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**Assessment of Radiological Impact of the
Inactive Uranium-Mill Tailings Pile at
Salt Lake City, Utah**

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HEALTH AND SAFETY RESEARCH DIVISION

ASSESSMENT OF RADIOLOGICAL IMPACT OF THE INACTIVE
URANIUM-MILL TAILINGS PILE AT SALT LAKE CITY, UTAH

F. F. Haywood, W. A. Goldsmith, P. T. Perdue,
W. F. Fox, and W. H. Shinpaugh

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URANIUM MILL TAILINGS PILE AT SALT LAKE CITY, UTAH

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ABSTRACT

One of the largest inactive uranium mill tailings piles in the United States is located within metropolitan Salt Lake City, Utah. A radiological survey was performed at this site during the fall of 1975. A series of field and laboratory analyses were performed to characterize the spread of contamination from the tailings pile to surrounding areas. These analyses provided a basis for discussion of the important pathways for transport of radioactivity to man. Measurements of radionuclide concentrations in soil indicated that radium had migrated up to one meter into soil beneath the tailings. Tailings material was found out to distances of several hundred meters, mostly in the prevailing wind directions. Elevated levels of ^{226}Ra , ^{238}U , ^{230}Th , and ^{210}Pb were found in sediments of streams running through the mill site; but in Jordan River samples, radionuclide concentrations were of the same magnitude as background samples collected in other parts of the Salt Lake Valley. Atmospheric dispersion of radon gas, which emanates from the pile continuously, was calculated using a model developed by W. Culkowski. The area source term for this model was determined from available data on radon emanation from tailings sands. Potential health effects for continuous exposure to radon progeny and external gamma radiation from the pile in its current state were estimated using risk estimators presented in the BEIR report. For the 400,000 residents within 11 km (7 mi) of the pile, 0.4 cases of lung cancer per year were estimated. For the same population exposed to natural levels of radon progeny, 7.2 cases per year were estimated. Based on the assumption that cancer induction is linear with dose down to the low dose range observed in this study, the mortality rate attributable to all cancers resulting from whole body gamma-ray exposure of the same population was estimated to be 0.04 deaths per year. The comparable mortality rate due to background gamma radiation was estimated to be 3.6 deaths per year. These potential health effects were extrapolated to medium and high density population projections. For lung cancer, the incidence was 3 and 4 cases per year, respectively. The major health risk was inhalation of radon progeny. The contributions of other pathways to potential health effects were less than the error associated with the estimates of radon concentration.

1. INTRODUCTION

The milling of uranium ore produces a uranium-concentrated product called "yellow cake" from an ore feed containing a few tenths of a percent uranium by weight. The processes involved in a milling operation, discussed in detail elsewhere,^(1,2) are grinding and crushing, leaching uranium from ore, and recovering the uranium from leach liquor. The leaching step produces a solid waste called tailings that is usually impounded on mill property. Mill tailings consist of uranium-depleted ore, processing chemicals, and process wash water. Practically all uranium daughter radionuclides in the ore appear in the tailings.

Uranium mills have been operating at various locations in the western United States for the past three decades. The average operating lifetime of a uranium mill has been about 12 to 15 years. Consequently, 24 of the older mills have been closed down. These inactive mill sites contain about 20% of all tailings impounded in the United States.⁽³⁾ Frequently mill properties were vacated with little if any attention given to preparation of tailings for long-term storage or disposal.

Tailings are a source of gamma radiation, radon and other airborne contamination, and water contamination. Furthermore, tailings material has been removed by individuals for private use. An uncontrolled inactive tailings pile may be regarded as:

1. A large, highly dispersed, uncontained source of radioactivity
2. An unattended source with minimal restraints to public access

3. A source which will yield continuous public exposure to radiation for several millenia
4. A source of waste-process chemicals and toxic elements continued in the discarded ore matrix.

Since tailings piles represent a public nuisance and potential radiological hazard, one of two approaches could be taken to reduce or minimize impacts of piles: in situ stabilization or removal to a site better suited for long-term management.

The inactive uranium mill in Salt Lake City, Utah, was formerly operated by the Vitro Corporation. At this mill site is a tailings pile that contains an inventory of ~ 1400 Ci of ^{226}Ra and its daughters spread over an area of $\sim 5 \times 10^5 \text{ m}^2$ (110 acres). The metropolitan area of Salt Lake City and its 400,000 inhabitants are within 11 km (7 mi) of the site.⁽⁴⁾

This report presents the results of field and laboratory analyses performed in support of a radiological assessment of this inactive mill site. Physical measurements included: (1) the depth of radionuclide contamination of soil underlying the tailings, (2) the concentration of radionuclides in groundwater, (3) the degree of spread of tailings due to wind and water erosion, (4) radon progeny concentrations in structures near the pile, and (5) external radiation levels on the pile, in structures near the pile, and at points of public access along the site perimeter. These results are interpreted to estimate present and extrapolated maximum public radiation exposures.

It is shown in this report that the radon diffusion pathway for exposure to man predominates at all points outside the perimeter of the mill site. If the nature of property development were to change from

commercial to high density residential at some time in the future, cumulative population exposures would be dependent on the degree of remedial action taken. Remedial action recommendations will be based on the results of this work combined with an engineering study of inactive uranium mills being conducted by the firm of Ford, Bacon and Davis, Utah, of Salt Lake City, Utah.⁽⁴⁾

2. BACKGROUND AND OBJECTIVES

The former Vitro mill was constructed during World War II to produce alumina. Uranium milling operations began at this site in 1951 and continued for 14 years. The mill was then converted to process ferrophos for vanadium recovery. Uranium milling was carried out in Salt Lake City because the abandoned alumina mill was available and the site was convenient to rail transportation.

Two leaching processes were used during the 14 years of uranium milling. The initial operation, from 1951 to 1957, included an acid leach and uranous phosphate precipitation. In 1957, a 20% increase in production was obtained by a plant expansion and process modification in which a solvent extraction process was substituted for the precipitation step. Then, prior to shutdown in 1968, the vanadium extraction work was carried out for about four years.

Wastes from these processes consisted of the remains of the fine ore particles (tailings), various chemical compounds used during the total separation process, and process water used for washing and for slurry transfer. Tailings represent a significant long-term management problem because this material is nearly equal to the feed ore in quantity and contains most of the uranium daughters which were not extracted with the uranium. Radioactivity contained in the tailings is found in both the sand and slime fractions. The slimes are made up of particles equal to or less than 200 mesh and make up approximately 20% of the total volume. However, these fine particles contain 80% or more of the natural

radioactivity. The concentration of ^{226}Ra in slimes resulting from the processing of western ores may reach 3000 pCi/g, whereas radioactivity in the sands is generally lower by a factor of 10.

Vanadium tailings cover parts of the waste area. These vanadium tailings, which are low in radium content, serve to retard radon emanation from the underlying uranium tailings. However, this area is not extensive and if the recommended remedial action for this site should be tailings removal, then these vanadium tailings would also require moving. A detailed radiation survey could be used to define the specific nature of the vanadium tailings area.

A sewage treatment plant on property adjacent to the Vitro mill began operation about the same time as the milling operations. Since 1967, sewage sludge from this plant has been pumped routinely onto several sections of the tailings pile to help reduce the quantity of wind blown material.

Figure 1 is an aerial photo of the site in its present condition. The overall site is approximately square. About three-fourths of the site area is covered by tailings; the original mill and associated structures were located on the remainder. The property is bounded on two sides by heavily traveled streets and highways. Intact fencing exists along 33rd South Street and along 900 West Street. However, no fencing exists along the access road to the Salt Lake City Suburban Sewage Treatment Plant, nor on the north and east boundaries of the Denver and Rio Grande Western Railroad property—see dashed lines in

ORNL Photo 0238-76

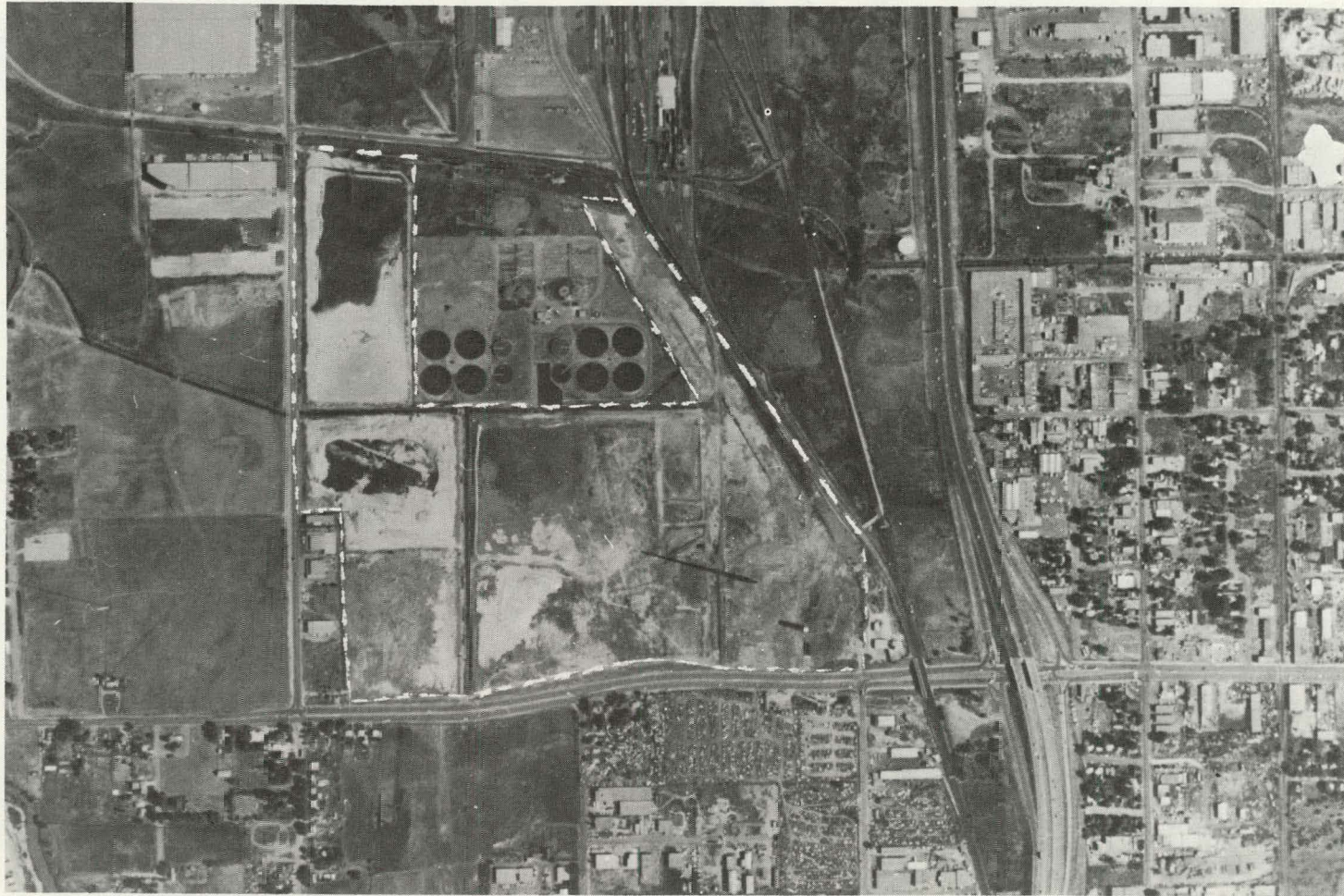


Fig. 1. Aerial view of Vitro mill site, tailings pile, and surrounding area.

Fig. 1. Therefore, no effective barrier exists to prevent access to the tailings.

The impounded areas grew steadily throughout the years of operation until an area of $4.7 \times 10^5 \text{ m}^2$ (~110 acres) was covered with tailings. Although some plant life exists along most of the tailings dikes, no effort toward stabilization has been attempted for the entire site. Therefore, erosion of tailings by wind and rain is not adequately inhibited. One area of about 20 acres was treated experimentally with a chemical stabilizer by the U. S. Bureau of Mines.⁽⁵⁾ Some vegetation grows in that location, but this does not eliminate dust blowing during high winds.

In addition to the tailings, a considerable amount of rubble from the dismantled plant is strewn over large areas. Much of this rubble appears to be contaminated with uranium bearing material.

The significance of this abandoned operation on the local area is not limited to radiological considerations. The mere presence of a restricted site within an otherwise busy commercial and industrial area creates an economic burden on the area. Because the tailings are radioactive, an indefinite ban has been created on construction within approximately a 0.5 mile radius of the site. Presently, there is some discussion about a relaxation on this ban, but there is no clear and well-defined policy on this matter. At present, the trend in property development is commercial; however, no one can predict the situation for this area more than thirty years⁽⁴⁾ in the future. Therefore, some degree

of restriction must be placed on the property as long as it remains in its present condition. This calls for maintenance of access control measures for the foreseeable future.

Scope of Work

A detailed characterization of the radiological hazards of the Vitro site, as viewed from the standpoint of health effects, is the specific objective of this project. In order to complete such a characterization, it is necessary to do the following:

1. Determine baseline background data on concentration of pertinent radionuclides, such as ^{226}Ra , ^{238}U , ^{232}Th , ^{230}Th , and ^{210}Pb . Numerous samples were collected throughout the Salt Lake Valley.
2. Assess the radiological impact of the tailings pile and associated mill area in its present condition. This entails a detailed study of the radiological variables pertinent to five major radiation dose-to-man pathways.
3. Assess the radiological impact of possible remedial actions. Specific actions which may be taken to reduce, minimize, or eliminate the radiological hazard to the public have been presented in detail elsewhere.⁽⁴⁾ These actions include, but are not limited to, erecting strong antipersonnel enclosures, stabilizing the tailings with various thicknesses of earth cover, or removal of tailings to a less populated area. The latter action would return the Vitro site to productive use and remove restrictions against its projected long-term use even for high density residential use.

In support of 2 above, it was necessary to conduct a series of radiological measurements. The objective of these measurements was to determine the magnitude of radiation levels both on and off site. Of principal concern were ground level air concentrations of radon and progeny in residential and commercial areas and the extent to which tailings had spread to areas surrounding the mill site. These radiation levels were evaluated by making measurements of the radionuclide concentration in samples of soil underlying the tailings, on property adjacent to the mill site, and in samples of surface stream sediments. In addition, water samples were obtained from several private wells, from surface streams, and from each hole which was drilled or dug on the site. Concentrations of radon (pCi/liter) were measured for continuous 24-hr periods both indoors and outdoors at many locations throughout the Salt Lake Valley by Ford, Bacon and Davis, Utah.⁽⁴⁾ Some corresponding measurements were made of the radon progeny concentrations within structures and in outdoor areas. Also, measurements were made of external gamma-ray exposure rates on the pile at locations around the perimeter of the site and at points of interest in the immediate surrounding area.

Radiological measurements made in this study are compared to and supplemented by the results of previous studies in the vicinity of the Vitro mill. These previous studies have yielded data on windblown contamination as well as data on background concentrations of nuclides in soil and air throughout a large area of the Salt Lake Valley and other parts of the state.

3. BEHAVIOR OF TAILINGS PILE RADIONUCLIDES

Milling processes remove about 90% of the uranium but only a few percent at most of other radionuclides in the original ore. Daughters of ^{238}U contribute about 95% of tailings activity; the daughters of ^{235}U , about 5%. Differences in geochemical properties of U and Th suggest that ^{232}Th will not be concentrated in postmagmatic minerals containing uranium.⁽⁶⁾ Ores commonly mined in the United States contain only thorium isotopes attributable to uranium decay.⁽²⁾ Tailings which resulted from processing ores older than a few hundred thousand years may contain activities of chain members approaching equilibrium with the original parent uranium activity. A listing of the major uranium and actinium chain members is shown in Table 1.

A recent study by Healy⁽⁷⁾ of plutonium soil contamination serves as a good outline of pathways leading to human exposure. A modification of Healy's approach has been applied to tailings pile radionuclides in Table 2. Each of the pathways listed in Table 2 should be evaluated in any assessment of the overall radiological impact of each specific mill site. The factors which influence the pathway processes shown in Table 2 will be evaluated in this chapter.

Radon Diffusion Pathway

Several generic radiological assessments have been made for inactive uranium mill sites.^(1,2,8,9) These generic assessments have usually shown that ^{222}Rn diffusion constitutes the most significant pathway because of inhalation of radon progeny. A wide range of estimated ^{222}Rn

Table 1. Principal tailings pile radionuclides

Nuclide ^a	Half-life	Primary decay modes		
		α	β	γ
Uranium series				
²³⁸ U	4.5 x 10 ⁹ y	x		
²³⁴ Th	24 d		x	x
^{234m} Pa	1 m		x	x
²³⁴ U	2.5 x 10 ⁵ y	x		
²³⁰ Th	8 x 10 ⁴ y	x		
²²⁶ Ra	1.6 x 10 ³ y	x		x
²²² Rn	3.8 d	x		
²¹⁸ Po	3.1 m	x		
²¹⁴ Pb	26.8 m		x	x
²¹⁴ Bi	19.7 m		x	x
²¹⁴ Po	160 μ s	x		
²¹⁰ Pb	21 y		x	x
²¹⁰ Bi	5 d		x	
²¹⁰ Po	138 d	x		
²⁰⁶ Pb	Stable			
Actinium series				
²³⁵ U	7 x 10 ⁸ y	x		x
²³¹ Th	26 h		x	x
²³¹ Pa	3.3 x 10 ⁴ y	x		x
²²⁷ Ac	21.6 y		x	
²²⁷ Th	18 d	x		x
²²³ Ra	11.4 d	x		x
²¹⁹ Tl	4 s	x		x
²¹⁵ Po	1.8 ms	x		
²¹¹ Pb	36 m		x	x
²¹¹ Bi	2.2 m	x		x
²⁰⁷ Tl	4.8 m		x	
²⁰⁷ Pb	Stable			

^aMinor branches and minor decay modes are not listed.

