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Evaluation of the Migration Potential for ^{60}Co and ^{137}Cs at the Maine Yankee Site

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Background

In July 1999, the Primary Auxiliary Building (PAB) Test Pit at the Maine Yankee Nuclear Power Plant was found to contain 3 feet of water that was contaminated primarily with ^{137}Cs, ^{60}Co, and ^{3}H. The test pit was drained of approximately 360 gallons. Two weeks later, the test pit was re-entered and the water level had returned to 3 feet. Sampling of the water indicated similar concentrations to those found 2 weeks earlier (^{137}Cs at 11,400 pCi/L, ^{3}H at 11,500 pCi/L, and ^{60}Co at 6,000 pCi/L). Almost all of the ^{60}Co was associated with filterable particulates. The two other contaminants were in solution.

The PAB test pit is located on the northeast corner of the containment building, Figure 1. The pit extends down to the 10 foot thick containment base mat (El –2 ft). Below the base mat at El –12 feet is a drainage layer containing 'popcorn' concrete (crushed stone coated with neat cement). This layer is drained by a 6 inch clay pipe to the containment foundation sump (CFS). Another layer of popcorn concrete is found at El – 46 feet. This layer has a much smaller radius than the upper layer and also drains to the containment sump. Groundwater flow modeling indicates that contaminants released from the PAB test pit would enter either one of these two layers and be collected in the CFS.

Sampling of the CFS in July 1999 indicated that ^{137}Cs and ^{60}Co were not detected and ^{3}H levels were 4560 pCi/L. Routine scheduled sampling was begun in February 2000 on approximately a monthly basis. Maine Yankee staff believes that this was possibly too late to detect a tritium pulse from the PAB test pit. In general, neither ^{137}Cs nor ^{60}Co have been detected in the CFS. Two positive measurements for ^{60}Co were just above the detection limit. The source of contamination and its duration could not be determined. However, potential sources included leakage from the PAB past the pit cover gasket and into the pit, leakage through the seal from a contaminated shake space, and use of a contaminated hose to pump down the test pit. The shake space was decontaminated in 1999 after the contamination was found.

Subsequent sampling of the test pit in 2002 showed no detectable ^{60}Co and low levels of ^{137}Cs (< 15 pCi/L on 1/30/02, 11 pCi/L on 2/27/02 and 28 pCi/L on 4/30/02).

Objectives

The objective of this report is to discuss the degree of sorption and desorption of ^{137}Cs and ^{60}Co that may be associated with the granite bedrock and the "popcorn" cement drain system that underlie the Maine Yankee Containment Foundation. The purpose is to estimate how much retardation of these two radionuclides takes place in groundwater that flows in the near-field of the Containment Foundation, specifically with respect to contamination originating at the PAB Test Pit. Specific concerns revolve around the potential for the contamination originating near the PAB to create a radioactive dose to a

hypothetical 'resident farmer' using a well intercepting this water to exceed 4 millirems/yr.

Figure 1 Containment Structure, Test Pit and drainage sumps at Maine Yankee

Approach

Our approach is to use a combination of: site specific observations, simplified transport and geochemical modeling, and information drawn from our own work and the literature. The geochemistry of Cs and Co differs significantly and must be treated separately. As a result, for ^{60}Co we have relied on geochemical modeling of groundwater chemistry, looking specifically for over-saturation of iron minerals and then confirmation of mineral precipitation by observation. For ^{137}Cs the approach is more complex requiring estimates of flow paths and rates to calculate bounding values for the retardation of the ^{137}Cs.

For the cobalt analysis estimates of solubility will be made based on literature information and calculations made using Maine Yankee groundwater chemistry information and the computer code MINTEQA2. A discussion of the range of conditions under which solubility control is likely will be presented. The potential for dissolution of the precipitated cobalt and facilitated transport of cobalt through complexation will be addressed.

