Figure 4, show only a difference at high pH. Note that for Figures 4-7 the shape of the curve around pH 8 is inferred from Figures 2 and 3. At this higher pH, the low-clay soils centrifuged very well but in the high-clay soils a significant amount of clay remained suspended, even after long periods of centrifuging. The lower K_d seen for the high-clay soil at the highest pH is almost certainly a result of strontium bound to the suspended clay. Note that the differences in strontium K_d with clay content which were reported earlier in this paper can also be explained by pH differences.

The importance of pH is further emphasized by considering again the cation data. The ions K^+, Ca^{2+}, and Mg^{2+} all caused a decrease in strontium sorption and a small increase in pH. When these data are superimposed on the strontium K_d/pH curve (Figure 5) it becomes apparent that all of the change in K_d is a result of the pH change.

The sorption of strontium in the presence of iron is plotted on the same graph. After the pH effects are accounted for, it is seen that iron causes a limited decrease in strontium sorption (e.g., from 2000 to 1100 mL/g). This may be due to the increased ionic strength of the solution. (NaOH was added to compensate for the acidity resulting from hydrolysis of the iron.)

Comparison of Data to Prout's Work -

The difference between the K_d/pH curve in current study (Figure 2) and in Prout's work (Figure 1) can be explained by differences in experimental conditions (Table V). Of the five differences, three are known to affect the pH; only one affects the K_d range. The effect of these experimental conditions on the shape of the K_d/pH curve is illustrated schematically in Figure 6. The measurement of pH before or after equilibration with the soil is a major factor in determining the shape of the K_d/pH curve. The soil resists any changes from the normal pH of 4.8 for the groundwater/soil system. As an example, a groundwater adjusted to pH 9.85 with OH^- has a pH of 6.15 after equilibrating with SRP soil for one day. Since the soil acts as a pH buffer the amount of soil or the groundwater/soil ratio will influence the final pH. The 2-hour equilibration time in Prout's work was probably too short for the full sorption capacity for strontium and pH buffering capacity of the soil to be realized.

Comparison of Data to Trench Water Results -

Because pH is such an important factor, Ryan's trench water data[9] are also plotted on a K_d/pH graph (Figure 7). All trench water K_d's are less than or equal to the well I-17 groundwater K_d's. There are at least three possible explanations for the decrease in K_d. Part of the decrease is due to the iron present (Figure 5). Part of the decrease may also be a result of increased ionic strength. This idea is supported to some extent by the significant correlation between conductivity and strontium K_d (Figure 8). Finally, it may be that organics are present in the trench water which decrease the strontium sorption. Further work in these areas should help to more clearly define their relative importance.

CONCLUSIONS

From this current study it has been found that:

1. Strontium K_d's obtained using distilled water or well I-17 groundwater compare well with previously published work.

2. The percentage decrease in the clay content of a soil correlates with a corresponding decrease in strontium K_d.

3. The method used to separate the water from the soil had no effect on the results.

4. 75 ppm of Mg^{2+}, Ca^{2+}, and K^+ decreased strontium sorption by up to two-thirds. The ions Na^+, Cl^-, and NO_3^- had no effect.

5. Addition of Fe^{2+} to the groundwater/soil caused a drastic decrease in K_d and pH.

6. The pH of the water was a major factor influencing strontium sorption. The pH changes are sufficient to explain the decrease in K_d caused by Ca^{2+}, Mg^{2+}, or K^+ and are sufficient to explain most of the change in K_d caused by Fe^{2+}. Even the K_d changes observed with clay content of the soil are mainly a result of pH changes.

7. The discrepancy between current work and Prout's data for K_d as a function of pH appears to be a result of differences in experimental conditions.

8. The decrease in strontium K_d in the presence of trench waters relative to the groundwater K_d's may be caused by the iron present in the trench waters, increased ionic strength, or organics in the trench waters.

QUALITY ASSURANCE

All of the data from these studies is recorded in accordance with the SRL Procedures Manual and Division QA guidelines. All work was performed by trained personnel using standard laboratory procedures. All reagents were analytical reagent grade and used as supplied. Working standards were submitted to Analytical Services on a regular basis to insure valid gamma spectroscopy.

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TABLE I. EFFECT OF PERCENT SAND/SILT/CLAY ON STRONTIUM K_d

SOIL	% S/S/C	FINAL pH	Sr K_d, mL/g
A	89/2/9	4.6	3.0, 2.6
B	72/5/23	4.7	9.2, 9.9, 8.9

TABLE II. EFFECTS OF CENTRIFUGATION AND FILTRATION* ON STRONTIUM K_d UNDER VARIOUS CONDITIONS[++]

Condition A	Condition B
1. Before using stock groundwater mix and stir the solution to suspend the settled clays.	1. Do not mix groundwater but decant the water leaving the clay settled on the bottom.
ASK = 12.0	ASK = 12.3
2. Use groundwater directly.	2. Filter groundwater before use.
ASK = 11.5	ASK = 12.7
3. Centrifuge final solution for 15, 30, or 60 minutes.	3. Filter final solution.
ASK = 12.1	ASK = 12.1