Table 2. Principal radionuclides involved in pathways from tailings piles to man

Pathway	Radionuclides	Pathway processes	Exposure hazard
Radon diffusion	^{222}Rn	Diffusion to pile surface Atmospheric transport Inhalation	Inhalation of ^{218}Po , ^{214}Pb , ^{214}Bi , ^{214}Po
Airborne activity	^{230}Th , ^{231}Pa	Resuspension from pile surface Atmospheric transport Inhalation Body surface contamination	Inhalation of airborne activity carried by respirable particles
Terrestrial	^{226}Ra , ^{210}Pb	Resuspension from pile surface Atmospheric transport Deposition on soil or foliage Uptake by plants Intake by animals Consumption of contaminated plants, meat, and milk Consumption of contaminated soil	Ingestion of ^{226}Ra , ^{210}Pb
Aquatic	^{226}Ra , ^{210}Pb	Dissolution of radionuclide in pile Migration through soil to water Migration in water a. Uptake by aquatic organisms b. Consumption of aquatic organism a. Use of water for irrigation h. Uptake by plants c. Consumption of plants Consumption of contaminated drinking water Immersion in contaminated water	Ingestion of ^{226}Ra ; whole body gamma irradiation
External gamma	Daughters of ^{226}Ra	Attenuation by pile material Air-distance attenuation External exposure	External exposure to gamma radiation produced by the decay of ^{226}Ra and its daughters

effects has been reported. Klement et al.⁽⁸⁾ concluded that no significant risk was imposed upon the public by active and inactive milling sites. They stated that population doses from these sources cannot be distinguished from background. On the other extreme, there are those who have extrapolated Environmental Protection Agency (EPA) assumptions⁽²⁾ and have concluded that ^{222}Rn released by uranium mill tailings will represent a fatal encounter for millions who will be exposed over a period of 80,000 years. The analysis offered by Sears et al.⁽⁹⁾ required many assumptions and was not intended to be applied to any specific site; however, their results indicate that ^{222}Rn progeny may account for > 90% of the dose to individuals residing in areas adjacent to a tailings pile.

The first step in the radon diffusion pathway is the migration of this inert gas to the surface of the tailings pile. Migration consists of two stages: diffusion of ^{222}Rn from its point of origination into the intergranular capillary network, and migration through the capillaries to the pile surface.

Only those ^{222}Rn atoms formed by the decay of ^{226}Ra at or very near the surface of a particle are likely to escape from the particle. The fraction of radon which escapes from a particle is called the "emanation factor" or "emanating power." Culot et al.⁽¹⁰⁾ gave an average value for this factor of about 23% for uranium mill tailings. This is in good agreement with the experimental results obtained for various soils by Sisigina⁽¹¹⁾ (14 to 29%) and by Megumi and Mamuro⁽¹²⁾ (25%).

Radon released to the interstitial pores migrates toward the surface of the pile. It behaves in this manner due to molecular diffusion whereby ^{222}Rn moves from a volume of air at a high concentration (pore air) to a volume whose concentration is lower (atmosphere). The movement of the pore air itself may either enhance or retard the diffusion process. The instantaneous flux of radon across a unit area of the pile surface may be influenced by factors affecting the movement of pore air, for example, atmospheric pressure,⁽¹³⁾ temperature differential between atmosphere and tailings,⁽¹⁴⁾ wind speed,⁽¹⁵⁾ atmospheric stability^(15,16) and presence or absence of freezing conditions.⁽¹⁵⁾ The movement of pore air has little effect on a long-term average flux because of the cyclical nature of the short-term variations.

The application of Fickian diffusion equations to describe the migration of ^{222}Rn through tailings material has been developed by Culot et al.⁽¹⁰⁾ who considered a layer of tailings material sandwiched between a concrete cover and an earthen base. A similar approach can be used to derive an equation describing the ^{222}Rn flux diffusing from a bare tailings pile:

$$J = \left[\frac{K_1 q_1 R E \rho}{p_1} \right] \times \left[\frac{K_2 q_2 \cosh q_1 t + K_1 q_1 \sinh q_1 t - K_2 q_2}{K_1 q_1 \cosh q_1 t + K_2 q_2 \sinh q_1 t} \right] \times 10^4, \quad (1)$$

where

$$q_i = \left[\frac{\lambda p_i}{K_i} \right]^{1/2} \quad (2)$$

and J is the ^{222}Rn flux from the pile surface ($\text{pCi}/\text{m}^2\text{-sec}$),
 K is the bulk diffusion coefficient (cm^2/sec),
 λ is the decay constant of ^{222}Rn ($2.10 \times 10^{-6} \text{ sec}^{-1}$),
 p is the porosity of the medium (fraction),
 E is the emanation factor,
 R is the ^{226}Ra content of the tailings (pCi/g),
 ρ is the bulk density of the tailings (g/cm^3),
 t is the depth of the tailings pile (cm),
subscript 1 refers to tailings,
subscript 2 refers to soil below tailings.

The inverse of q is called the "relaxation length"—the thickness of source-free material required to reduce the radon flux to 37% ($1/e$) of its original value. The diffusion coefficient, K , represents the effect of the properties of the medium on the diffusion process.

Baver⁽¹⁷⁾ presents evidence which indicates that the diffusion coefficient of a porous medium varies inversely with the tortuosity of the medium. The porosity, p , which represents the ratio of the cross-sectional area of pores compared to the total cross section of the bulk medium, directly affects the value of the diffusion coefficient.^(10,15,18,19) The pore diffusion coefficient, D , is related to the bulk diffusion coefficient by

$$D = Kp. \quad (3)$$

The effect of moisture in the medium is shown in Fig. 2 where Tanner's⁽¹⁸⁾ data is used to solve Eq. (1). These data are in general agreement with the trends noticed by Guedalia et al.⁽¹⁴⁾

In the application of Eq. (1), it is assumed that the porosity of the medium does not vary with depth; for example, there is no compaction of the medium at lower depths. Furthermore, the properties of the medium represented in Eq. (1) are assumed to be isotropic.

The average ^{222}Rn flux attributable to diffusion can be estimated by applying Eq. (1) to data obtained from onsite physical measurements. The atmospheric dispersion of the ^{222}Rn from the tailings pile may then be used to estimate the annual average human exposure attributable to this source. Atmospheric dispersion may be calculated by representing the tailings pile as an area or strip source^(20,21) and using local meteorological data in a Gaussian diffusion model.⁽²²⁾

The results of short-term (~ 1 week) atmospheric sampling of ^{222}Rn on and near tailings piles have been reported by Snelling^(23,24) and Snelling and Shearer.⁽²⁵⁾ However, the results reported by Shearer and Sill⁽²⁶⁾ indicate that using the results of a short-term sampling program to represent the annual average can lead to serious errors. Their⁽²⁶⁾ 57 sampling stations were operated for 48-hr periods at three-week intervals for a one-year period or about 10% of the time. The results reported for each sampling station showed ranges of about an order of magnitude because the short-term ^{222}Rn emanation rate from the pile is affected by pore air movements and atmospheric dilution, and dispersion of the emanated radon is affected by the meteorological

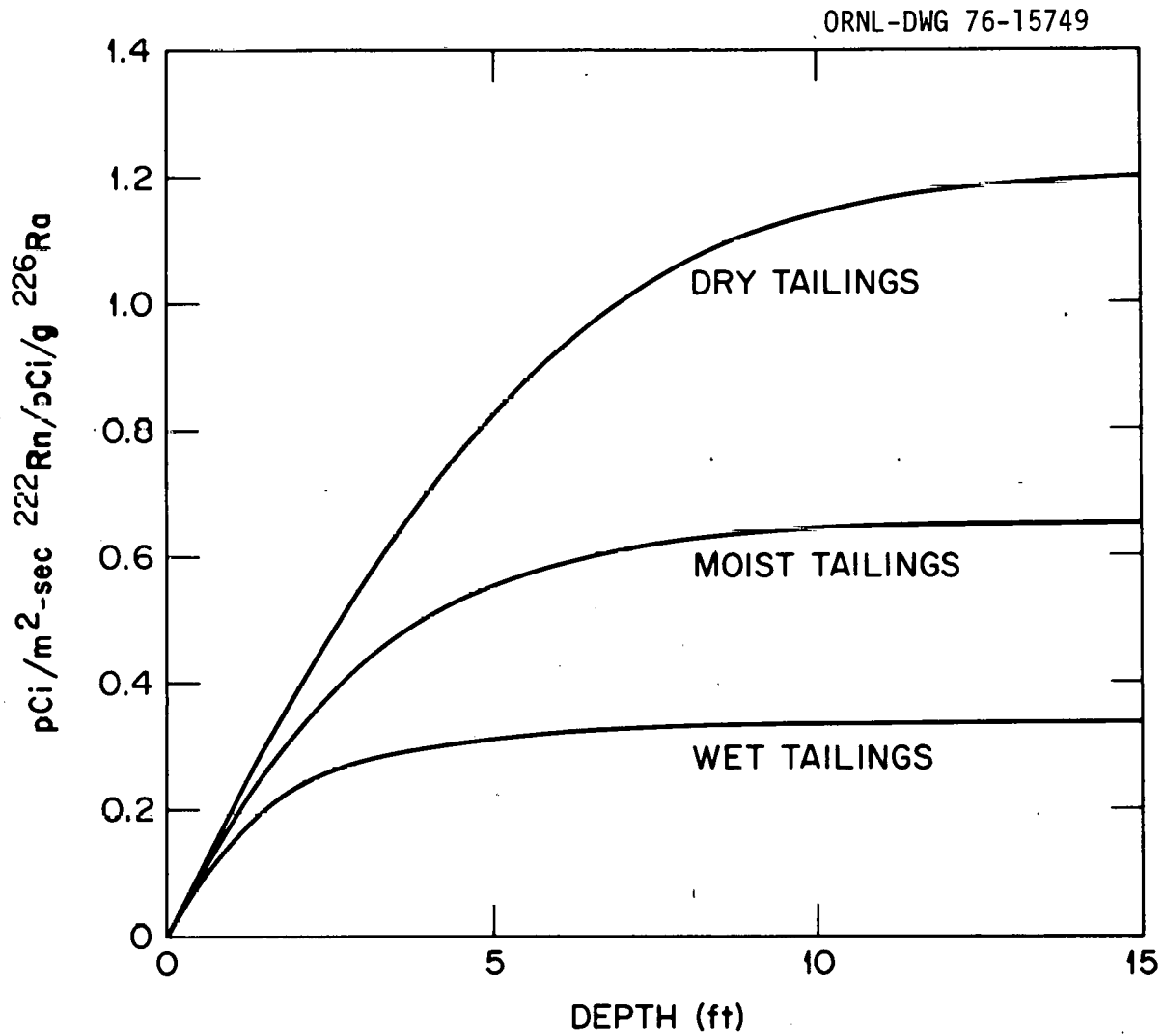


Fig. 2. Graphic representation of radon emanation from wet, moist, and dry tailings as a function of ^{226}Ra concentration and depth of tailings.

conditions prevailing at the time of sampling. Thus, extrapolation of short-term measurements to represent annual average human exposures can lead to serious errors.

Human exposure attributable to ^{222}Rn emitted by the tailings pile may be estimated conservatively by making the following assumptions:

1. During transport from pile to receptor, no ^{222}Rn is lost by decay, dry deposition, or rainout.
2. The outdoor airborne radon concentration at a height of 3 m is the same as the ground level concentration calculated by the dispersion model.
3. The fraction of the airborne radon concentration within a habitable structure due to the tailings pile is the same as the outdoor radon concentration attributable to the tailings pile.
4. The individual resides continuously in the structure.

The conversion of an indoor ^{222}Rn concentration to a dose equivalent is dependent on an extensive knowledge of exposure environment and lung dosimetry conditions. Presently, an adequate definition of these conditions is not available for non-mining situations.⁽²⁷⁾ Epidemiological studies of lung cancer incidence in uranium miners have used exposures estimated in terms of working-level months* (WLM).^(28,29) The conversion of the WLM unit to rems can be done only with major uncertainties.⁽³⁰⁾

* One working level = 1 WL = any combination of short-lived radon daughters in 1 liter of air that has a potential alpha energy of 1.3×10^5 MeV; 1 WLM = exposure at a level of 1 WL for a duration of 170 hr.

One can, however, develop a rough correlation between the two measurable factors: cumulative exposure in WLM and number of lung cancers. Even these two factors are difficult to quantify in real life situations. Consequently, radon progeny exposure calculations made in this document will be expressed in terms of WLM, and results of radon progeny measurements will be expressed in terms of WL.

Airborne Activity Pathway

The suspension of tailings and subsequent transport by wind creates another potential pathway for human exposure. Wind may cause transport by one of three modes: surface creep, saltation, and suspension. Surface creep is illustrated through the movement of large particles moving along the surface due to wind or by the impaction of other particles. Intermediate sized particles move by saltation, wherein the particles are lifted a few feet in the air by wind and quickly return to the surface. The smaller particles, including those in the respirable range, become suspended and remain suspended due to vertical turbulence. The theoretical and experimental studies of these mechanisms are well summarized by Healy⁽⁷⁾ and Anspaugh et al.⁽³¹⁾

The quantity of radioactive material suspended is affected by variables such as the topography and physical properties of the surface, the amount and type of vegetative cover, the extent of surface contamination, local micrometeorology, and the time elapsed since the surface was contaminated. Previous resuspension studies have been concerned with agricultural erosion and resuspension of plutonium contamination. No attempts have been made to determine resuspension rates of tailings,

although an unverified, theoretical approach has been attempted for instantaneous circumstances.⁽³²⁾

Most of the previous surveys of airborne activity near uranium mill tailings piles have been of short-term duration.^(23-25, 33) The work of Breslin and Glauberman^(33, 34) compared the airborne activities measured near three abandoned tailings piles. They reported that airborne activity concentrations were "trivial" near two of the piles. The third pile was considerably drier than the other two, and "dust devils" were prevalent during sampling. Consequently, the airborne concentrations exceeded maximum permissible concentrations* (MPC's) only at sampling stations downwind of and in close proximity to this pile. Comparable results were obtained in the other short-term surveys.⁽²⁴⁻²⁶⁾

The results obtained by Breslin and Glauberman^(33,34) indicate that about one-third of the total suspended activity was associated with respirable particles. Furthermore, about 10% of the total airborne alpha activity was determined to be water soluble.

A long-term survey was conducted from January, 1974, thru December, 1974, consisting of 26 separate samples of airborne activity at a station on the Salt Lake City, Utah, tailings pile.⁽³⁵⁾ None of the reported individual samples had activities greater than MPC_a (10 CFR 20)⁽³⁶⁾ for ^{226}Ra , ^{210}Pb , ^{210}Po , ^{230}Th , ^{234}U , ^{235}U , or ^{238}U .

*Breslin and Glauberman^(33,34) compared their measured values with those recommended in National Bureau of Standards Handbook 69.

In addition to inhalation of respirable airborne activity, the contamination of body surfaces and clothing by airborne particles should be considered. This has been done by Healy⁽⁷⁾ for plutonium contamination. He concluded that inhalation of airborne material imparts a much greater hazard.

Deposition of particles between the source and the receptor will reduce the airborne activity pathway hazard in relation to that associated with the radon diffusion pathway. Thus, it appears that the contribution of the airborne activity pathway to total exposure is only a small fraction of that attributable to radon diffusion; hence it is ignored in this assessment study.

Terrestrial Pathway

The terrestrial pathway is a means by which the deposited airborne activity may reach man. Radionuclides may be deposited on edible foliar surfaces, migrate through the soil into edible crops, or be consumed by animals which provide meat or milk. The radionuclides found in uranium tailings have concentration factors in crops, meat, and milk which are generally much less than unity.⁽⁶⁾ The worst case would likely be the direct ingestion of contaminated soil. If it is assumed that a total of 1 g of the top 1 cm of contaminated soil is ingested daily,⁽⁷⁾ the upper limit of 2 pCi/day⁽³⁷⁾ would restrict the soil activity to 2 pCi/g of ^{226}Ra . This would result in an areal activity of 3.2×10^4 pCi/m² for a soil density of 1.6 g/cm³. The observed annual average airborne concentration of ^{226}Ra measured over the Salt Lake City pile was 0.002 pCi/m³.⁽³⁵⁾ According to data presented by Healy,⁽⁷⁾ deposition velocities

are on the order of 0.5 cm/sec (1.6×10^5 m/year) for a semi-arid topography. The deposition of contamination, assuming that a concentration of 0.002 pCi/m³ remains undiluted and is deposited on contamination-free land, will be 2×10^{-3} pCi/m³ \times 1.6×10^5 m/year or 3.2×10^2 pCi/m² year. Assuming that there are no erosional losses of activity, the time required for the activity of the top 1 cm of soil to approach 2 pCi/g is on the order of 100 years. Because the radionuclides are fixed in fine-grained sands, these values hold for most other uranium chain daughters.