For the cesium analysis estimates of sorption on the granite host rock and in the fractures will be based on literature values. Estimates of pore velocities for non-sorbing species (e.g. ^{3}H) and ^{137}Cs will be provided in the fractured granite and in the popcorn concrete regions based on site-specific flow characteristics. The potential for increased sorption on minerals in the fractures and matrix diffusion will be qualitatively addressed. Estimates of ^{137}Cs concentrations in the CFS, resulting from migration of contaminated water from the PAB Test Pit, will be provided.

Transport of ^{60}Co in groundwater at Maine Yankee

Several observations indicate that groundwater transport of ^{60}Co is highly attenuated at the Maine Yankee site. Just after the initial discovery of the contaminated water at the PAB Test Pit, which contained approximately similar concentrations of ^{60}Co and ^{137}Cs, the water was pumped through a filter and sampled. While the filter was apparently contaminated and actually increased the activity of ^{137}Cs in the water sample, all of the ^{60}Co was removed from the sample. This indicates that all of the ^{60}Co was associated with solid phases, either through adsorption or precipitation. In the Test Pit, with its significant exposure to concrete and the bedrock, it is likely that solubilities of Co and Fe are extremely low because of the chemistry of the groundwater. This includes groundwater pH values of about 9.5, which are the result of contact with concrete. Solubilities of these minerals are also low at more natural pH values of around 7, as discussed later. Therefore it appears that the precipitation of Fe minerals at the Maine Yankee Site is not necessarily associated with the presence of concrete and would be expected to take place in the bedrock as well.

At nuclear power stations there is often the concern for the presence of transition metals that are complexed with decontamination reagents, making a very stable and mobile form of the radionuclide. Observation of the readily filterable form of ^{60}Co in the Test Pit

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water indicates that the ^{60}Co was not complexed and was apparently a simple ion that then reacted with other ions and solids.

Analysis of water sampled from the PAB Test Pit, taken in June of 2002, was used as input to the geochemical speciation code MINTEQA2. Setting pH to 9.5, a typical value for water in the Test Pit, it was determined that both Fe and carbonate phases all had positive Saturation Indices. A reasonable charge imbalance of 5% was obtained. These results are shown in Table 1, which is a printout of geochemical species in the system, as calculated by MINTEQA2. A positive value of the Saturation Index indicates that the species is above saturation in that solution, with a potential for precipitation of that specific species. All major Fe phases, such as ferrihydrite, goethite and hematite, were over-saturated. Altering pH to values ranging from 9.2 to 7.0 all provided similar calculation results. One would not expect any discrete Co phases since Co is present only in extremely low concentrations. Instead, Co would substitute for Fe in these minerals and would co-precipitate. Accumulation of Fe minerals in fractures in the granite has been observed, confirming the model calculations.

Table 2 is the result of a MINTEQA2 calculation based on data from Bailey Point well B-203. Concentrations are much lower in this well than in the Test Pit water and pH is 6.4. Nevertheless, iron oxyhydroxides have positive saturation indices in this water, indicating that ^{60}Co is not expected to be mobile. Based on the modeling results, it seems that precipitation of iron oxyhydroxides that form as Fe (II) dissolves from minerals in the granite, is robust; therefore reasonable changes in water chemistry would be unlikely to result in dissolution of secondary Fe/Co phases. Groundwater composition at the Maine Yankee site, in areas not influenced by concrete, is similar to that of groundwater in the Lac du Bonnet granite. Weathering of minerals in this granite, to produce iron oxyhydroxides, is relatively rapid and ^{60}Co sorption increases with increasing Fe bearing weathering products [Walton, 1984]. While reversible in the short term (days), sorption of ^{60}Co on iron oxyhydroxides in the long-term can be essentially irreversible [Cerling and Turner, 1982].