* All filtration through a 0.45 m membrane filter

[++] ASK = average strontium K_d, mL/g

TABLE III. THE EFFECT OF VARIOUS SALTS ON STRONTIUM SORPTION ONTO SRP SOIL

Salt Added	Added Cation Conc. M x 10^4	Final pH	Strontium K_d
NaCl	2.5	5.0	26
	25	5.0	23
$NaNO_3$	2.5	5.0	24
	25	5.0	22
KCl	2.5	5.0	20
	25	4.7	7
KNO_3	2.5	5.0	21
	25	4.8	7
No Salt	-	5.0	22
$MgCl_2$	2.5	4.9	25
	25	4.6	5
$Mg(NO_3)_2$	2.5	4.8	14
	25	4.5	6
$CaCl_2$	2.5	4.8	17
	25	4.6	5
Na_2SO_4	2.5	5.2	52
	25	5.5	86
No Salt	-	4.9	16

TABLE IV. THE EFFECT OF FERROUS ION ON STRONTIUM SORPTION ONTO SRP SOIL

A. Effect of quinhydrone, QH, in the presence of 100 ppm Fe

	Strontium K_d
Sr in I-17	22
Sr in I-17 + QH	25
Sr in I-17 + Fe	0.0
Sr in I-17 + Fe + QH	4

B. Effect of iron salt on strontium K_d

	Mohr's Salt		Ferrous Chloride	
	Sr K_d	w/QH	Sr K_d	w/QH
I-17	22	25	23	26
I-17+20 ppm Fe	2	15	2.6	1.4
100 ppm Fe	2	4	0.0	1.5
200 ppm Fe	3	2	1.0	0.6

C. Kinetics of strontium sorption in the presence of $FeCl_2$

Elapsed Time (days)	Sr K_d	pH	Eh, mV
1	4	3.60	+564
3	2	3.68	+604
6	2	3.52	+595
10	6	3.90	+578

TABLE V. EXPERIMENTAL CONDITIONS OF PROUT AND CURRENT WORK

Experimental Condition	Prout	Current	Effect on Final pH	Effect on Final K_d
Water	Distilled	Distilled, Ground		
pH measurement	Initial	Final	X	
soil/solution, g/mL	3/30	2/30	X	
% sand/silt/clay	80/0/20	72/5/23		
Equilibration time, h	2	24	X	X

FIGURE 1. EFFECT OF pH ON STRONTIUM K_d
COMPARISON OF BATCH AND COLUMN DATA
(Prout, reference 15).