A study of foodstuffs in the Durango and Grand Junction, Colorado, areas⁽³⁸⁾ concluded that daily dietary intakes of ²²⁶Ra in these areas were similar to intakes reported by other U.S. cities which have no uranium tailings nearby.

Thus, based on present information, the terrestrial pathway appears to be inconsequential as a source of human exposure.

Aquatic Pathway

Aquatic pathways are highly site specific. The proximity of the site to ground-or surface-water supplies, the presence of leaky aquifers, and the values of local geohydrologic parameters will determine the magnitude of the hazard associated with this pathway. Surface-water erosion would result in runoff water contaminated primarily by particulate activity scoured from the tailings pile surface.

No evidence of water contamination by ²²⁶Ra existed in the surveys performed by Snelling^(23,24) and Snelling and Shearer.⁽²⁵⁾ However, concentrations of uranium three orders of magnitude above background were found in groundwater sources near Mexican Hat, Utah, tailings

pile.⁽²³⁾ Gera⁽⁶⁾ attributes the mobility of uranium in groundwater to the presence of oxidizing conditions. The uranyl ion (UO_2^{++}) is chemically stable, highly soluble, and readily forms anionic complexes. These properties which enhance the mobility of uranium are not exhibited to an appreciable extent by the other nuclides in tailings.

The leachability of ^{226}Ra from tailings and river sediments was studied by Shearer and Lee⁽³⁹⁾ who found that only the liquid to solid ratio was of any importance. On the other hand, Havlik et al.^(40,41) found that the chemical composition of the leaching solution, particularly the presence of the Cl^- ion, had the greatest significance. A comparison of the results of these leachability studies indicates that the mineralogy of the adsorbent may also influence the solution behavior of ^{226}Ra . Gera⁽⁶⁾ presents data which indicate that the distribution coefficient, K_d , for radium in some Russian mineral springs ranges from 87 to well over 10^6 ml/g. Measured values of K_d for carbonate leached tailings have ranged between 1000 and 2000 ml/g for ^{226}Ra .⁽⁴²⁾ Typical values of K_d for Ra with a sandy soil adsorbent are reported as about 100, whereas K_d for U, Th, and Pb is in excess of 1000.⁽⁹⁾

The velocity for a groundwater contaminant is given by the expression⁽⁴³⁾

$$V_c = V_w \left[\frac{1}{1 + \frac{\rho K_d}{p}} \right], \quad (4)$$

where V_c is the velocity of the contaminant (m/year),

V_w is the groundwater velocity (m/year),

K_d is the distribution coefficient (ml/g),

ρ is the bulk density of the medium (g/cc),

p is the porosity of the medium.

For a system with a K_d of 100, ρ of 1.6, and p of 0.4, the contaminant velocity will be $2.5 \times 10^{-3} \times V_w$. This calculation indicates that the annual migration of ^{226}Ra for a water velocity of 300 m/year will be about 0.75 m. On the other hand, uranium migrating as anionic uranyl complex will migrate nearly as fast as the groundwater. Thus, uranium may have greater mobility in this pathway than radium.

The aquatic pathway may be of importance if a leaky aquifer containing potable water is in contact with the leached contamination. The adsorptive capacity of the subsoil will usually inhibit the movement of all tailings radionuclides except uranium migrating as an anionic uranyl complex. Thus, site specific parameters affect the importance of this pathway. The clayey nature of the soil underlying the Vitro tailings coupled with the fact that the site rests on an artesian groundwater system helps to minimize the downward movement of radionuclides through groundwater.

External Gamma Radiation

Gamma radiation coming directly from the tailings pile or from windblown material is expected to make a minor contribution to the overall exposure of man. However, there are some areas where increased levels of gamma radiation have been observed. Included are areas along public roads near the Vitro mill property, in commercial buildings built

over tailings, and in or near structures where tailings have been removed from the pile itself. It has been shown in reports of investigations at other mill sites⁽²³⁻²⁵⁾ that external gamma radiation levels decrease rapidly with distance from the pile. This observed decrease with distance does not follow the typical inverse square relationship because of the large physical size of the pile.

Exposure of individuals on or near the pile is due to both "primary beam" or unscattered photons from nearby tailings material and gamma rays which originate at other more distant points and reach the individual through air scattering processes. In cases where structures are built over tailings, practically all gamma rays are attenuated to some degree by the floor and other building material and, because of the relatively small total source area, represent a hazard only in the immediate vicinity (i.e., it is unlikely that tailings underlying one building would create an exposure hazard at an adjacent site).

4. RADIOLOGICAL MEASUREMENTS AND TECHNIQUES

Radionuclide Concentration as a Function of Depth in Soil

A monitoring and sampling procedure was established for this project in conjunction with the engineering firm of Ford, Bacon and Davis, Utah. A set of 6-in. holes was drilled through the tailings and into the subsoil. A length of PVC pipe (3-in. O.D.), sealed on one end, was lowered into the hole and measurements were made of gamma-ray intensities as a function of depth. A 6-in.-long Geiger-Muller tube shielded with lead containing collimating slits was positioned inside the PVC tube for measurements. Signals from this tube were counted using a portable scaler.⁽⁴⁴⁾

After gamma ray vs depth profiles were determined, the position of the interface between tailings and subsoil was estimated. The drilling rig was moved approximately 4 ft and another hole was drilled with an auger bit to the interface level. Samples of soil core were then collected as a function of depth using a split-spoon sampler (each core section was 2 ft long). Most of the holes in the tailings area contained water due to the shallow groundwater table. This water made it difficult to obtain uniform samples, introduced some uncertainty in the location of samples, and contributed to contamination of the subsoil by washing fine particles from the side wall of the tailings hole into the water and, subsequently, into the bottom of the hole.

Most of the penetrating gamma radiation monitored in these holes was attributable to radium and its daughters. Therefore, a calibration

factor for ^{226}Ra concentration was determined for the collimated gamma-ray probe by comparing the response of this unit (counts per unit time) with an estimate of the radium concentration (pCi/g) determined by counting soil samples using gamma spectroscopy techniques. A least squares fit of the data from this comparison yields the equation

$$R = 0.29C - 20. \quad (5)$$

For this case, R is the ^{226}Ra activity in pCi/g and C is the observed response of the collimated gamma-ray detector in counts/min. The correlation coefficient for this equation is 0.996.

The above expression was useful in estimating the overall distribution of radioactivity in the tailings as well as the total quantity of radium in the impounded tailings area. On the basis of the quantity and grade of ore processed in the plant, the total radium activity in the tailings pile is 1380 curies.⁽³⁾ The total ^{226}Ra activity based on the field survey of total tailings mass and overall distribution as determined by the gamma logging technique was 1400 curies.

Soil samples taken from the split-spoon core sections were prepared using facilities at the U. S. Bureau of Mines Laboratory in Salt Lake City. Each sample was dried for 24 hr at 110°C in order to remove all moisture. The samples were then pulverized in a high speed rotary crusher and sieved for a particle size $\leq 500 \mu\text{m}$. The soil was dispensed into 30-ml polyethylene vials normally used for liquid scintillation counting.

Each vial was weighed, sealed tightly and labeled as follows:

Denotes Salt Lake City ———— SV-10-RS1 ———— Number of sample location
 Vitro site ———— 1-ft deep
 ———— Below interface (subsoil)
 ———— Radiological sample

DATE - September 26, 1975
 Date bottle sealed ————

Net mass of sample ———— 33.6 g

Unfortunately, the presence of water in most of the core holes prevented the collection of a uniform sample, and at many specific points the mass of sample was significantly less than optimum. The sample vials were sealed tightly and stored for a period sufficient to allow the equilibration of radon and its daughters. Radon-222, which has a radioactive half-life of 3.8 days, will reach the same activity as its long-lived parent, ^{226}Ra , in about 30 days. The short-lived progeny of ^{222}Rn will have reached equilibrium within the same time period. Determination of the activity of any of the daughters in the sample will reflect ^{226}Ra activity. After equilibration of radon daughters, up to 12 sample vials were inserted into a sample carousel or holder (Fig. 3) that was placed on a Ge(Li) detector.

Water samples were collected at several points in a ditch used to drain outfall from the nearby sewage treatment plant. Water samples were also collected from Mill Creek, Jordan River, four wells off-site,

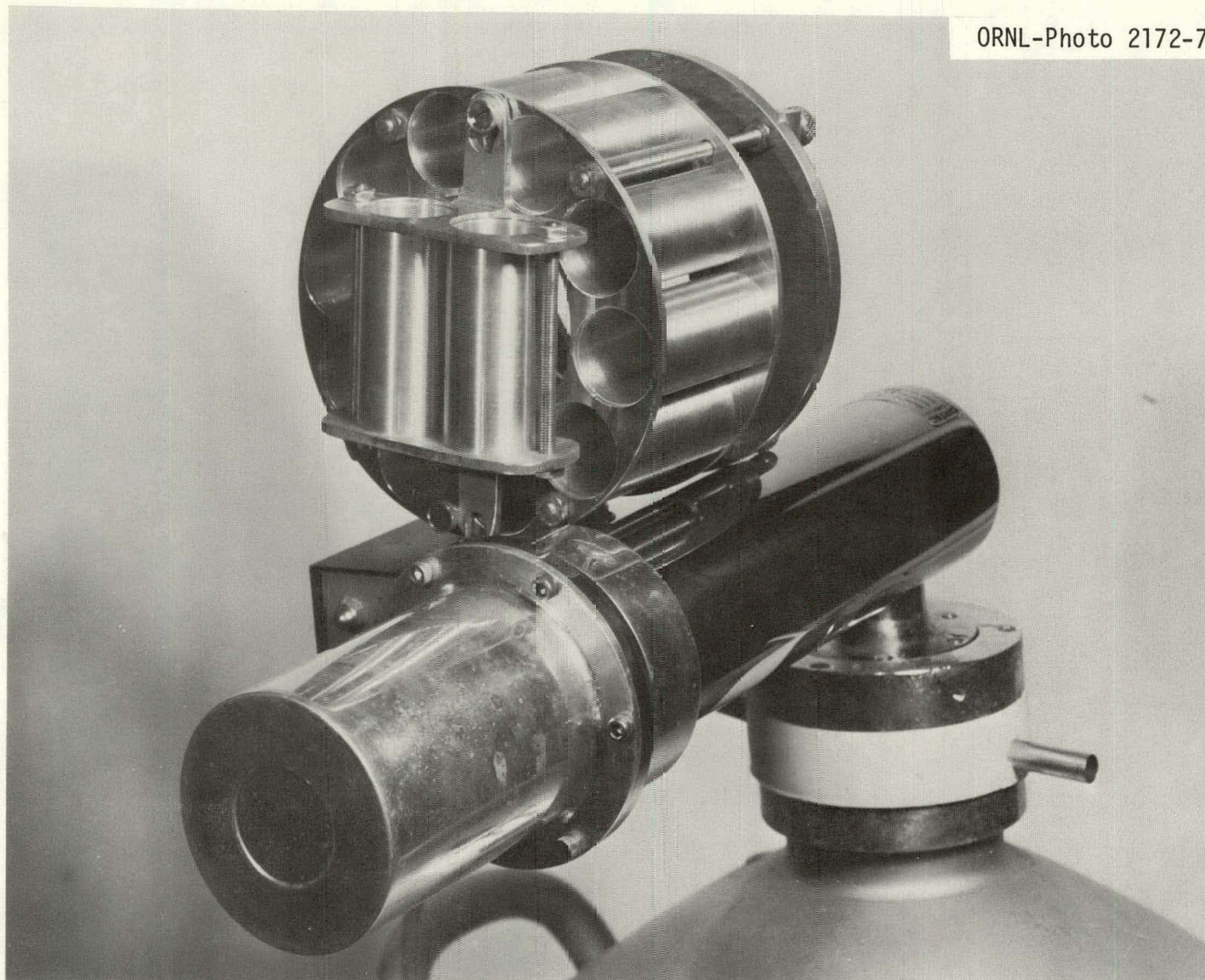


Fig. 3. View of horizontal Ge(Li) detector and sample "carousel" holder used for gamma-ray spectroscopy.

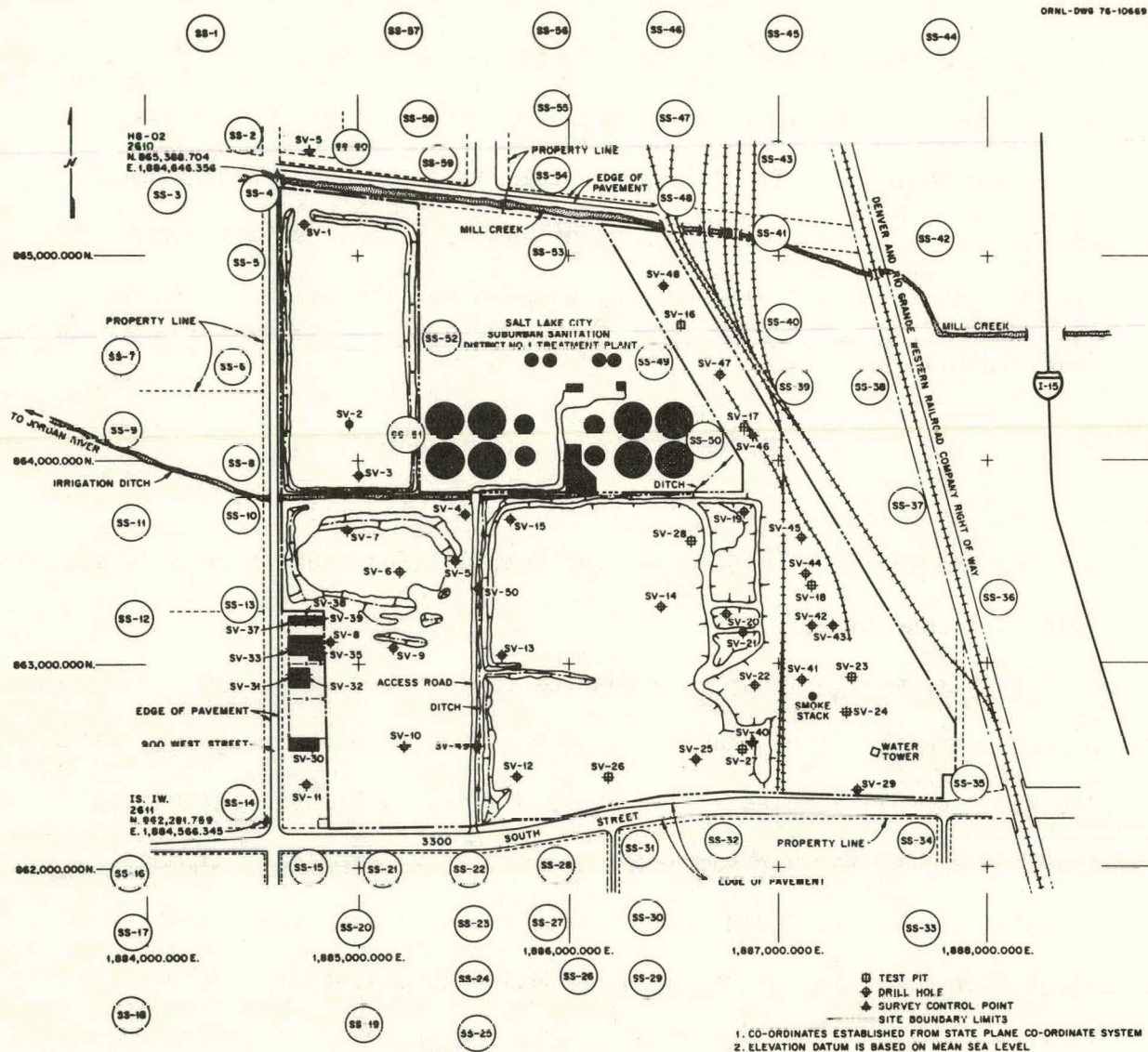
and each hole which had been bored through the tailings. These latter samples, taken evenly throughout the depth of water in each hole, were collected by the survey crew and stored for later analysis. All water samples were centrifuged and pressed through a 0.45- μ m filter to clear suspended solids. The samples were then analyzed using radiochemistry techniques. A plan view of the major sample locations is shown in Fig. 4. Water and stream sediment samples were collected at points shown on Fig. 5.

Field Laboratory Facilities and Equipment

Two facilities were established in Salt Lake City, at the General Services Administration's Administration Building and compound in southwest Salt Lake City.

A 20-ft mobile laboratory van was used to determine radon and radon progeny concentrations, gamma-ray intensities, and for weighing, packaging, and storing soil samples. This lab contained an alpha spectroscopy counting system for air samples along with air sampling equipment, a Johnston Laboratory radon monitor complete with six Lucas flasks and evacuation manifold, gamma-ray detectors, miscellaneous electronic testing equipment, and standard calibration sources. A trailer-mounted, gasoline-powered 12 kW motor generator was pulled by the van to supply electrical power in remote locations. A voltage stabilizer was used to provide regulated power for instruments.