Batch sorption experiments were conducted to examine uptake of Cs and Co on crushed rock that is similar mineralogically to the on–site granite at Maine Yankee [Fuhrmann, 2001]. Values of K_d for Co and Cs were 200 and 1100 ml/g respectively, at a pH of 6.85. Granites from several sites have been tested, under fracture-flow regimes, for their ability to sequester radionuclides. A block of granite from the Lac du Bonnet batholith was used in laboratory experiments to define retardation of radionuclides during flow through fractures [Vandergraaf et al., 1997]. ^{60}Co (as well as Np and Pu) were sorbed or precipitated immediately and did not migrate over an experiment period of 110 days

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Table 1. MINTEQA2 calculation of Maine Yankee PAB Test Pit water at pH= 9.5

Charge imbalance = 5%

Mineral	log IAP	Sat. Index
Al(OH)3 (am)	8.706	-2.094
Al(OH)3 (Soil)	8.706	0.416
Al2O3	17.413	-2.239
ARAGONITE	-5.819	2.517
ARTINITE	5.527	-4.073
Ba(OH)2:8H2O	12.317	-12.077
BOEHMITE	8.706	0.128
BRUCITE	13.686	-3.158
CaCO3xH2O	-5.82	1.325
CALCITE	-5.819	2.66
DIASPORE	8.706	1.833
DOLOMITE (disordered)	-13.978	2.562
DOLOMITE (ordered)	-13.978	3.112
Fe(OH)2.7Cl.3	1.988	5.028
FERRIHYDRITE	6.629	3.438
GIBBSITE (C)	8.706	0.966
GOETHITE	6.629	6.138
HALITE	-8.001	-9.604
HEMATITE	13.258	14.676
HUNTITE	-30.295	-0.327
HYDROMAGNESITE	-18.949	-10.183
LEPIDOCROCITE	6.629	5.258
LIME	16.026	-16.674
MAGHEMITE	13.258	6.872
MAGNESIOFERRITE	26.944	10.085
MAGNESITE	-8.158	-0.698
Mg(OH)2 (active)	13.686	-5.108
NATRON	-6.913	-5.602
NESQUEHONITE	-8.159	-3.489
PERICLASE	13.686	-7.898
PORTLANDITE	16.025	-6.779
SPINEL	31.1	-5.748
THERMONATRITE	-6.911	-7.548
Vaterite	-5.819	2.094
WITHERITE	-9.525	-0.955

Table 2. MINTEQA2 Calculations of Saturation Indices for Water at Bailey Point Well B-203

Mineral	log IAP	Sat. Index
ARAGONITE	-10.382	-2.076
Ca3(PO4)2 (beta)	-34.659	-5.578
Ca4H(PO4)3:3H2O	-56.755	-9.675
CaCO3xH2O	-10.382	-3.271
CaHPO4	-22.095	-2.728
CaHPO4:2H2O	-22.095	-3.032
CALCITE	-10.382	-1.928
Fe(OH)2.7Cl.3	1.949	4.989
FERRIHYDRITE	4.917	1.507
GOETHITE	4.917	4.245
HALITE	-6.499	-8.09
HEMATITE	9.834	10.867
HYDROXYLAPATITE	-47.223	-2.89
LEPIDOCROCITE	4.917	3.546
LIME	9.531	-23.748
MAGHEMITE	9.834	3.448
NATRON	-13.123	-11.615
PORTLANDITE	9.531	-13.657
STRENGITE	-26.709	-0.337
THERMONATRITE	-13.123	-13.791
Vaterite	-10.382	-2.508
pH constrained at 6.4		

Sorption in Fractured Systems

When a dissolved species is sorbed onto a fracture surface, its movement will be retarded in relation to the water. For fracture systems it is more appropriate (Freeze, 1979) to express the distribution coefficient on a per unit surface area basis.

Ka (cm) = mass of solute on the solid phase per unit area of solid phase/
         concentration in solution.

Ka is sometimes reported in units of ml/cm^2.

The retardation equation is now expressed as:

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$$V/V_s = 1 + A*Ka$$

Where Vs is the pore velocity of the sorbing solute, V is the pore water velocity, and A is the surface area to void-space volume ratio. For a plane fracture,

$$A = 2/b$$

Where b is the aperture thickness.

In this case,

$$V/V_s = 1 + 2Ka/b$$

In practice, fractures are of variable thickness and may be coated by weathering minerals. These coatings may have substantially different and often greater sorption capacity than the host rock.