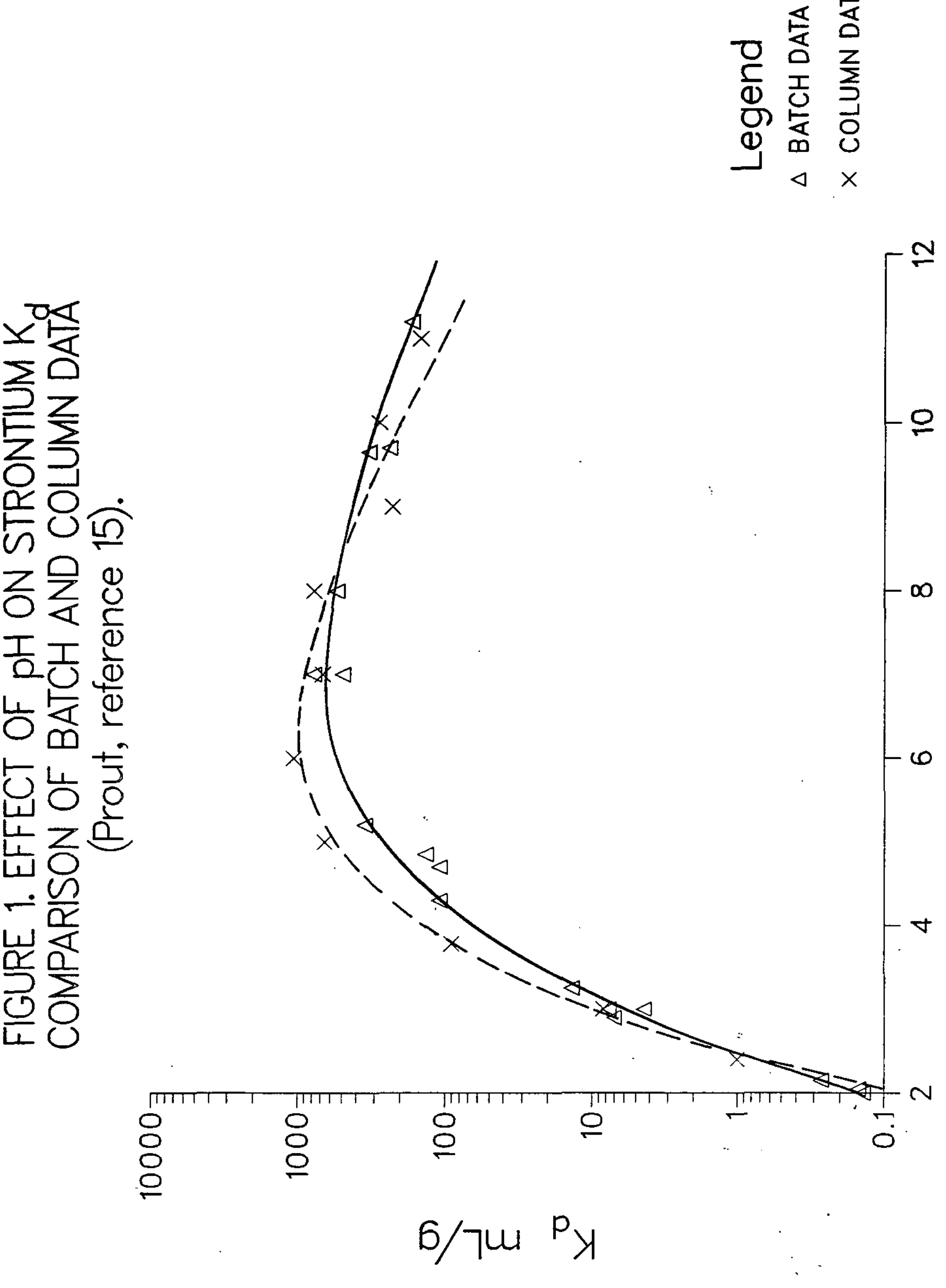
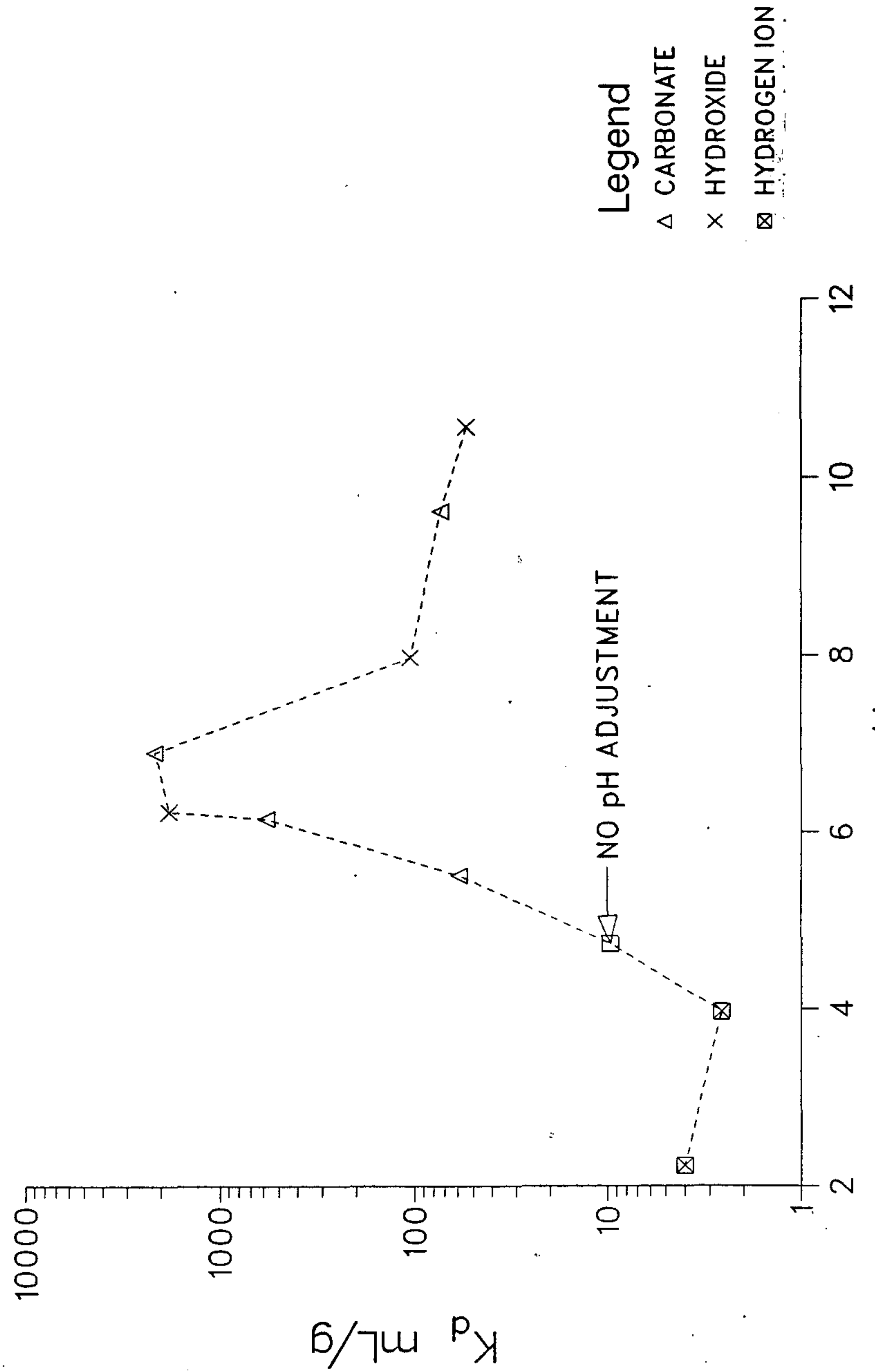