The second field laboratory used in the project was an 8 x 35 ft air-conditioned semitrailer with running water, tools, miscellaneous supplies, two gamma counting systems, and computer. It served as a counting room, office, and workshop. This trailer required electrical



76-10669 Fig. 4. Plan view of the Vitro mill site showing core hole and offsite sample locations.

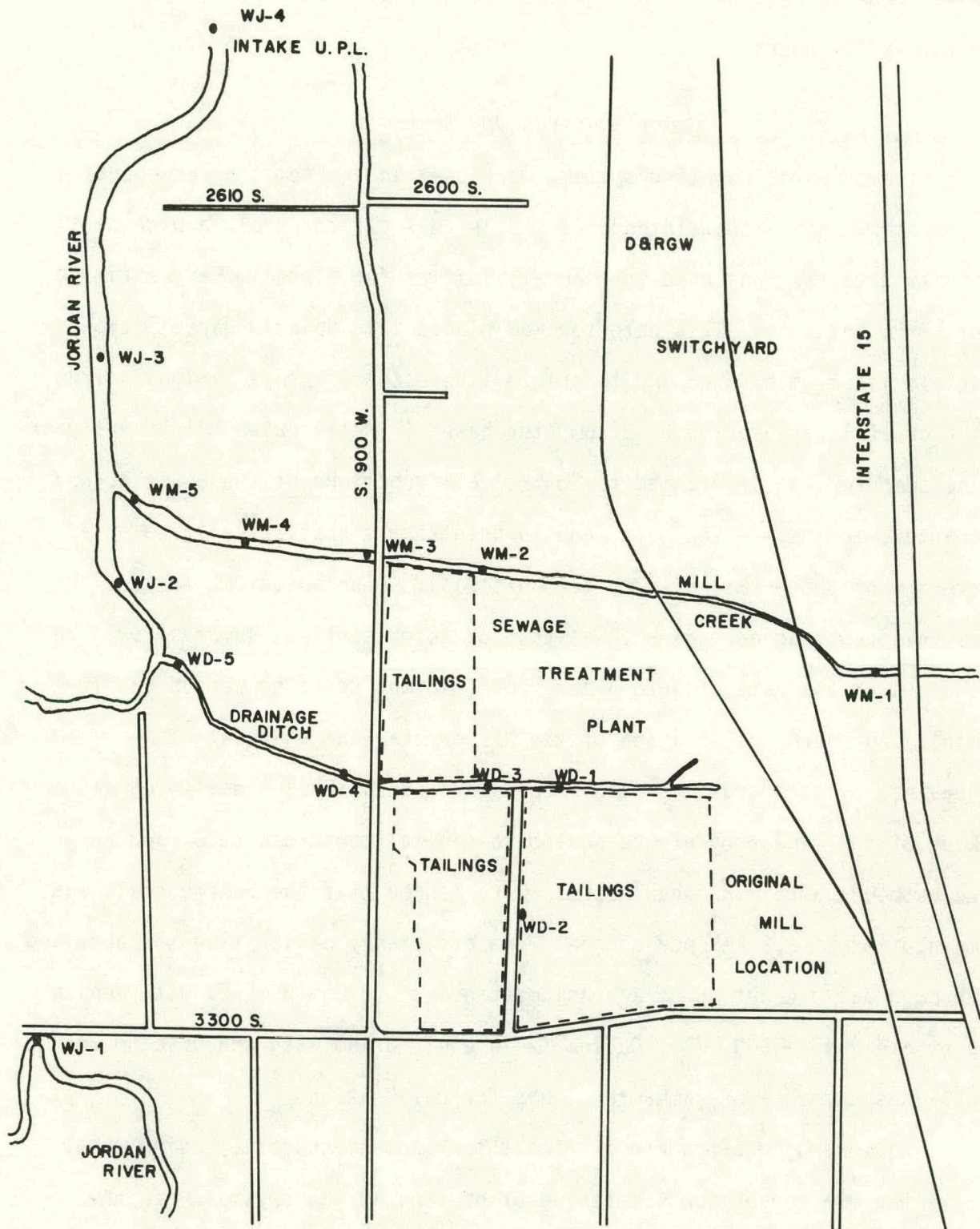


Fig. 5. Plan view of the Vitro mill site showing location of water samples collected in surface streams.

power from an external source, but could generally be placed in service within a few hours.

Gamma Spectroscopy Systems

Two complete counting systems were used in the semitrailer counting laboratory. A Harshaw integral 3 x 3 in. NaI (Tl) crystal, a high sensitivity detector, was used to scan all samples for a preliminary estimate of ^{226}Ra activity. This detector was placed in a "pickle barrel" type shield lined with Cu and Cd to shield fluorescence x rays. Signals from the crystal were sorted by a computer based (PDP-11) pulse height analyzer. The computer was programmed to control all functions of the analyzer and counter, to analyze the data, and to print out a statistically weighted average of the ^{226}Ra activity per unit mass. One convenient feature of this counting arrangement was that of quick sorting; samples could be scanned at the rate of 4 to 5 per hour (minimum counting period was 10 min). An energy calibration of the NaI crystal and analyzer was obtained by standardizing with ^{57}Co , ^{137}Cs , and ^{60}Co . A daily log was kept of the linearity of the analog-to-digital converter (ADC) and adjustments were made when necessary to assure that the energy scale was maintained at 3.8 keV per channel. An efficiency calibration was obtained through daily counting of a uranium standard* (0.05% U mixed with dunitc, particle size = 500 μm). Radium is in equilibrium with the uranium and provides a source of gamma-ray lines for calibration.

Low energy photons are difficult to resolve accurately using a NaI detector due to Compton scattering of photons in the crystal. In the interest of time, rapid screening was undertaken and preliminary ^{226}Ra

* Standard uranium sample obtained from the New Brunswick Laboratory.

concentrations were determined by observing signals from three high energy photons of ^{214}Bi , one of the short-lived daughters of radon. The energies of these photons are 609 keV, 1120 keV, and 1764 keV. The samples were stored as described above to allow equilibrium of ^{226}Ra with radon and daughters. In order to accomplish screening, a code was written for the PDP-11 computer which controlled data acquisition for the NaI system. The reduction of data was simplified by subtracting the Compton contribution from a region comprising the photopeaks of interest. The Compton contribution was determined by taking the average counts in three channels on each side of the photopeak region of interest; this value times the total number of channels comprising each photopeak region was subtracted from the total number of counts in the region. The net value was then compared to the instrument's response to the above-mentioned counting standard whose concentration of ^{226}Ra was known. The concentration of ^{226}Ra in each soil sample was determined by comparing the unknown soil concentration to the standard.

Final results of ^{226}Ra in soil were determined by counting all samples with a high resolution Ge(Li) detector. This high resolution counting system consisted of a horizontally mounted 50-cm³ Ge(Li) crystal positioned on a platform for movement into and out of a lead shield (see Fig. 6). The detector was used to obtain a complete photon spectrum of the soil samples. Signals from the crystal were routed to a 4096-channel pulse height analyzer and counted for a period long enough to evaluate the ^{226}Ra concentration to a statistical accuracy of $\pm 5\%$ or better. The spectra were recorded on magnetic tape and stored for

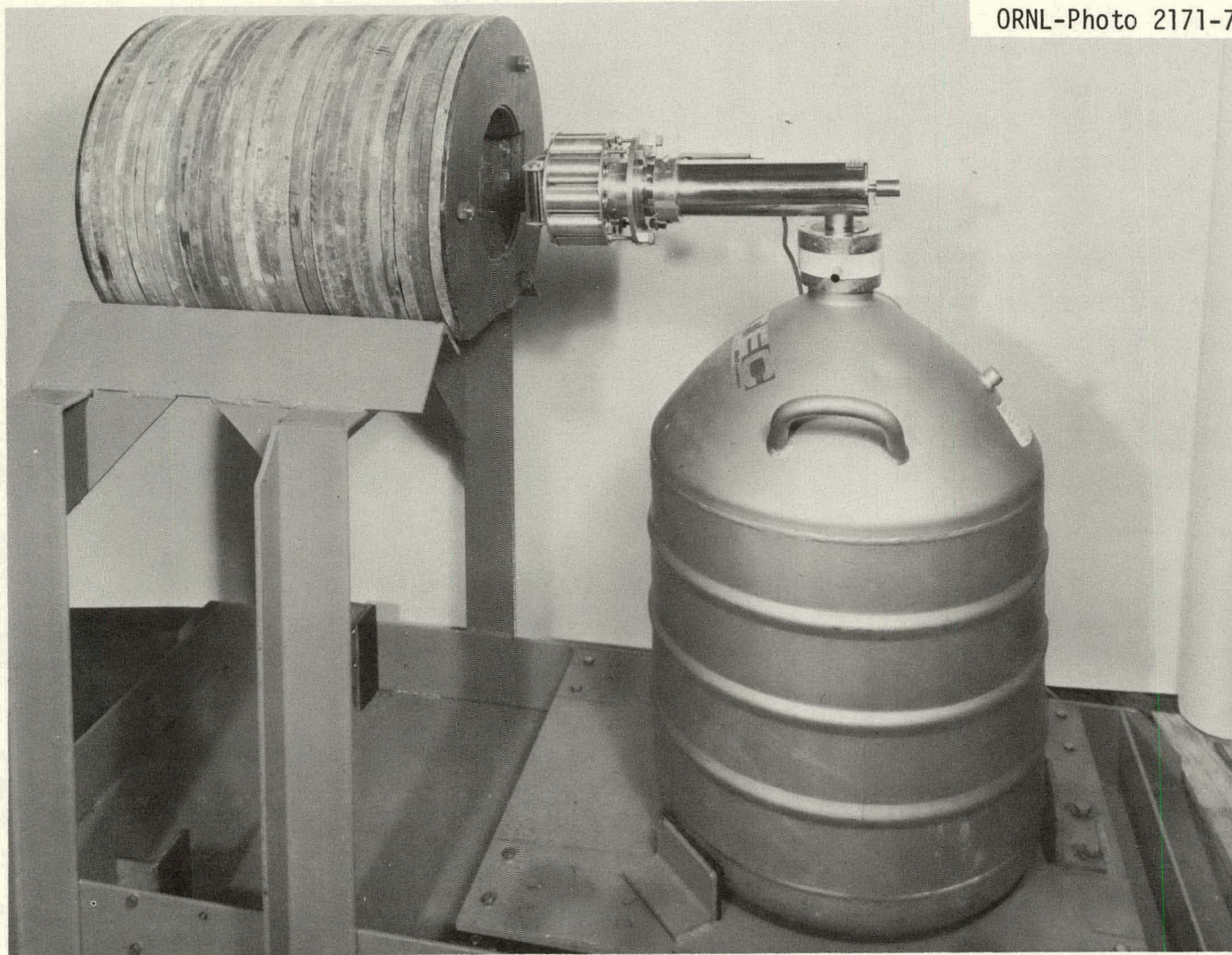


Fig. 6. Ge(Li) detector and lead shield arrangement used for counting soil and sediment samples

complete analysis by computer later. In addition, data for each of several peaks corresponding to individual photons from radon daughters were recorded on paper tape. From the field laboratory in Salt Lake City, these paper tapes were read into the PDP-10 computer at ORNL on a time sharing basis via teletype and FTS phone service at the end of each day.

Energy calibration of the Ge(Li) detector was controlled through the use of isotopic sources of ^{57}Co , ^{22}Na , ^{137}Cs , ^{60}Co , ^{88}Y , and ^{40}K . A calibration check was completed each day prior to beginning sample counting. In order to maintain linearity of the ADC, a spectrum stabilizer was utilized. This instrument is adjusted so that two individual photon energies are detected and maintained in two channels at separate ends of the scale. These two calibration points helped maintain an energy span of 1 keV per channel. Efficiency calibration was obtained through the use of the uranium ore standard samples as were used for the NaI crystal. An analysis of the counting data is accomplished through a linear least squares fitting routine. Net adjusted areas under photo peaks of interest are compared with an extensive radionuclide library.⁽⁴⁵⁾ Data from the computer are presented for each radionuclide as a weighted mean with standard deviation.

External Gamma-Ray Detector

A gamma radiation survey was made on and around the mill site and tailings pile. The instrument used for these measurements was a "Phil" gamma-ray dosimeter.⁽⁴⁶⁾ The basic unit is a 6-in. long 30 mg/cm² glass-walled organic filled Geiger-Muller (G-M) tube with an energy compensation shield made of tin and lead. Pulses from this unit are counted

with a battery-powered portable scaler. Typically, G-M counters are not used for dosimeters because of a peaked response at low photon energies. However, perforated layers of tin (1.0 mm), and lead (0.1 mm), may be used as an energy compensation filter to flatten this peaked response at photon energies below about 200 keV. The energy response of this counter and filter arrangement has been demonstrated by Thorngate.⁽⁴⁷⁾ Sealed sources of ^{137}Cs and ^{226}Ra were used for calibration. It was found that the response of this detector was:

$$1 \text{ mR/hr} = 3400 \text{ counts/min.} \quad (6)$$

For each gamma-exposure rate reading, a minimum of three 1-min counts were recorded. The means of these readings (less background) were used to determine the exposure rate to external gamma rays.

Radon Daughter Sampler

Radon daughter concentrations were measured with a sampling and counting instrument which has been in use at ORNL for several years⁽⁴⁸⁾ and used briefly in the remedial action program in Grand Junction.⁽⁴⁹⁾ The filter counter for this sampling device, as shown in Fig. 7, utilizes a modified gas flow alpha counter for housing a 600 mm² silicon diode. Normally, this type detector is operated in a vacuum chamber. However, in this case, it was found that by flowing helium at atmospheric pressure through the assembly, absorption of alpha particles was small relative to absorption in air. Furthermore, it was not necessary to cycle the bias voltage of the diode because there were no changes in pressure.

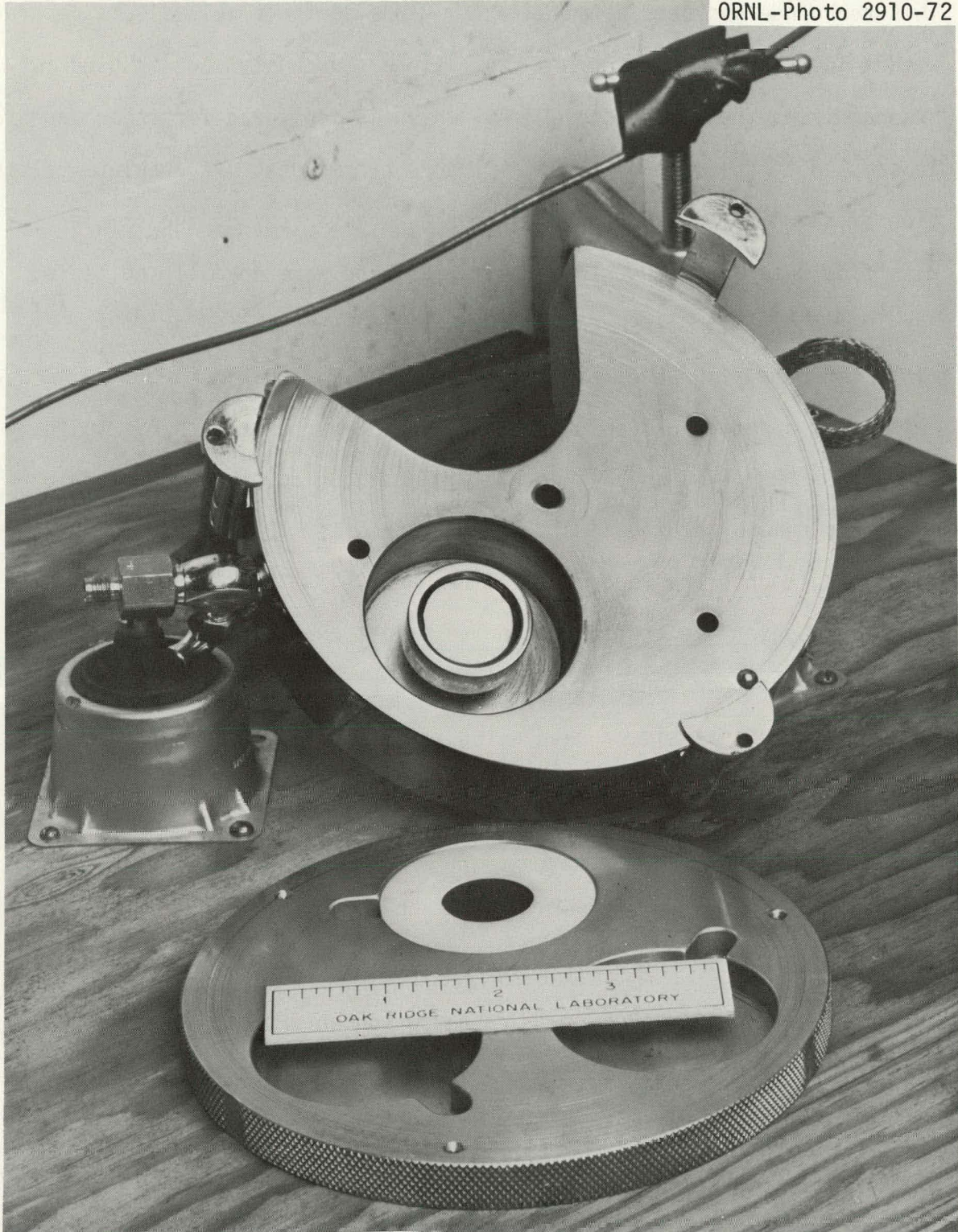


Fig. 7. Exploded view of system used to count radon daughters collected on filters, showing silicon detector, housing, and sample tray.