Sorption Data

Sorption in granite systems has been studied at a number of sites. The most relevant data was reported in Vandergraaf, 1997. This study measured sorption of Cs and Co on granite and typical weathering minerals found in granite (Table 3). Their batch K_d value for ^{60}Co is similar to that of Fuhrmann [2001] for material from near the Maine Yankee site, 290 compared to 200 ml/g. For ^{137}Cs there is a greater difference, 99 compared to 1100 ml/g, with the material from near Maine Yankee having much greater sorption. They also examined flow along a fracture surface and measured Ka values as a function of time (Table 4). The data indicate that Ka increased over time for periods of up to 30 days for both Cs and Co.

Table 3. K_d values (Vandergraaf, 1997)

Isotope	Material	Kd (ml/g)
Co ^{60}	Goethite*	1200
	Granite	290
	Illite	2100
	Kaolinite	680
^{137}Cs	Goethite*	20
	Granite	99
	Illite	2400
	Kaolinite	550

* Kd in ml/h.

Table 4. Ka values (Vandergraaf, 1997)

Isotope	Time (days)	Ka (cm)
Co – [60]	1	0.48
	3	1.4
	10	3.5
	30	8.3
^{137}Cs	1	1.4
	3	2
	10	2.7
	30	2.9

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Another major study of sorption in granite was conducted at the Grimsel test facility by Hoehn, et. al (Hoehn, 1998). They injected tracers in a fractured system to examine the effects of sorption and found the following selectivity order Na < Sr < Rb < Cs. Two test lengths, 1.7 and 4.9 m, were used in the study. They determined the reduction in peak measured concentration as compared to the injected concentration and retardation in time for 50% of the injected mass to exit the test facility as compared to a nonsorbing tracer. For Cs, reduction in peak concentration in the 1.7 m tests ranged from 5 to 20 and the retardation in breakthrough time ranged from 68 to 209. In the 4.9 m tests, the reduction in peak concentration was over 6000 and the retardation in breakthrough time was approximately 5450. For all sorbing tracers, the longer flow field experiment exhibited greater retardation. This was attributed to longer times for matrix diffusion to occur. Breakthrough curves for sorbing species were characterized by a relatively early peak with a long tail. This is strong evidence that matrix diffusion played an important role in the transport of the species. In these experiments, flow rates were relatively high (1 m/hr) compared to those expected at Maine Yankee (0.012 m/hr). Thus, matrix diffusion might be even more important at the slower flow rates.   Measured laboratory values for Cs K_d were 134 – 1420 ml/g on the Grimsel Test site granite.

Sorption of Cs onto cements was measured in the early 1980's at BNL. Data from these studies (Neilson, 1982) indicate that minimal sorption occurred for Cs and K_d values ranged from 1 – 2. In these studies, the sorption of Cs on illitic shale was also determined and a K_d value of 53 cm^3/g was measured.

Reversibility of ^{137}Cs Sorption

Of the minerals in granite, sorption of ^{137}Cs is expected to be greatest on the micaceous minerals, muscovite and biotite, and on the related clay mineral illite. These minerals have a pronounced affinity for Cs [Francis and Brinkley, 1976, and Sawney, 1972]. Adsorption of Cs onto these minerals is not completely reversible. The fraction of ^{137}Cs that can be desorbed depends very much on the adsorption time, with longer sorption times resulting in less reversible Cs [Evans et al., 1983].   For example a 180 day sorption period resulted in about 20% recovery of Cs tracer, while only 10% was recovered from sediment containing Cs for 15 years [Evans et al., 1983]. It appears that

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sorption of Cs (as a carrier-free tracer radionuclide) takes place on several types of sites: planar surfaces, frayed-edge sites, and interlayer spaces [Sawney, 1972]. Migration into the inner-layer spaces requires loss of waters of hydration from the Cs atom and proceeds slowly but with very low reversibility. Significant differences in sorption and desorption of ^{137}Cs take place depending on if the mica/illite minerals are loaded with K or Ca prior to Cs sorption [Comans et al., 1991]. Greater reversibility takes place with K-loaded illite. From this, it seems reasonable to conclude that only a small fraction of ^{137}Cs sorbed on micaceous minerals of the granite bedrock would be desorbed. This will depend in part on the background elements associated with the mica/illite and on the time period during which sorption took place. Water at the PAB Test Pit contains 4.55 mM Ca and 3.67 mM K, as a result it is unclear if the degree of sorption/desorption is influenced by these background ions.