FIGURE 2. EFFECT OF pH ON STRONTIUM K_d



QMS LASERGRAFIX 1200 DATE=84.157 TIME=14.17.02 BADGE=59103 BOX=1004

FIGURE 3. EFFECT OF pH ON STRONTIUM K_d IN THE PRESENCE OF DEIONIZED OR GROUND WATER

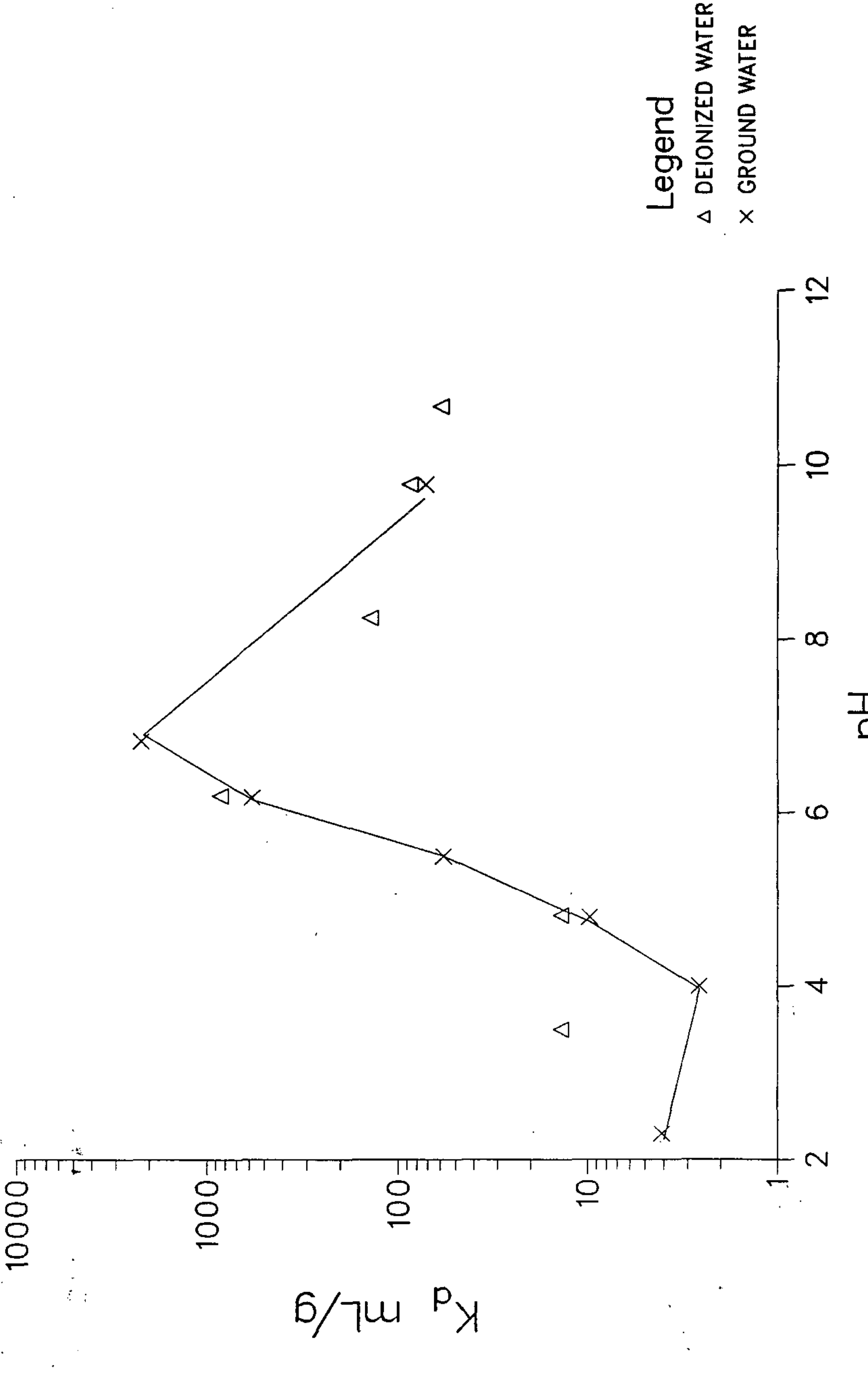
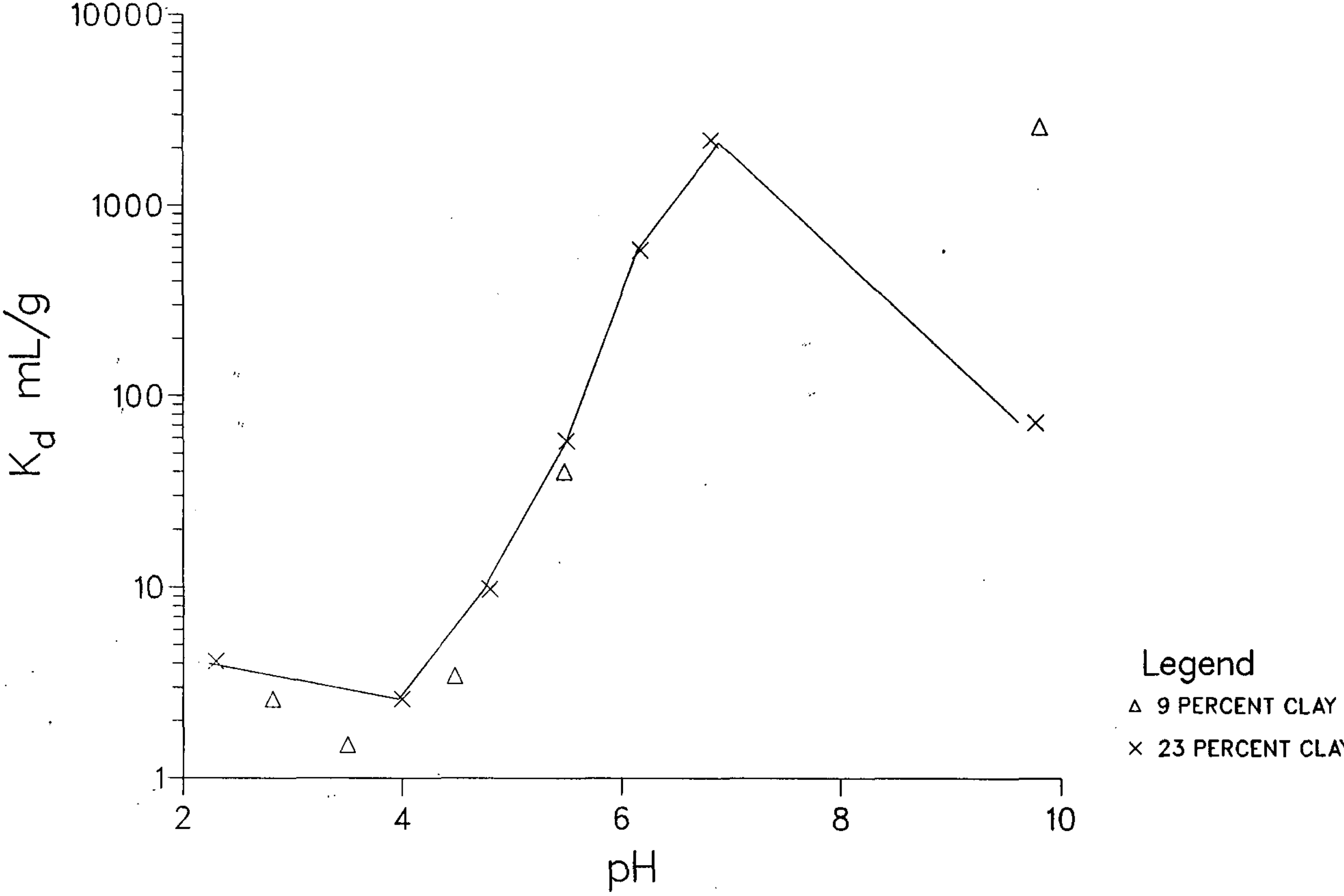
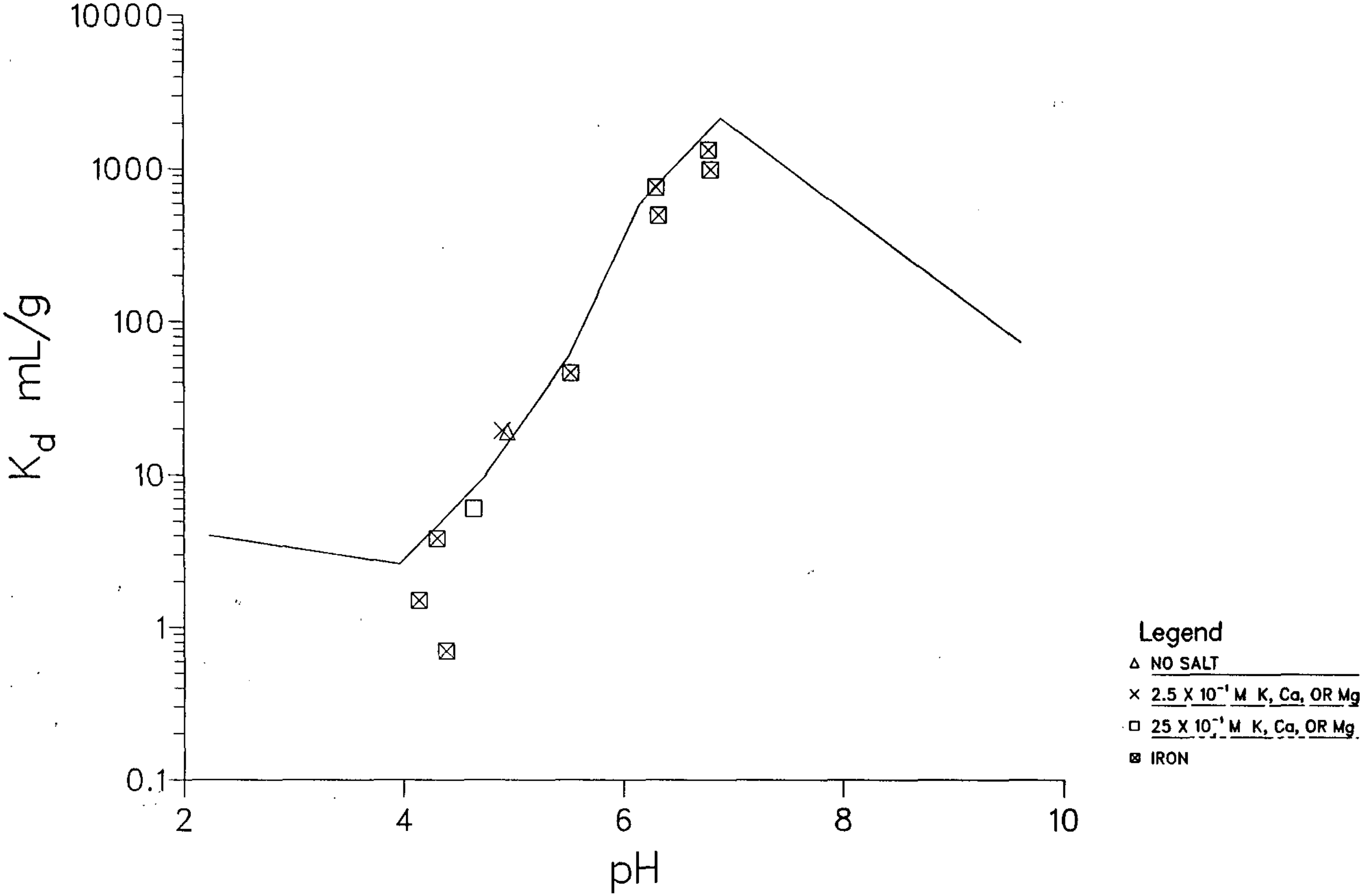