This technique has been patented.⁽⁵⁰⁾ Alpha particle pulses were recorded with a 100 channel analyzer. A small alpha source standard is used for standardizing the energy scale. Air which is monitored for radon daughters is sampled at the rate of 12 to 14 liters per minute. An absolute calibration of the airflow was provided through a comparison of the sampler's mass flow meter and a wet test meter. Samples were normally collected for 10 min, and the first count of the filter was started at 2 min after removal of the sample and continued for 10 min. For this case, a determination was made of the number of counts due to the decay of ^{218}Po (RaA) and ^{214}Po (RaC'). A second count was started 15 min after removal of the sample and continued for 15 min. In this case, counts are recorded from the decay of ^{214}Po (RaC'). Data from the counter are stored in a pulse height analyzer and reduced by computer. The code for this analysis is explained in detail elsewhere.⁽⁵¹⁾ Results of the analysis of data using this code are presented as concentrations of RaA, RaB, and RaC. In addition, a value for the working level concentration is also provided along with an estimate of the error associated with each reported value.

Radon Monitor

The instrument used to measure radon concentrations in air consists of a series of 95-ml Lucas chambers and a readout unit.* Each chamber is evacuated to approximately 1-mm Hg and then opened to atmospheric pressure in the area where a radon measurement is required. No filtration is used for sampled air. The short-lived daughters of radon drawn into the chamber are allowed to decay prior to counting the flask. The

*System was manufactured by Johnston Laboratories, Inc.

results from this instrument and the radon progeny monitor provided an estimate of the degree of equilibrium between radon and its daughters in the selected locations where air samples were taken.

5. EXPERIMENTAL RESULTS

During the course of this assessment program, numerous measurements were made to characterize the spread of contamination from the tailings pile to adjacent areas. When the program began, plans called for the measurement of radium concentrations in samples of soil and water and the determination of concentrations of radon and radon progeny in air in selected locations. While field work was in progress, it was decided to extend the studies to include assays for ^{210}Pb and ^{230}Th in soil and water samples collected in or near points of public access. Most of these additional measurements were made for samples collected in surface stream beds in the immediate area and in areas influenced by surface runoff. Only those results which are important to the consideration of health effects are reported here.

Background Radionuclide Concentrations

Remedial action may be required at the Vitro mill site and at others in western states. The extent of remedial action is dependent on the projected health effects for each site and on the projected cost per health effect avoided. In some cases, remedial action may involve removal of the tailings to a remote site of low population density. In the event tailings are removed, an acceptable standard must be adopted.

In order to evaluate the practicability of a cleanup criterion of twice background for the residual radium contamination levels in soil, it was necessary that the average concentration of radium in soils be determined for the general area of the Vitro site. Samples of soil were collected throughout the Salt Lake Valley ranging from Ogden to Provo

on a north-to-south line and from the shore of the great Salt Lake to the foothills of the Wasatch Mountain range as shown on the map in Fig. 8. These soil samples were analyzed for ^{226}Ra , ^{232}Th , and ^{238}U . The results of these determinations are given in Table 3. Background ^{226}Ra concentrations are distributed log-normally and are best described by a geometric mean of 1.46 pCi/g and a geometric standard deviation of 1.23.

Anspaugh⁽⁵²⁾ reports that in approximately 100 sample locations in Utah (in no case was it likely that the Vitro pile was contributing to the measurement) determinations of the soil radium concentration varied from 0.38 pCi/g to 2.97 pCi/g. The geometric mean of these values was found to be 0.87 pCi/g, and the geometric standard deviation was 1.4. Most of these measurements were made in situ using a Ge(Li) detector and a computer based pulse height analysis system.^(52,53) Results obtained by this technique were within 10% of the values determined by radiochemical and mass spectrometric analyses.

Pendleton⁽⁵⁴⁾ reported measurements of radium "equivalents" in soil samples which had been sealed for ingrowth of radon daughters—a technique similar to that used in this investigation. His samples were collected over a wide area of the Salt Lake Basin and were generally representative of the concentration of radium to a depth of 10 cm below the surface. Individual samples of each centimeter of soil were collected. A comparison was made between some of Pendleton's results of soil activity measurements and results of this study. In two sets of samples taken in Magna, Utah, he reported an average for ^{226}Ra of 1.1 ± 0.17 pCi/g

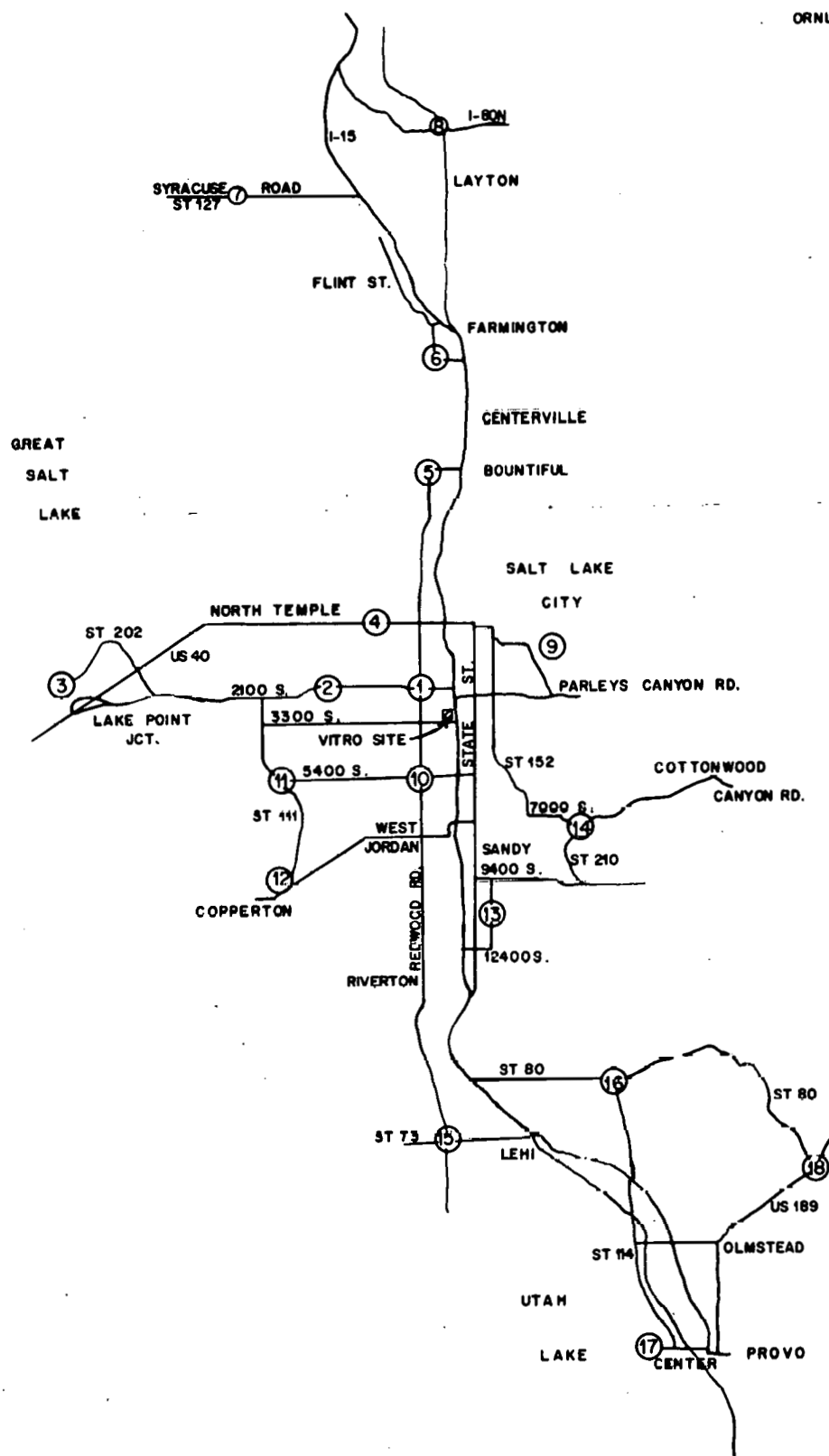


Fig. 8. Map of Salt Lake Valley showing location of background soil samples.

Table 3. Background radionuclide concentrations in soil samples in the Salt Lake Valley

Sample number	Nuclide concentration		
	^{226}Ra (pCi/g)	^{238}U (pCi/g)	^{232}Th (pCi/g)
UT-1	1.84 \pm 0.02	1.61	0.79
UT-2	1.42 \pm 0.02	1.12	1.26
UT-3	1.03 \pm 0.03	2.44	0.26
UT-4	1.59 \pm 0.05	2.15	1.22
UT-5	1.10 \pm 0.04	1.40	0.88
UT-6	1.76 \pm 0.05	1.34	1.82
UT-7	2.05 \pm 0.05	2.19	2.38
UT-8	1.76 \pm 0.04	2.72	1.78
UT-9	1.07 \pm 0.03	1.19	1.00
UT-10	1.72 \pm 0.05	2.28	1.24
UT-11	1.40 \pm 0.03	1.74	1.22
UT-12	1.53 \pm 0.06	1.94	1.52
UT-13	1.12 \pm 0.03	1.03	1.40
UT-14	1.74 \pm 0.04	0.80	1.68
UT-15	1.65 \pm 0.04	1.18	1.69
UT-16	1.70 \pm 0.05	1.23	1.40
UT-17	1.35 \pm 0.04	0.82	0.91
UT-18	1.55 \pm 0.05	1.53	0.81

from 0 to 2 cm in one set and 1.68 ± 0.122 pCi/g from 0 to 9 cm in the other set. These values may be compared in a general way to our background measurements; see, for example, samples UT-2 (1.42 ± 0.02 pCi/g) and UT-11 (1.40 ± 0.03 pCi/g) in Table 3. Pendleton also presented, for these two sets of samples, ^{232}Th concentrations of 1.3 pCi/g and 1.5 pCi/g, respectively; we obtained concentrations of 1.26 pCi/g and 1.22 pCi/g in our respective samples. A sample collected by Pendleton in April 1974 near the west side of the sewage treatment plant property at the Vitro site was at the approximate position of sample SS-52 in Fig. 4. He reported a ^{226}Ra concentration of 104.8 pCi/g from 0 to 2 cm whereas our results indicate 108 pCi/g. In a second sample taken at the sewage treatment plant between two settling tanks, he reported 762 pCi/g from 0 to 2 cm; and from this study (SS-51) a concentration of 570 pCi/g was found. The variation in this latter comparison is not surprising since it is unlikely that the samples were collected within 15 to 20 m of one another, and more than a year had elapsed between times of sample collection. The variability of radium activity in soil taken from areas lying close to the pile is discussed later in this chapter.

Background water samples were collected from four wells (see Fig. 9) located near the Vitro site, but at locations far enough off the site to be considered unaffected by the pile. The uranium and radium concentrations found in these samples are given in Table 4. Analyses were restricted to these radionuclides because uranium migrates at the fastest rate in groundwater and ^{226}Ra has the lowest $\text{MPC}_w^{(37)}$ among the tailings

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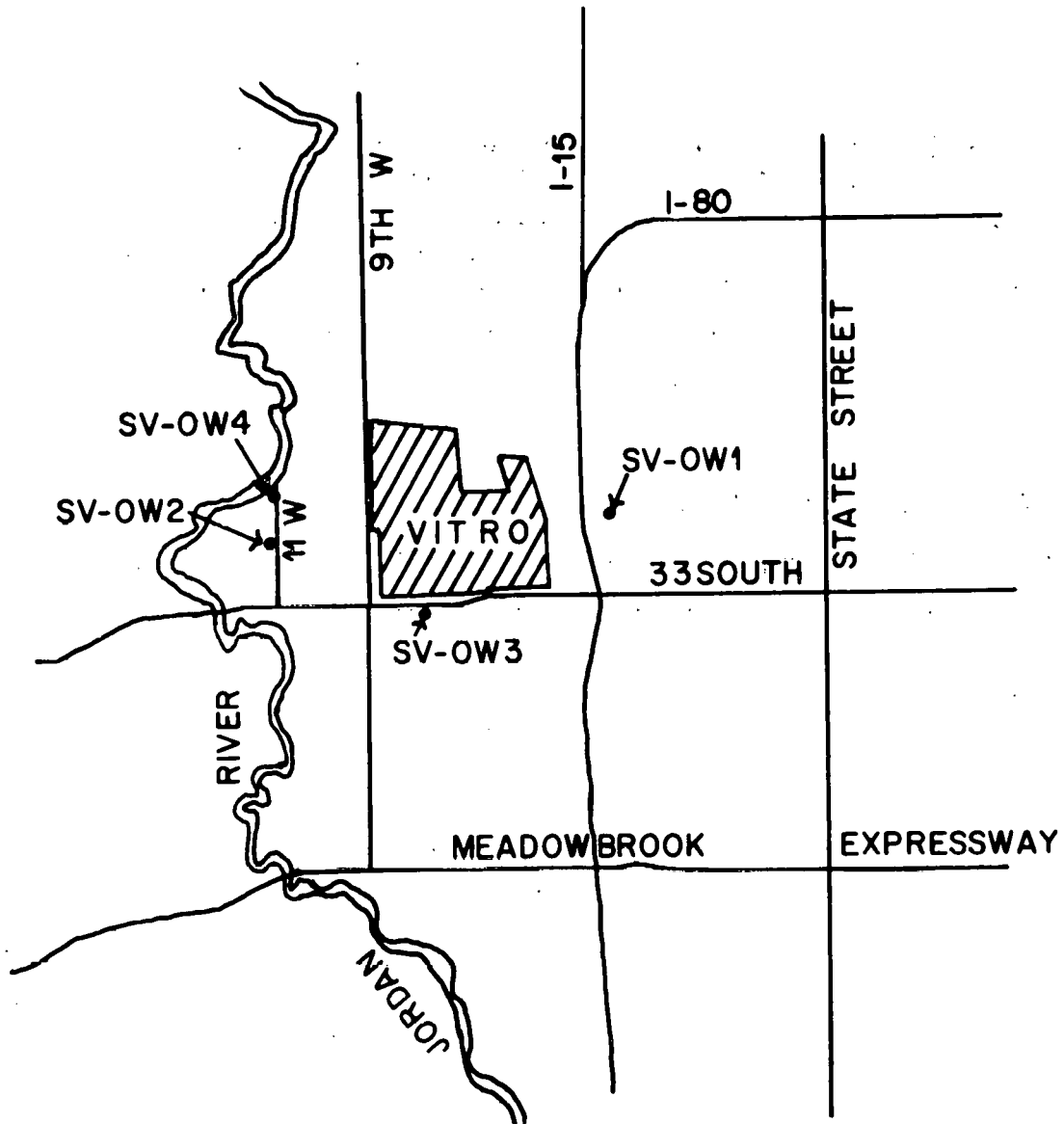


Fig. 9. Plan view of Vitro mill site showing location of water samples from private wells.

Table 4. Radionuclide concentrations in well water near the Vitro site, Salt Lake City, Utah

Sample number	Location of station	^{238}U (pCi/liter)	^{226}Ra (pCi/liter)
SV-OW1	3106 South 400 West	0.71	0.19
SV-OW2	3200 South 1100 West	0.22	0.08
SV-OW3	837 West 3300 South	neg.	0.48
SV-OW4	3090 South 1100 West	0.06	0.15

components. Tanner⁽⁵⁵⁾ has reported radium concentrations in water from wells in the Salt Lake Valley at locations remote from the Vitro site. These values ranged from 0.1 pCi/liter to 2 pCi/liter of ^{226}Ra .

Radioactivity in Groundwater in Tailings Pile

During the drilling operations at the Vitro site, it was observed that groundwater existed 3 to 4 ft below the surface even in areas where the tailings surface was 15 or more ft higher than normal grade elevation. The overburden of tailings and the presence of an unconfined aquifer in the subsoil account for this shallow groundwater table. Samples were taken from the water present in each of the holes drilled into the tailings pile. Water was also collected from all surface streams. After centrifuging for removal of bulk solids, each sample was pressed through a 0.45- μm filter. The water was then dispensed (25 ml) into the previously described plastic scintillation counting vials, sealed, and stored for equilibration. The activity levels were so low that it was difficult to obtain statistically accurate estimates of the nuclides present by gamma-ray-spectroscopy techniques. Therefore, a composite sample was made up of one 25-ml sample of water from each hole in the tailings pile. Radiochemical techniques were used to analyze this composite water sample. The results are given in Table 5. The level of radioactivity in flowing streams was indistinguishable from background.

A confined, artesian aquifer exists about 50 ft below the base of the tailings pile. Because the piezometric surface of this aquifer is higher than grade elevation, groundwater flow is toward the surface.⁽⁴⁾

Table 5. Radionuclide concentration in composite sample from groundwater in the Vitro tailings pile

Radionuclide	Concentration (pCi/liter)
^{210}Po	10
^{226}Ra	100
^{230}Th	5
^{238}U	400

Horizontal movement of the surface aquifer is toward the Great Salt Lake at a hydraulic gradient of about 0.06% through a relatively impermeable clay.⁽⁴⁾ Thus, minimal vertical and horizontal movement of the activities shown in Table 5 can be expected.