Transport of Cs likely will be significantly influenced by minerals that accumulate in the rock fractures. These are weathering products that form as constituent minerals of the granite are contacted with water and are altered. These products can be expected to be iron oxyhydroxides, calcite (depending on pH), and the clay minerals kaolinite and illite. Visual observation confirms that weathering products are present in many of the fractures in granite at the Maine Yankee site. Sorption of Cs onto these minerals will take place but the effect is difficult to quantify. This will be very much dependant on the quantity of these minerals associated with a length of fracture. Nevertheless it is apparent from Table 3 that the illite and kaolinite have K_d values for Cs that are significantly greater than the granite itself. Since water flows through these fractures and necessarily is in intimate contact with the fracture filling minerals, these minerals contribute significantly to the retardation of Cs transport. Analysis of fracture filling minerals in the Lac du Bonnet granite indicates that ^{60}Co was associated primarily with amorphous and crystalline iron oxides, and ^{137}Cs was associated with the clay minerals [Vandergraff, 1997].

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Estimated Pore Velocities

Bedrock
Flow through the granite bedrock will be carried out in the fractures that are present and will be impacted by sorption processes. Values for the distribution coefficient, Ka, of ^{137}Cs have not been measured at Maine Yankee, however, travel times can be estimated using literature values and flow rates. The estimated groundwater flow rate through the granite is 0.0025 ft/d and the effective porosity is 0.001 (Gerber, personal communication). Therefore, the pore velocity is 912 ft/ (278 m/yr).

To estimate the travel times to the CFS requires estimates of the aperture thickness, distribution coefficient along the fracture, and travel distance. The aperture thickness of the fractures is expected to range between 0.1 and 0.01 cm. Based on the data of Vandergraaf, a value of 2.9 is used for Ka. This corresponds to the measured value at 30 days. With these assumptions, the pore velocity for ^{137}Cs is estimated to range from 0.48 – 4.7 m/yr.

Table 5: Estimated ^{137}Cs pore velocity and retardation factors

V (water pore velocity) (m/yr)	Ka (cm)	Aperture width (cm)	Vs (solute pore velocity) (m/y)	Retardation Factor (V/Vs)
278	2.9	0.1	4.7	59
278	2.9	0.01	0.48	581

The precise travel distance to the drain is not known due to the lack of knowledge of fracture pathways in the granite. For the upper drain, it would be on the order of a few meters. For the lower drain, it would be on the order of 20 meters. For transport to the upper drain, the travel velocities indicate that it would take between 0.6 to 6 years. For transport to the lower drain (20 m travel distance), travel times would range from 4.2 to 42 years.

The above numbers do not account for diffusion into the pores of the granite. A measured value for Cs diffusion coefficient in granite from the Mirror Lake, NH site (NAS, 1996) was 6.9 10^{-13} m^2/s. Diffusion into the bulk rock would restrict migration along the fracture even further. This is supported by the Grimsel test data (Hoehn, 1988). Analytical solutions for the pore concentration have been developed for the case of diffusion from the fracture into a porous rock matrix (Chen, 1997 and Park, 2001). Assuming that the distribution coefficient in the bulk rock for Cs is 100 (Vandergraff, 1997) a substantial reduction in travel time may be anticipated.

The estimated travel times are sensitive to estimates for the distribution coefficient, Ka, and the fracture width. If more precise estimates of transport of ^{137}Cs are needed, site specific measurements may be necessary.