FIGURE 4. EFFECT OF pH ON STRONTIUM K_d IN THE PRESENCE OF SOIL WITH 9 OR 23 PERCENT CLAY



QMS LASERGRAFIX 1200 DATE=84.157 TIME=14.36.04 BADGE=S9103 BOX=L004

FIGURE 5. EFFECT OF IONS AND IRON ON Sr K_d



MS LASERGRAFIX 1200 DATE=84.157 TIME=14.23.58 BADGE=S9103 BOX=L004

FIGURE 6. EFFECT OF EXPERIMENTAL CONDITIONS ON THE SHAPE OF THE K_d/pH CURVE

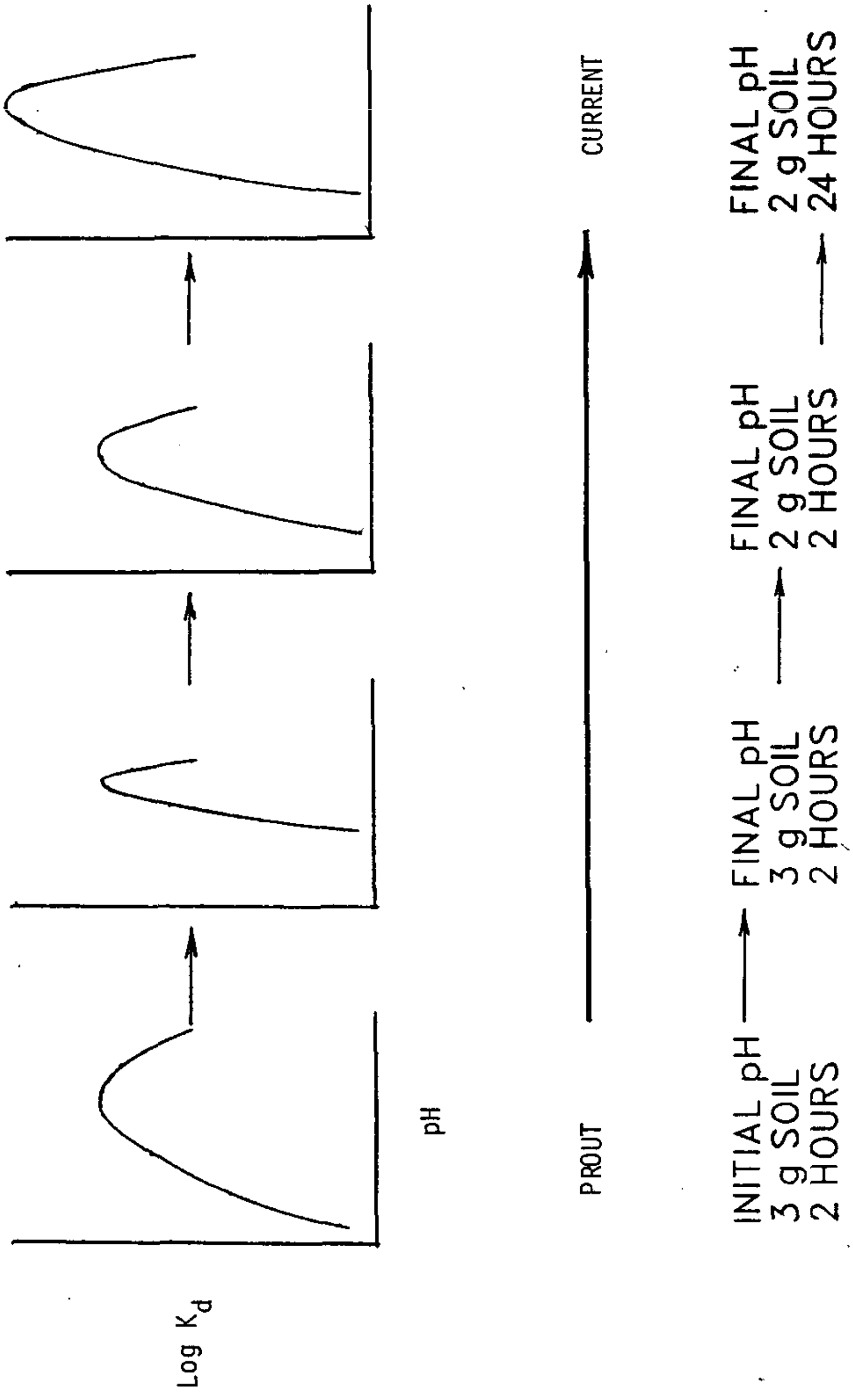
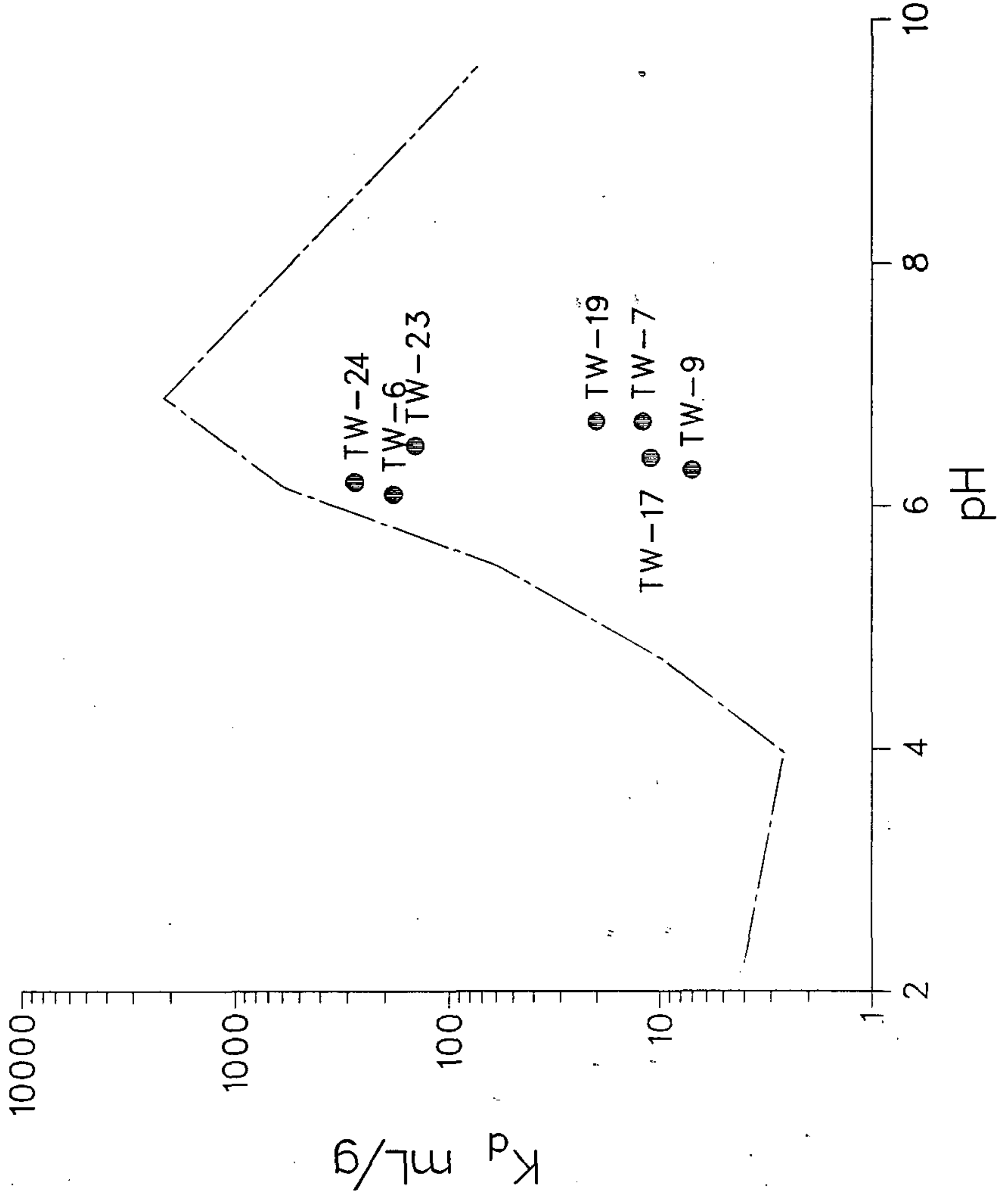
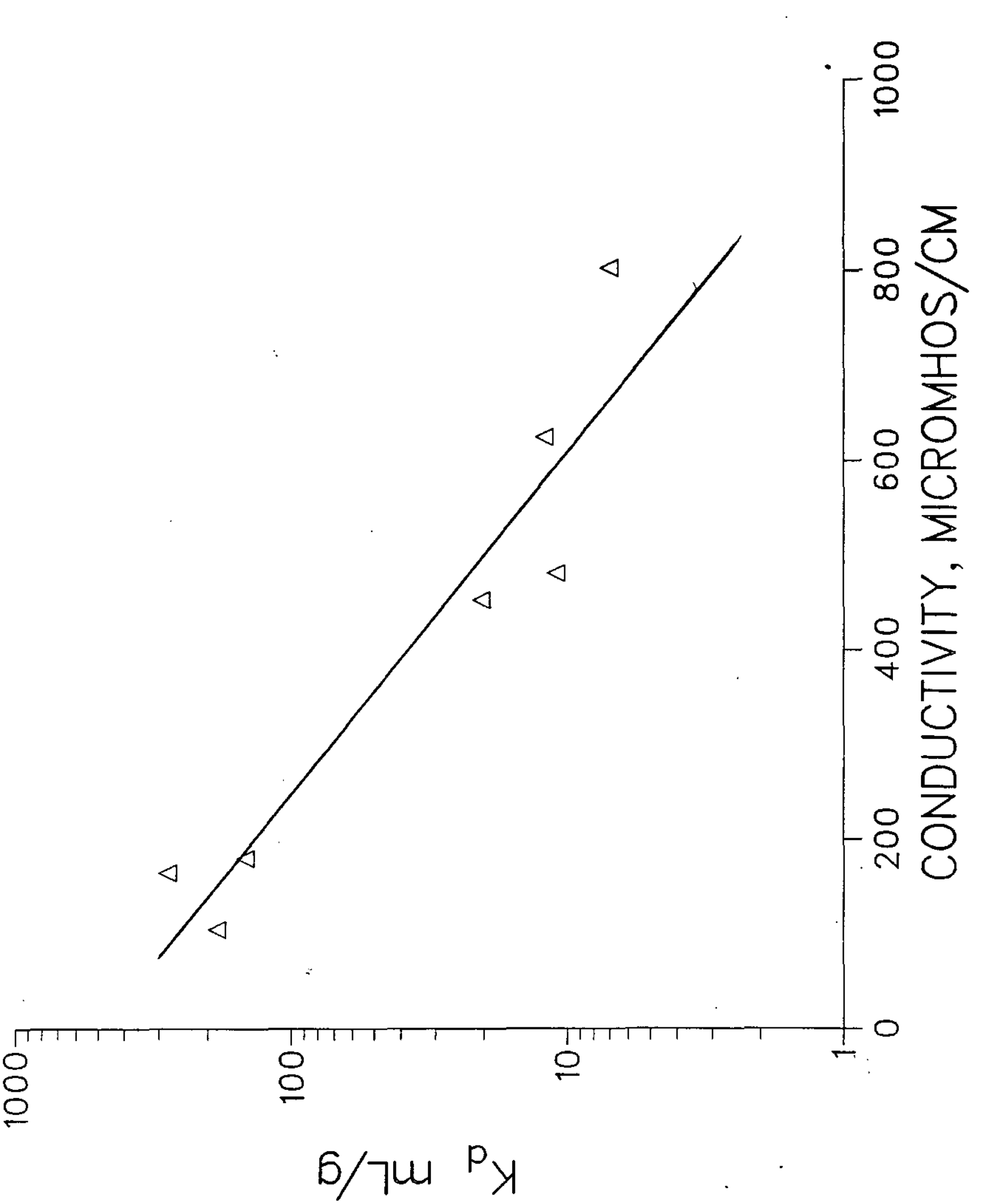