Radionuclide Concentrations On the Site

The Vitro site was divided into three general areas. Sampling locations in each area were used to obtain a profile of the gamma-ray intensity in the soil and, hence, a plot of the radium concentration as a function of depth below the surface. Typical radioactivity profiles are shown in Fig. 10 (note the interface between tailings and subsoil as indicated by an arrow). In some cases, it was possible to plot the measured concentration of uranium and radium at the same depth. It was noted earlier that an anionic uranyl complex may be expected to migrate more rapidly than radium, provided the groundwater is moving. Therefore, any noted imbalance of radium and uranium activities could indicate a preferential leaching of uranium. The interface between tailings and subsoil is detected easily, and its position may be estimated from the data in Fig. 10. However, some difficulties were encountered frequently in the insertion of a length of PVC tubing into the drilled holes. The possibility exists that some high specific activity tailings were brushed into the bottom of the hole or that the ever present groundwater washed some of the sandy textured material from upper levels into the bottom. It should also be pointed out that the split-spoon sampler used for sampling soil below the interface may have caused some side wall contamination into the sampling area. Therefore, we cannot place an estimate on the probable error in the position

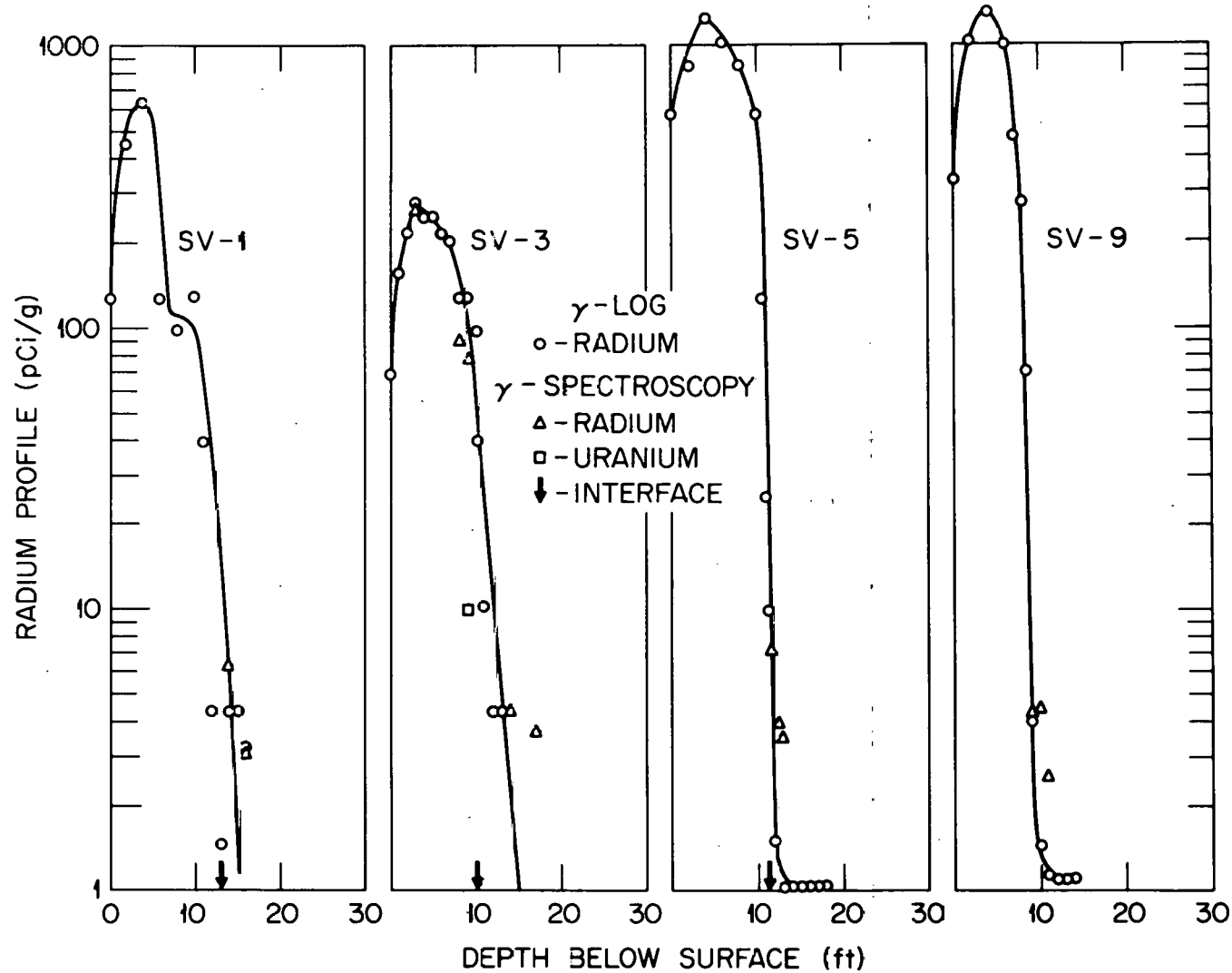


Fig. 10. Typical profile of ^{226}Ra concentration as a function of depth below the surface of Vitro tailings.

of the interface or in the depth below the interface to which radium or uranium has leached. However, the downward migration of radionuclides is unlikely to be greater than that shown in Fig. 10 because the groundwater flow is toward the surface.

Radionuclide Concentrations Off the Site

A series of soil samples was collected in areas away from the tailings pile to determine the spread of contamination due to wind and excessive runoff. This series included a set of 60 locations (labeled in Fig. 4 as SS-1 through SS-60) at distances of 400 to 1000 m from the center of the Vitro site; soil was taken from the surface and at one foot below the surface. Also included in this series was a group of sediment samples from the Jordan River, Mill Creek, and a drainage ditch used to carry effluent from the sewage treatment plant to the Jordan River.

Concentrations of ^{226}Ra in these surrounding site (SS) locations are presented in Table 6. Some data are given for uranium concentrations. It is seen that some of the samples were taken from areas around the sewage treatment plant. On the west side of this site, some tailings material was used in construction. The rather high radium concentrations for samples SS-51 and SS-52 verify the presence of tailings in that area. In a number of locations, the radium concentration is greater at a depth of one foot than on the surface. This is an indication that the ground on which tailings were blown was disturbed after deposition, or that tailings were moved to that location and used in grading or construction. The results of these offsite analyses indicate that minimal activity is available for producing exposure through the terrestrial pathway.

Table 6. ^{226}Ra concentration in soil collected off the site
in areas surrounding the Vitro pile

Sample location ^a	^{226}Ra Concentration		Sample location	^{226}Ra Concentration	
	Surface (pCi/g)	12-in. deep (pCi/g)		Surface (pCi/g)	12-in. deep (pCi/g)
SS-1	12.42 ± 0.37		SS-31	2.05 ± 0.08	20.3 ± 0.47
SS-2	1.85 ± 0.08		SS-33	5.46 ± 0.16	1.22 ± 0.05
SS-3	2.50		SS-34	3.04 ± 0.10	2.14 ± 0.07
SS-4	0.87 ± 0.04	4.1	SS-35	1.43 ± 0.06	2.23 ± 0.07
SS-6	34.2		SS-36	0.82 ± 0.03	1.04 ± 0.06
SS-7	1.6		SS-37	35.93 ± 0.89	1.6 ± 0.08
SS-8	11.		SS-38	18.0 ± 0.44	
SS-9	6.5		SS-39	45.6 ± 1.04	22.23 ± 0.51
SS-10	6.	2.5	SS-40	88.8 ± 2.1	2.85 ± 0.17
SS-11	3.44	2.44	SS-41	37.6 ± 0.94	67.4 ± 1.62
SS-12	2.5	2.5	SS-42	7.47 ± 0.27	2.38 ± 0.10
SS-13	20.9 ± 0.48	2.42 ± 0.08	SS-43	22.8 ± 0.51	5.66 ± 0.21
SS-14	1.91 ± 0.07	6.6 ±	SS-44	1.01 ± 0.05	1.06 ± 0.05
SS-15	2.88 ± 0.08	5.6 ± 0.16	SS-45	4.65 ± 0.13	1.81 ± 0.08
SS-16	7.34 ± 0.02	3.26 ± 0.01	SS-46	13.3 ± 0.33	2.87 ± 0.11
SS-17	1.97 ± 0.08	1.37 ± 0.05	SS-47	108.9 ± 2.5	2.52 ± 0.10
SS-18	1.27 ± 0.04	1.06 ± 0.06	SS-48	65.5 ± 1.6	7.72 ± 0.21
SS-19	1.56 ± 0.05	1.21 ± 0.05	SS-49	65.	15.4 ± 0.45
SS-20	1.46 ± 0.07	1.33 ± 0.05	SS-50	43.4 ± 1.0	18.4 ± 0.45
SS-21	11.13 ± 0.29	289.3 ± 6.45	SS-51	559.2 ± 12.5	465. ± 10.5
SS-22	7.37 ± 0.19	5.34 ± 0.15	SS-52	110.1 ± 2.6	115.4 ± 2.6
SS-23	50.97 ± 1.1	1.85 ± 0.06	SS-53	4.52 ± 0.12	6.99 ± 0.18
SS-24	3.98 ± 0.11	1.26 ± 0.05	SS-54	5.44 ± 0.17	2.80 ± 0.09
SS-25	3.	1.36 ± 0.06	SS-55	11.9 ± 0.3	8.7 ± 0.23
SS-26	10.72 ± 0.28	1.61 ± 0.07	SS-56	1.93 ± 0.07	2.07 ± 0.07
SS-27	31.	1.61 ± 0.07	SS-57	2.08 ± 0.09	0.96 ± 0.04
SS-28	12.9 ± 0.31	7.43 ± 0.21	SS-58	151.3 ± 3.7	16.6 ± 0.45
SS-29	1.62 ± 0.05	1.15 ± 0.04	SS-59	7.90 ± 0.22	8.48 ± 0.23
SS-30	8.38 ± 0.22	2.52 ± 0.09	SS-60	12.1 ± 0.31	138.3 ± 3.3

^a No samples collected for locations SS-5 and SS-32.

Radionuclide concentrations in waterway sediments are given in Table 7. These results show no indication of contamination in the Jordan River. The contamination present in the drainage ditch (particularly ^{230}Th) and in Mill Creek may be the result of previous liquid discharges from the mill. The absence of contamination in the surface water samples and in Jordan River sediments precludes the possibility of exposure through the aquatic pathway. Nevertheless, dredging of Mill Creek and the drainage ditch and proper disposal of the spoils may be a desirable remedial action.

There are several locations in Salt Lake City to which tailings have been moved for private use. These locations are discussed by Ford, Bacon and Davis, Utah, in their report of the Vitro site.⁽⁴⁾ No results are presented here for measurements made at these locations. However, several commercial structures have been constructed in recent years along the east side of 900 West Street between 33rd South and the drainage ditch for the sewage treatment plant. Several holes were drilled through the concrete floors in these buildings and soil was collected at several depths (see Fig. 11). It was found that two of these structures had tailings under the floors. In one of these buildings, tailings ranged to at least 5 ft deep. Results of measurements on samples collected under these buildings are given in Table 8. The result of this type situation is a continuously elevated concentration of radon inside the building. During some periods of the day the radon progeny concentration may exceed 1 to 2 working levels (WL). For residences, the Surgeon General of the U. S. has recommended⁽³⁾ that the annual average working level concentration not exceed 0.01 WL.

Table 7. Radionuclide concentration in sediment samples

Sample number	Concentration			
	^{226}Ra (pCi/g)	^{230}Th (pCi/g)	^{238}U (pCi/g)	^{210}Pb (pCi/g)
WJ-1	2.1	0.4	3.6	0.2
WJ-2	2.0	2.4	1.1	0.7
WJ-3	0.9	0.6	2.1	0.3
WJ-4	1.2	3.4	2.6	1.5
WD-1	51.	221.	33.	95.
WD-2	67.	18.	182.	7.5
WD-3	51.	18.	41.	7.8
WD-4	116.	284.	129.	122.
WD-5	6.5	6.3	6.	2.4
WM-1	1.8	1.3	15	0.6
WM-2	4.3	2.4	3.1	1.0
WM-3	5.4	1.8	2.2	0.8
WM-4	2.4	1.3	0.9	0.6
WM-5	2.7	3.0	0.6	1.3

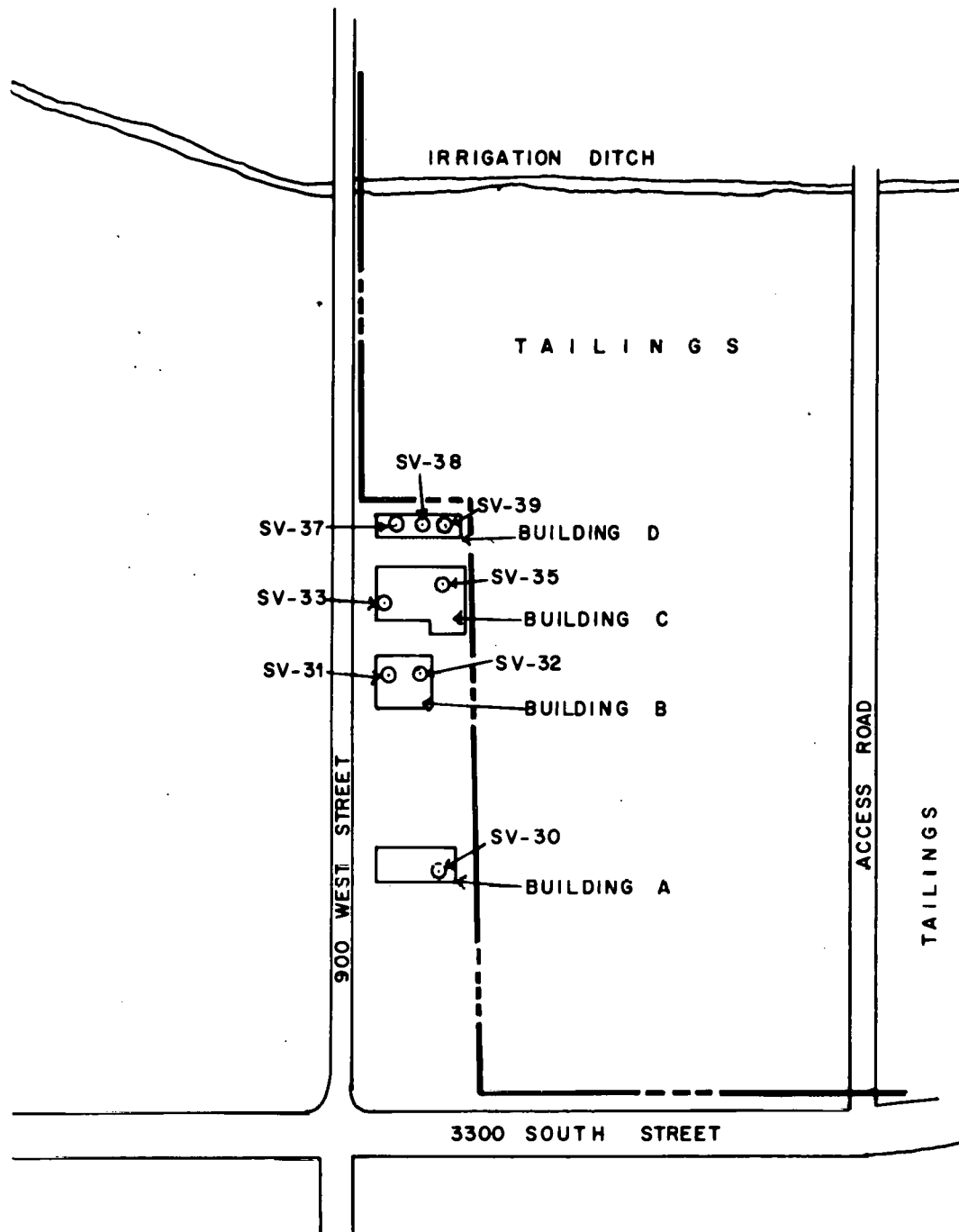


Fig. 11. Plot of southwest area of tailings pile showing the location of commercial structures along 900 West Street and the location of sampling points in these buildings.

Table 8. ^{226}Ra concentration in soil beneath the floor of commercial structures along southwest edge of Vitro tailings pile

Building	Sample location	Depth of sample (in.)	^{226}Ra concentration (pCi/g)
A	SV-30	12	1.07
A	SV-30	24	1.56
A	SV-30	36	2.20
B	SV-31	0-48	0.84
B	SV-31	60	2.82
B	SV-31	72	2.19
B	SV-32	0-48	1.54
B	SV-32	60	3.35
B	SV-32	72	2.75
B	SV-32	96	2.28
C	SV-33	0-12	594.
C	SV-33	12-16	741.
C	SV-33	16-23	914.
C	SV-33	23-29	175.
C	SV-33	29-37	340.
C	SV-33	37-50	54.
C	SV-33	50-59	3.
C	SV-35	0-12	49.
C	SV-35	20	830.
C	SV-35	30	31.
C	SV-35	48-60	7.3
D	SV-37	0-12	1.39
D	SV-37	12-24	1.53
D	SV-37	36-48	1.64
D	SV-38	0-12	765.
D	SV-38	12-24	775.
D	SV-38	24-36	781.
D	SV-38	36-48	812.
D	SV-38	48-60	600.
D	SV-39	0-12	770.
D	SV-39	12-24	730.
D	SV-39	24-36	800.
D	SV-39	36-48	810.
D	SV-39	48	590.