Drain System
The upper and lower drain systems were constructed by filling a 6 inch (upper) and 18 inch (lower) excavated region with popcorn concrete consisting of small pieces of

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crushed stone (< 3/8 inch) coated in cement. The drainage layers have much higher permeability and porosity (30 – 40%.) than the granite host rock. Water from the layers is collected into a sump and pumped out to a permitted discharge system. The flow rate to the sump varies from 0.25 – 3 gpm and averages about 1.8 gpm (Gerber, personal communication). Approximately 96% of the flow (1.73 gpm) occurs in the upper drain.

Estimating the pore velocity through the drainage layers is difficult due to the 3-dimensional nature of flow around and through the layer and the lack of precise knowledge of the flow through each layer.

The lower drainage layer is assumed to be 45 feet in diameter, as precise measurements were not provided. Water may enter the layer from above, below or through the sides. Flow modeling indicates that drainage is unlikely to be from above. Assuming that all drainage occurs through the sides provides a drainage area of 212 ft^2 (π *1.5 ft * 45 ft). A larger area would be estimated for flow through the top and bottom of this layer. With an area of 212 ft^2 and a volumetric flow rate of 0.07 gpm, the Darcy velocity through this layer is 23.2 ft/yr (7.1 m/yr). For 30% porosity, the pore velocity would be 77.3 ft/yr (23.6 m/yr).

The upper drainage layer is overlain by the concrete basemat and substantial drainage through this surface is unlikely. This layer is continually drained to air and it is expected that only the bottom ¾ inch of the layer is saturated (Gerber, personal communication). Assuming that all drainage is through the sides and the diameter is 100 feet, the drainage area is 19.6 ft^2 (π * 100 ft* 0.0625 ft). Using this flow area and a volumetric flow rate of 1.73 gpm, the Darcy velocity through this layer is 6200 ft/y (1900 m/y). For 30% porosity, the pore velocity would be 20670 ft/yr (6333 m/yr).

Flow through the drainage layer can be described as flow through a porous material. In this case,

$V/Vs = R$

Where R is the retardation coefficient which is expressed as:

$R = 1 + \rho Kd/\eta$

ρ = bulk density of the popcorn cement = 1.8 g/cm^3
η = porosity = 0.3
Kd = distribution coefficient.

Sorption of Cs on cement surfaces is generally low and measured values for the distribution coefficient are on the order of 1 – 2 (Neilson, 1980). Using a Kd value of 1, yields a retardation coefficient of 7. Therefore, the pore velocity for Cs in the lower drainage layer would be 11 ft/yr (3.4 m/yr). Thus, Cs contamination entering this layer would enter the sump after 4.4 years. In the upper drainage layer, the pore velocity would be 2950 ft/yr. Therefore, Cs contamination entering this layer would reach the

sump within two weeks.. Table 6 summarizes the estimated pore velocity and travel times in the bedrock and drain layers. Total travel time through the upper drain to the CFS is years and through the lower drain is years.

Table 6: Estimated Pore Velocity and Travel Times in different pathways of the system

Material	Path Length (m)	Non-sorbing tracer pore velocity (m/y)	^{137}Cs pore velocity (m/y)	Travel time through the layer (y)
Granite	3	278	0.48 – 4.7.	< 1 –6.25
Upper Drain	30	6333	900	0.03
Granite	20	278	0.48 – 4.7	4.2 - 42
Lower Drain	15	23.7	3.4	4.4

Repeating the estimates for flow through the top and bottom of the drain would increase the surface area, but decrease the travel distance to the sump. In any case, the conclusion that the ^{137}Cs would move rapidly through the drainage system remains unchanged due to the low amount of sorption expected. The estimates provided are sensitive to the value of Kd. Increasing Kd by a factor of 2 approximately doubles the transport time to the sump. If the cement coatings wore off or weathered with minerals that exhibit more sorption for Cs, travel times could increase substantially. The sorption properties of the gravel within the cement coating are unknown.