FIGURE 7. EFFECT OF TRENCH WATER ON Sr K_d



DMS LASERGRAFIX 1200 DATE=84.157 TIME=14.37.29 BADGE=S9103 BOX=1004

FIGURE 8. STRONTIUM K_d AS A FUNCTION OF TRENCH WATER CONDUCTIVITY



APPENDIX A

EXPERIMENTAL CONDITIONS AND PROCEDURES

Soil

Most of studies were conducted using a Z-Area soil. The soil selected was 8-10 ft below grade. A second soil, used mainly in initial studies, was formed from a blend of well PDO-5 core samples from depths of 40-42, 44-45, 45-46, 46-47, and 47-48 ft.

The soils were homogenized by breaking up any clumps, separating into an amount easy to handle, and mixing on a plastic sheet for 10-15 minutes by repeatedly folding the sheet over to alternating corners. Also, at various times the soil was randomly moved in small portions from one area to another using a spatula. The small piles were all combined and the total soil was mixed in a similar fashion to give the stock soil.

The soils were double bagged in heavy plastic and stored for use. The dry soil weight was calculated from periodic determination of the moisture content.

Water

Groundwater was collected from well I-17 in 2-L polyethylene bottles. Upon arrival at the lab pH, Eh, conductivity and dissolved oxygen were measured. The water was stored in the 2-L polyethylene bottles until used. Water from the trenches was generally collected in 500 mL polyethylene bottles. Upon arrival at the laboratory, the pH, Eh, conductivity and dissolved oxygen were measured. The water was stored in the refrigerator in glass BOD bottles until time was available to prepare samples for K_d studies, bicarbonate analysis, and sulfide analysis.

Procedure for K_d Determination

The procedure used to determine radionuclide K_d's was to weigh out 2 grams of moist soil into a 50 mL centrifuge cone and then add 30 mL of water (usually I-17 well water but could also be a trench water or distilled water). Any spike additions, other than radionuclide, were added at this time. For example, sodium carbonate was commonly added to raise the pH. In other instances, various salts were added to I-17 well water to raise the concentration of a given cation or anion to that observed in the trench waters.

The centrifuge cone was placed on a reciprocating shaker at 330 rpm for 24 hours. Then the cone was centrifuged for 15 minutes and decanted into a 3 mL glass vial for Sr-85 analysis by gamma spectroscopy. In addition to the internal standards maintained by the ADD counting facility for quality assurance, a control sample containing a known amount of strontium-85 was included with every batch of samples submitted for counting. The distribution coefficient, K_d, was calculated using the equation:[15]

$$K_d(\mathrm{mL/g}) = \frac{(I-S)R}{S} \times M$$

where I = initial concentration of Sr-85 (μCi/L)

S = final concentration of Sr-85 remaining in the solution at the end of the experiment (μCi/L)

R = ratio of water/moist soil, 15 mL/g in these studies

M = moist weight to dry weight soil conversion factor

SLH:ske
Disc 1