Elemental Composition of Material from the Vitro Tailings Pile

A total of 12 samples of tailings material were gathered at random locations on the pile. These samples were analyzed by the ORNL Analytical Chemistry Division utilizing a spark-source mass spectrometer. A semi-quantitative analysis (\pm factor of 2) was requested to determine if potentially hazardous or toxic materials were present in significant quantities. The elemental concentrations in these samples are given in Table 9 for those elements whose wt. % exceeded 0.05. After this analysis had been completed, a question was raised about the concentration of arsenic, selenium, and uranium. Because of analytical costs, a composite was formed of these 12 samples and one additional analysis was made. The concentration of arsenic was 0.03%, selenium was 0.003%, and uranium was 0.03%.

Direct Gamma-Ray Exposure Rates On and Near the Tailings Pile

The direct gamma-ray exposure around most of the inactive uranium mills does not constitute a significant hazard because of the fact that most of these sites are in secluded areas. However, there are some notable exceptions, such as the Vitro site, where a large population group resides near the pile. It was decided to calculate the potential health effects from direct gamma exposure; thus, a series of measurements were made at each place a soil sample had been taken and at locations near the pile and along public roads. Gamma-ray exposure as a function of distance from the center of the pile (sample point SV-15) is plotted in Fig. 12. There is a reduction of exposure rate with distance; however, the relationship does not follow the inverse square law because of the large physical size of the pile.

Table 9. Elemental composition of material from the Vitro tailings pile
(Listed values are wt.% if greater than 0.05)

Element	Random sample numbers											
	1	2	3	4	5	6	7	8	9	10	11	12
Al	2.	>5.	1.	4.	2.	>5.	>5.	>5.	>5.	>5.	2.	5.
Ca	0.5	5.	0.1	1.5	0.6	1.	5.	5.	>5.	0.7	5.	>5.
Cu	-	-	-	-	-	-	-	-	-	-	-	0.15
Fe	0.7	1.	1.	0.9	0.9	1.	1.5	1.	1.	4.	≥5.	≥5.
K	2.	0.9	3.	0.5	1.	2.	0.5	0.5	0.7	2.	0.4	1
Mg	0.5	0.8	0.1	1.	0.4	0.8	5.	3.	2.	1.	0.8	0.8
Mn	-	-	-	0.1	-	-	0.1	-	-	0.09	0.2	0.7
Mo	-	-	-	-	-	0.1	-	-	-	0.09	-	0.06
Na	0.8	0.3	2.	0.4	1.	1.	2.	1.	0.6	1.5	0.7	0.9
Ni	-	-	-	-	-	-	-	-	-	-	0.09	0.1+
Pb	-	0.05	-	0.05	-	0.2	-	-	-	0.1	-	-
Rb	0.05	-	0.1	-	0.05	0.08	-	-	-	0.2	-	0.05
Si	← Major Element →											
Ti	0.05	0.3	0.08	0.3	0.07	0.3	0.2	0.15	0.2	0.4	0.2	0.15
V	-	-	-	-	-	-	-	-	0.075	-	0.15	0.5
Zn	0.05	0.06	0.05	0.05	-	0.06	0.06	0.05	-	0.15	0.09	0.1

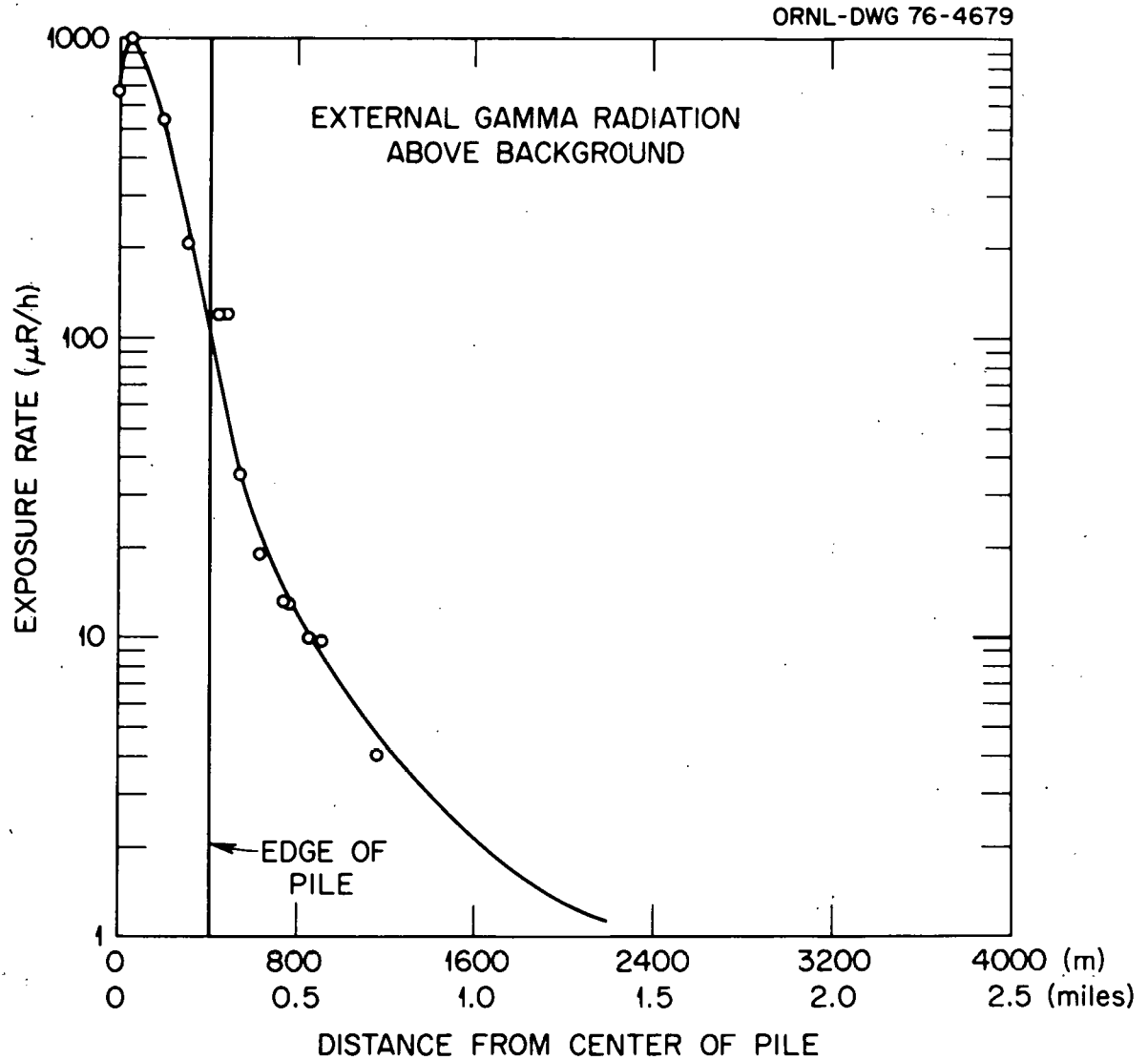


Fig. 12. External gamma radiation ($\mu\text{R}/\text{hr}$) as a function of distance from the center of the Vitro tailings pile.

Gamma-ray measurements were also made inside three of the commercial structures shown in Fig. 11. These levels are listed in Table 10. In building A, it is observed that the level is just over 200 $\mu\text{R/hr}$. For persons working in this building for a period of 40 hours per week for 50 weeks per year, the integrated exposure is 400 mrem/year.

Radon and Radon Daughter Concentration in Structures

During the course of this study, a series of radon and radon daughter samples were taken in the sewage treatment plant office building, several commercial buildings located near the pile, and in some residences. Measurements of radon concentrations were made by others and are given in reference (4). In most cases, air was sampled for radon with a Lucas chamber at the same time air was sampled for radon daughters. Given radon and radon progeny concentrations, one can determine the degree of equilibrium between radon and its daughters. A ratio of the activity of each successive daughter in the chain gives some information about the ventilation rate for the structure and permits the calculation of the working level concentration in that area. It should be pointed out that in a short period of time (such as the period of this investigation), the only physical measurements of significance are those which are non-varying. Measurements of radon and/or radon progeny concentrations made continuously for a short period (≤ 30 days) are insufficient for use in evaluating the radon exposure conditions over a long period (i.e. ≥ 1 year). Annual average concentrations require a knowledge of the seasonal variation in the distribution of values. The results of spot sampling for the concentration of radon daughters are presented in Table 11 for

Table 10. Gamma-ray exposure rates inside commercial structures along southwest edge of Vitro tailings pile

Building	Location of measurement	Exposure rate (μ R/hr)
B	10 ft from rear door	40.
B	Near front entrance	26.4
B	Near center of building	28.4
C	In storage room	104.
C	25 ft from east wall	191.
C	Near center of building	98.
D	Near front office	34.7
D	Near east wall	219.

Table 11. Concentration of radon and radon daughters

Date 1975	Time	Location	Concentration				Total daughters (pCi/liter)	Working levels (WL)	% Equil.	RaA Rn	RaB Rn	RaC Rn
			Rn (pCi/liter)	RaA (pCi/liter)	RaB (pCi/liter)	RaC (pCi/liter)						
9-25	11:00	Outside, 1745 S. Redwood		0.43 (19)	0.23 (36)	0.17 (34)	0.83 (53)	0.0022				
9-26	10:40	"		0.96 (14)	0.17 (75)	0.46 (20)	1.58 (78)	0.004				
9-26	14:40	Sewage treatment plant		0.78 (15)	0.5 (26)	0.26 (35)	1.56 (46)	0.0044				
9-26	14:40	Southwest room		0.6 (17)	0.33 (34)	0.19 (35)	1.09 (52)	0.0028				
10-2	09:35	"		3.19 (9)	2.76 (23)	2.11 (20)	8.07 (32)	0.025				
10-2	11:55	"		2.31 (10)	1.85 (25)	1.67 (19)	5.83 (33)	0.018				
10-2	16:10	"		1.02 (13)	0.5 (34)	0.6 (21)	2.18 (42)	0.0061				
9-26	11:25	Bldg. C, Fig. 11, store room 195.		132. (6)	77. (21)	63.21 (5)	272. (27)	0.76	39	0.68	0.39	0.32
10-1	09:43	"		139. (6)	108. (19)	61.1 (22)	308. (29)	0.92				
10-1	16:05	"	557.	372. (6)	179. (22)	130. (20)	681. (30)	1.77	32	0.67	0.32	0.23
9-26	13:00	Bldg. D, Fig. 11, shop	76.8	49.1 (6)	32.8 (23)	29. (18)	111. (30)	0.32	42	0.64	0.43	0.38
9-26	13:40	"		16.4 (7)	6.7 (25)	5.9 (19)	29. (32)	0.073 ^a				
10-1	10:22	"		2.17 (10)	0.4 (38)	0.4 (27)	2.97 (47)	0.0057 ^b				
10-1	11:22	"	6.95	4.94 (8)	0.9 (27)	0.4 (40)	6.18 (49)	0.011 ^b	16	0.71	0.12	0.06
10-1	14:50	"	16.2	9.46 (7)	1.76 (24)	0.8 (34)	12.0 (43)	0.021 ^b	13	0.58	0.11	0.05
10-1	15:30	"		6.38 (8)	1.16 (26)	0.6 (34)	8.12 (44)	0.015 ^b				
10-1	17:10	"	81.3	43.6 (5)	11.3 (20)	4.22 (32)	59.1 (38)	0.12 ^c	15	0.54	0.14	0.05
9-26	16:50	3226 W. 300 W inside		0.28 (24)	0.06 (93)	0.11 (34)	0.45 (102)	0.001				
9-26	17:30	" outside		0.34 (23)	0.10 (46)	0.05 (77)	0.48 (93)	0.001				
9-26	18:35	863 W. 33rd S. front room		1.54 (11)	0.49 (31)	0.36 (28)	2.39 (43)	0.0054				
9-27	09:00	Bldg. B, Fig. 11, store room		13.2 (7)	9.82 (22)	7.83 (19)	30.8 (30)	0.092				
9-27	10:45	"		7.66 (7)	5.23 (28)	6.18 (16)	19.1 (33)	0.057				
9-27	11:20	" outside		2.13 (10)	1.66 (24)	1.33 (21)	5.13 (34)	0.016				
9-29	12:15	Bldg. A, Fig. 11, outside		0.68 (16)	0.02 (283)	0.19 (26)	0.9 (284)	0.0015				
9-29	14:10	738 E 1930 S, basement		0.38 (21)	0.05 (68)	0.14 (33)	0.6 (78)	0.0014				
9-29	14:50	"		0.61 (17)	0.06 (107)	0.17 (29)	0.8 (112)	0.0015				
9-29	15:25	" outside roof		0.4 (21)	-	0.1 (26)	0.4 (74)	0.0005				
9-29	16:45	G.C. warehouse	6.56	0.9 (14)	0.07 (28)	0.6 (23)	2.27 (39)	0.0069 ^d	11	0.14	0.11	0.09
9-29	17:20	"		0.8 (16)	0.6 (30)	0.5 (24)	1.86 (41)	0.0057				

^aDoor opened approximately one hour prior to measurement.^bDoors open.^cSample collected 30 minutes after shop closed.^dNear open door.

several locations near the tailings pile. Only very tentative conclusions may be drawn from these data because of the short-term nature of the measurements. It is possible, however, to observe the disequilibrium which exists between RaA, RaB, and RaC and in some cases between these nuclides and Rn.

Calculation of ^{222}Rn Emanation and Dispersion From the Tailings Pile

Average ^{226}Ra concentrations in tailings were determined from gamma logs and γ spectroscopy for each of the test holes SV-1 thru SV-29. The representative samples shown in Table 12 were then averaged to estimate the overall average ^{226}Ra content of the pile as 460 pCi/g. The average depth of contamination was estimated to be 13 ft. The contaminated area was estimated from a large scale drawing of Fig. 4 as approximately 117 acres ($4.7 \times 10^5 \text{ m}^2$). Assuming that the specific gravity of the tailings material is 1.6 g/cm³,^(10,56) the total ^{226}Ra inventory may be calculated as 1400 Ci which compares favorably with the estimate of 1380 Ci based on the assays of ore processed by the Vitro mill.⁽³⁾

The estimated mass of contaminated material, 3.3×10^6 tons, is almost double the amount of uranium ore processed by the mill, 1.7×10^6 tons.⁽³⁾ The additional mass includes an undetermined volume of low-activity vanadium tailings mixed with and piled upon the uranium tailings, the contaminated soil beneath the tailings piles and the ore storage area, and a large quantity of sewage sludge.

Emanation of ^{222}Rn from the surface of the Vitro pile was estimated by use of Eq. (1) in Chapter 3 of this report. Porosity was assumed to be 0.4,^(10,55) specific gravity 1.6 g/cm³,^(10,56) bulk diffusion

Table 12. Determination of average ^{226}Ra concentration of Vitro tailings pile

Hole number	^{226}Ra (pCi/g)	Depth (ft)
SV- 1	180	16
SV- 2	170	16
SV- 3	140	14
SV- 4	580	13
SV- 5	790	12
SV- 8	100	5
SV- 9	760	10
SV-10	720	14
SV-12	600	18
SV-13	550	23
SV-14	260	21
SV-15	360	17
SV-16	990	5
SV-17	940	5
SV-18	1570	12
SV-19	290	5
SV-20	280	10
SV-21	170	12
SV-22	140	12
SV-23	170	3
SV-25	180	18
SV-26	640	12
SV-27	90	17
SV-28	310	11
Average	460	13

coefficient $0.028 \text{ cm}^2/\text{sec}$ (equivalent to that observed for sand @ 0% moisture⁽¹⁸⁾), and emanation fraction 0.2 as discussed in Chapter 3. Diffusion properties of the subsoil were assumed to be similar to those for water-saturated clay.⁽¹⁸⁾ These assumptions coupled with an average ^{226}Ra content of 460 pCi/g and an average depth of 13 ft. yielded a ^{222}Rn emanation rate of $550 \text{ pCi/m}^2\text{-sec}$.

Radon emanation rates have been measured at various locations on the Vitro tailings pile.⁽⁵⁷⁾ The average of these short-term measurements was $320 \text{ pCi/m}^2\text{-sec}$. This measured value is only about 60% of the calculated value. However, the physical descriptions of the measured sites imply that the radon emanation rate from the ore storage area was not measured. This is the area containing the highest concentration of ^{226}Ra contamination. Furthermore, all assumptions related to the calculated emanation rate were made to maximize computed results. Thus, the calculated emanation rate of $550 \text{ pCi/m}^2\text{-sec}$ should be a conservative estimate of the annual average.