Estimated Sump Concentrations

The only point in the system that easily can be used to monitor radionuclide concentrations is the CFS. Therefore, this location can be used to assess the effectiveness of sorption and dilution effects in reducing concentration levels from the PAB test pit. The concentration in the sump will depend on the pathway taken to reach the sump, the travel time, and dilution with other waters collected in the sump. Detailed numerical or analytical flow and transport models that incorporates matrix diffusion, and sorption could be used to estimate sump concentrations. However, this is beyond the scope of this project. Instead, bounding estimates based on retardation and mixing will be made to provide an estimate of potential concentrations.

Darcy velocities estimated to be 0.0025 ft/day (Gerber, personal communication). The flow area in the PAB test pit is 12 ft^2. Therefore, the volumetric flow through the contaminated source region is 0.03 ft^3/d (0.22 gpd). In the sumps combined flow rates averaged 1.8 gpm. The low flow through the contaminated zone as compared to the amount being collected in the sumps indicates that substantial dilution will occur. A dilution factor of 8.5 10^{-5} is obtained by taking a ratio of the two flow rates.. With this dilution and an initial concentration of 11,000 pCi/L in the PAB test pit, ^{137}Cs concentrations in the CFS would be less than 1 pCi/L in the sump.

Although the concentration in the sump would be low, concentrations in the host rock could be high in local areas in the granite. Using the worst case scenario, a resident farmer could collect all of the Cs released from the PAB test pit in their well. Assuming

a volumetric flow rate of 0.03ft^3/d at a continuous source concentration of 11,000 pCi/L of ^{137}Cs leads to a release rate of 9300 pCi/d from the PAB Test Pit area. For the resident farmer scenario, water usage is 250 gpd in the absence of irrigation. Therefore, if the entire source entered into the resident farmers well, the concentration in the well water would be 9.8 pCi/L.

The above concentration estimates do not account for retardation, dispersion, and diffusion in the flow paths which would further lower the groundwater and sump concentrations. In addition, matrix diffusion or sorption in the host rock, which would further lower the sump concentrations, are not accounted for.

Knowledge of the flow rate through the PAB test pit is essential to estimating source strength and ultimately sump concentration. Refined estimates of this flow rate are needed to provide a better estimate of potential sump concentrations. Refined estimates of aperture size could also improve estimates of transport.

Conclusions

Analysis indicates the following:
- ^{60}Co is likely to have very low solution concentrations due to low solubility. Under the pH range of 6.4 – 9.5, ^{60}Co will co-precipitate with iron. Dissolution of these secondary phases is not expected under anticipated conditions.
- An upper bound estimate for ^{137}Cs in the sump system of 1 pCi/L has been determined. The upper bound is estimated using only dilution based on the ratio of flow through the PAB test pit and the sump. Other transport effects, sorption, dispersion, and matrix diffusion would make the actual concentrations lower. Transport time to the sump could be less than a year if the upper drain layer collects the ^{137}Cs and the parameter assumptions used in the analysis are valid (large aperture fracture 0.1 cm, and Ka = 2.9). Transport times could be 8 to perhaps 45 years if the transport pathway is to the lower drain.
- Concentrations in the fractures of the granite could be substantially higher than in the sump. An upper bound estimate for ^{137}Cs in a resident farmer's well was obtained based on the daily release rate from the sump and a water usage rate of 250 gpd. In this case, ^{137}Cs concentrations would be less than 10 pCi/L.
- The low solubility of ^{60}Co and the sorption of ^{137}Cs indicate that most of the contamination is associated with the solid phase of the system.

Recommendations

The analysis provided in this paper relies on literature values to assess the transport of Cs in the system. If higher confidence is needed in the analysis, the following measurements should be considered:
- site-specific measurements of sorption on fracture surfaces.
- site-specific fracture geometry and aperture sizes,
- flow rate through the PAB. This is crucial to estimating source strength and dilution.

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- non-sorbing tracer studies to define flow rates out of the PAB test pit and travel times.

The utility of the information provided by these tests needs to be weighed against the costs of conducting the tests.

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