An area source model⁽²¹⁾ was used in conjunction with the most recent wind profile and stability data for Salt Lake City⁽⁵⁸⁾ to estimate the outdoor ^{222}Rn concentrations attributable to the tailings pile. Results of dispersion calculations for an area source strength of $2.6 \times 10^8 \text{ pCi/sec}$ are shown in Table 13. The horizontal lines on Table 13 correspond to distances at which the ^{222}Rn concentration attributable to the tailings pile is approximately equal to background (0.38 pCi/liter)⁽²⁶⁾ and 10% of background. As may be seen from Table 13, radon attributable to the tailings pile is about 10% of background at a distance of 2 miles

Table 13. ^{222}Rn concentration (pCi/liter) resulting from the Vitro tailings pile

Dist. from edge of pile (mile)	Compass direction															
	N	NNE	NE	ENE	E	ESE	SE	SSE	S	SSW	SW	WSW	W	WNW	NW	NNW
0	4.1	3.1	2.3	1.5	1.7	1.8	2.4	2.4	2.3	2.1	1.7	1.2	1.7	2.6	4.1	4.5
0.25	1.2	0.72	0.34	0.36	0.42	0.47	0.45	0.64	0.72	0.58	0.31	0.21	0.27	0.57	1.1	1.4
0.50	0.68	0.27	0.17	0.16	0.21	0.23	0.22	0.28	0.40	0.27	0.15	0.09	0.13	0.25	0.54	0.70
0.75	0.46	0.12	0.13	0.95	0.13	0.13	0.13	0.15	0.28	0.14	0.09	0.05	0.08	0.13	0.33	0.42
1.75	0.15	0.04	0.03	0.03	0.04	0.04	0.04	0.05	0.09	0.04	0.02	0.01	0.02	0.04	0.10	0.13
2.75	0.08	0.02	0.02	0.01	0.02	0.02	0.02	0.02	0.05	0.02	0.01	0.007	0.01	0.02	0.05	0.07
3.75	0.05	0.01	0.01	0.009	0.01	0.01	0.01	0.01	0.03	0.001	0.005	0.004	0.007	0.01	0.03	0.04
4.75	0.03	0.005	0.008	0.006	0.009	0.009	0.01	0.009	0.02	0.009	0.006	0.003	0.005	0.009	0.02	0.03
5.75	0.03	0.005	0.005	0.005	0.007	0.007	0.007	0.007	0.02	0.007	0.005	0.002	0.004	0.006	0.02	0.02
6.75	0.02	0.005	0.004	0.004	0.005	0.006	0.005	0.006	0.01	0.006	0.004	0.002	0.003	0.005	0.01	0.02

NOTE: _____ Pile contribution \sim background.
 ----- Pile contribution \sim 10% of background.

from the tailings pile in all but 4 sectors. Radon concentration decreases to 10% of background in all directions at about 4 miles from the pile.

A comparison of calculated and measured annual ^{222}Rn concentrations is shown in Table 14. Good agreement was obtained at all points except Shearer and Sill's⁽²⁶⁾ station number 83. The sampling location map given by Shearer and Sill⁽²⁶⁾ indicates that this station was located on 3300 South across the street from the tailings pile.

Atmospheric dispersion of ^{222}Rn attributable to the tailings pile was calculated for a distance of 7 miles from the center of the pile in 16 compass directions. The area included in this 7 mile radius is shown in Fig. 13. Extrapolations beyond this distance involve concentrations approaching 1% of background and would affect a small percentage of the metropolitan population. Furthermore, the presence of mountain ranges at distances beyond 7 miles would invalidate the turbulent diffusion model used to perform the calculations.

It was assumed that no ^{222}Rn was lost between the source and the receptor by decay, rainout, or dry deposition. Thus, the concentrations shown in Table 13 are conservative estimates of annual averages.

Concentrations of Airborne Radionuclides

Results of an air sampling program conducted by the USEPA are given in Table 15. The average concentration was determined from 26 consecutive samples of two weeks duration each. The air sampler was located at the Suburban Sewage Treatment Plant which is located about 0.1 mile north of the largest tailings area. The STAR wind profile analysis⁽⁵⁸⁾ indicates that winds with velocity >18 mph (>41 m/sec)

Table 14. Comparison of calculated and measured ^{222}Rn concentrations in Salt Lake City

Station number	Measured ^a (pCi/liter)	Calculated ^b (pCi/liter)
83	0.43 ± 0.31	2.6 ± 0.4
84	0.39 ± 0.24	0.59 ± 0.26
85	0.68 ± 0.30	0.84 ± 0.27
86	0.28 ± 0.14	0.51 ± 0.26
91	0.44 ± 0.30	0.53 ± 0.26
92	0.42 ± 0.24	0.51 ± 0.26
93	0.24 ± 0.12	0.45 ± 0.26
94	0.44 ± 0.26	0.53 ± 0.26
95	0.29 ± 0.16	0.41 ± 0.26
96	0.22 ± 0.12	0.41 ± 0.26

^aRef. 26.

^bThis value represents the calculated concentration plus the background of 0.38 pCi/liter as observed by Sill and Shearer.(26)

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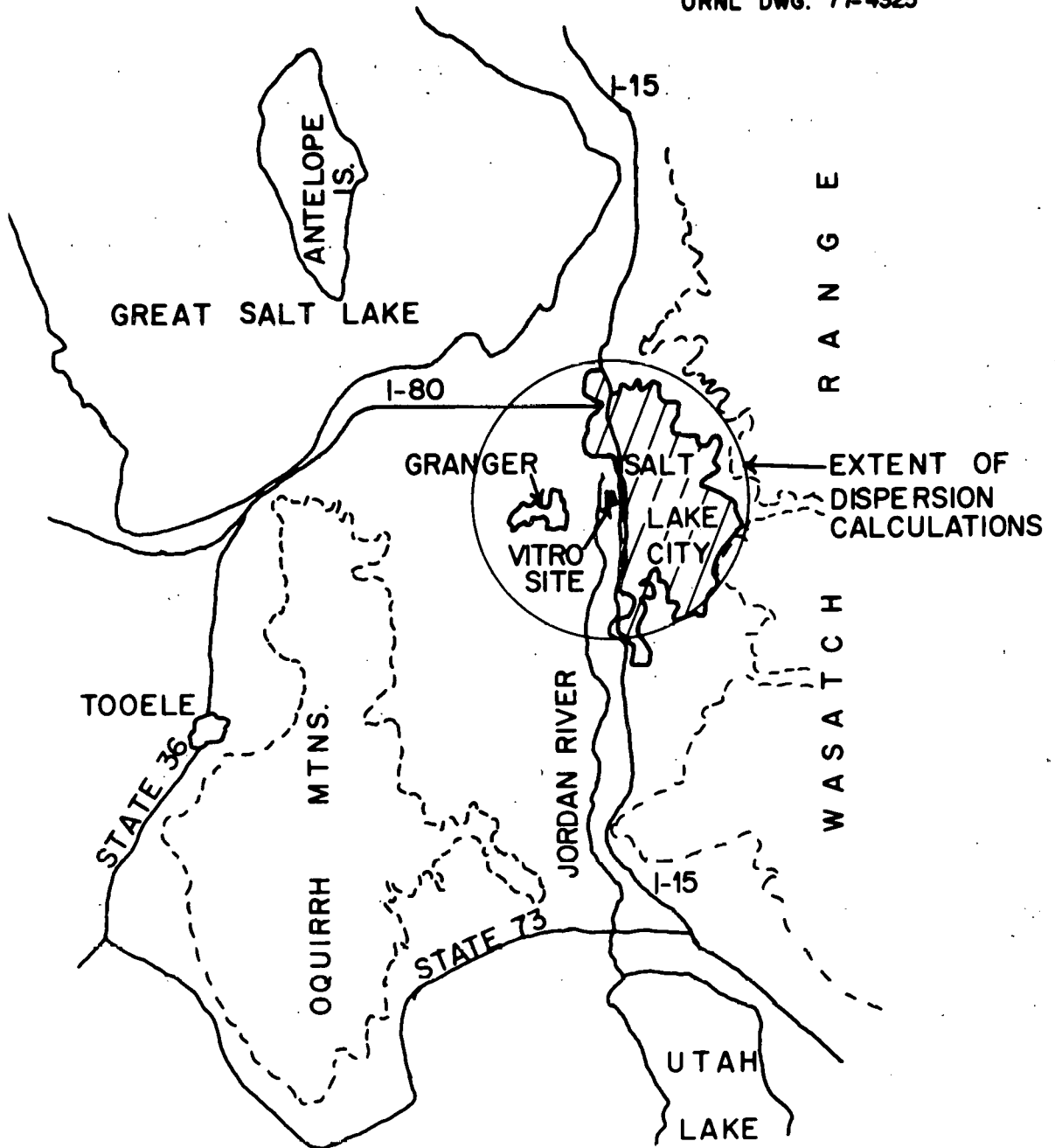


Fig. 13. Map of Salt Lake Valley showing area included in calculations of radon dispersion.

Table 15. Average concentrations of airborne radionuclides at Suburban Sewage Treatment Plant, Salt Lake City, Utah

Radionuclide	Av. conc. ^a (pCi/m ³)	MPC _a ^{bc} (pCi/m ³)	Av. conc./MPC _a
²²⁶ Ra	0.0018 ± 0.0026	2. (I)	0.001
²³⁰ Th	0.00088 ± 0.0012	0.08 (S)	0.011
²¹⁰ Pb	0.010 ± 0.0056	4. (S)	0.003
²¹⁰ Po	0.061 ± 0.166	7. (I)	0.009
²³⁴ U	0.00022 ± 0.00012	4. (I)	0.00006
²³⁵ U	0.000025 ± 0.000024	4. (I)	0.000006
²³⁸ U	0.00020 ± 0.00014	3. (S)	0.00007
²³² Th	0.000064 ± 0.000102	1. (S, I)	0.00007
²²⁸ Ra	0.002 ± 0.002	1. (I)	<u>0.002</u>
		Σ Ci/MPC _i	0.026
²²² Rn (26)	4200 ± 1400	3000 (S)	1.4 ± 0.5

^aRef. 35.

^bThe most restrictive MPC_a is given; S = Soluble, I = Insoluble.

^cRef. 37.

occur 1.3% of the time. These higher velocity winds suspend particulate activity much more effectively than low velocity winds. Almost 80% of >18 mph winds blow toward N, NNW, or NNE directions. Thus, samples collected at this location should be representative of airborne activity concentrations attributable to the tailings pile.

The total average annual airborne activity is less than 3% of the MPC_a as is shown in Table 15. This activity may be compared with the measured ^{222}Rn concentration⁽²⁶⁾ and its error shown in Table 15. The contribution of airborne activity is far less than the error of the radon activity. Furthermore, application of the criteria contained in the Note to Appendix B, 10 CFR 20,⁽³⁶⁾ reveals that all airborne activity except ^{222}Rn may be administratively considered as "not present." All of these measurements were made in a non-habitable location. Atmospheric dispersion of the airborne activity would further reduce the concentrations of radionuclides involved.

6. HEALTH EFFECTS ATTRIBUTABLE TO THE VITRO TAILINGS PILE

The results of this investigation may be used to estimate potential health effects resulting from the Vitro tailings pile. This chapter presents the methods used for calculating health effects and the results obtained.

Radon Diffusion Pathway

Maximum exposure to ^{222}Rn progeny will occur in poorly ventilated structures. For purposes of these calculations, we assume one air exchange per hour⁽⁵⁹⁾ and that the rate of molecular diffusion of radon from outside air into a structure is insignificant compared to that from ventilation. Radon in the outside air arising from the tailings pile is assumed to enter a structure at the rate of $C_o \lambda_v$ (pCi/liter-sec), where C_o is the outside radon concentration and λ_v is the ventilation rate. Radon inside the structure is removed by ventilation and decay; so that the concentration is decreased at a rate of $C_i(\lambda_v + \lambda_R)$ (pCi/liter-sec), where λ_R is the radiological decay rate and C_i is the concentration of radon inside the structure. Over a long period of time, equilibrium will be established such that

$$C_o \lambda_v = C_i (\lambda_v + \lambda_R) , \quad (7)$$

since $\lambda_v \gg \lambda_R$

$$C_o \approx C_i . \quad (8)$$

Thus the indoor concentration of ^{222}Rn attributable to the tailings pile will approach the annual average outdoor ^{222}Rn concentration resulting from atmospheric dispersion.

Concentrations of airborne ^{222}Rn progeny can be calculated using formulas relating these values to ventilation rate.⁽⁶⁰⁾ For a ventilation rate of 1 hr^{-1} , an indoor concentration of 1 pCi/liter of ^{222}Rn will produce 0.005 WL of progeny and continuous exposure will result in an annual exposure of 0.25 WLM. Therefore, we will assume that an outdoor concentration of 1 pCi/liter ^{222}Rn attributable to the tailings pile will result in an annual exposure of 0.25 WLM inside a structure.

Annual average ^{222}Rn progeny exposures may be calculated at each point listed in Table 13 by multiplying the values in this table by 0.25. Thus, maximum exposure of 1.1 WLM/year would occur inside a structure at the north-northwest edge of the tailings pile. In contrast, the lower limit of the Surgeon General's Action Guide, 0.01 WL, would result in an annual exposure of 0.5 WLM using the continuous occupancy assumptions. Furthermore, an annual outdoor exposure at this point would be about 0.1 WLM. This value is based on average wind speed in the NNW direction, distance from the center of the pile, time of transport from source to receptor, and rate of growth of radon decay products during transport. The resulting value of 0.1 WLM is for continuous exposure at an outside working level concentration of 0.002 WL.

A summary of estimated indoor exposures to ^{222}Rn progeny attributable to the Vitro tailings pile is given in Table 16. Population exposures were calculated for each of the 16 compass directions within each radial sector and summed to provide the estimate for that radial sector. Thus, total population exposure of 2000 person-WLM/year represents a best estimate obtainable. It is only in the first sector (up

Table 16. Summary of ^{222}Rn progeny exposures attributable to Vitro tailings pile

Distance from center of tailings pile (miles)	Maximum annual average exposure ^a (WLM/year)	Minimum annual average exposure ^b (WLM/year)	Average annual exposure ^c (WLM/year)	Population ^d	Population exposure ^e (person-WLM/year)
0 - 1	0.35	0.053	0.15	1,513	213
1 - 2	0.066	0.0062	0.025	19,792	341
2 - 3	0.027	0.0024	0.0098	60,047	448
3 - 4	0.015	0.0014	0.0056	72,585	358
4 - 5	0.0099	0.00091	0.0037	97,365	281
5 - 6	0.0072	0.00067	0.0027	102,613	240
6 - 7	0.0056	0.00051	0.0021	44,592	118
0 - 7			0.0061	398,513	1,999
Background ^f			0.095 ± 0.065	398,513	38,000 ± 26,000

^aMaximum exposures occur in the north or north-northwest direction from the tailings pile.

^bMinimum exposures occur in the west-southwest direction from the tailings pile.

^cAverage of all directions.

^dRef. 61.

^eObtained for each direction and summed for each distance from tailings pile. This may not be equivalent to the product of third and fourth columns because of population distribution.

^fCalculated from average of 0.38 pCi/liter with standard deviation of 0.26 pCi/liter as reported in ref. 26.

to 0.75 mile from the edge of the pile) that annual average exposure exceeds the error associated with exposure due to background ^{222}Rn levels.

Exposure summaries for two separate population projections⁽⁴⁾ are shown in Tables 17 and 18. These exposures considered a maximum, uniform population distribution for medium (12,000 persons/sq mile) and high (~20,000 persons/sq mile) density urban growth. In these two exposure projections, it was assumed that the only remedial action was prohibition of public access to the tailings area.

Health effects attributable to these ^{222}Rn progeny exposures were estimated using data provided in the BEIR Report.^(62,63) An absolute risk estimator for death due to lung cancer was derived on the basis of a life-time duration of risk. The footnote of Table 3-2 of the BEIR Report⁽⁶³⁾ indicates that lung cancer accounts for 26% of the absolute risk of "5 deaths/ 10^6 /year/rem" for those over age 9 in the population. It was assumed that this ratio of 0.26 was also valid for individuals aged 0 to 9 years. Table 3-1 of the BEIR Report shows that all cancers account for 1410 deaths among persons in a population of 198×10^6 exposed to 0.1 rem/year at all times after birth. Deaths from in-utero exposure were not included in our estimate because the short half-lives of ^{222}Rn progeny, the low penetrating power of alpha particles and the exposure route of inhalation preclude any significant in-utero exposure to airborne ^{222}Rn progeny. Lung cancer death risk is estimated as $0.26 \times 1410 \div 197.9 \div 0.1$, or 18 deaths/year in a population of 10^6 exposed to 1 rem/year postpartum.

Table 17. Extrapolated ^{222}Rn exposures attributable to Vitro tailings pile for medium density population projection

Sector	Population ^a	Population exposure (person-WLM/year)
0-1	40,000	6,000
1-2	110,000	2,750
2-3	190,000	1,860
3-4	260,000	1,460
4-5	340,000	1,260
5-6	420,000	1,130
6-7	490,000	1,030
0-7	1,850,000	15,500
Background ⁽²⁶⁾		176,000 \pm 120,000

^aRef. 4.

Table 18. Extrapolated ^{222}Rn exposures attributable to Vitro tailings pile for high density population projection

Distance (miles)	Population ^a	Population exposures (person-WLM/year)
0-1	60,000	9,000
1-2	180,000	4,500
2-3	310,000	3,000
3-4	430,000	2,400
4-5	550,000	2,000
5-6	670,000	1,800
6-7	800,000	1,700
0-7	3,000,000	24,400
Background ⁽²⁶⁾		285,000 \pm 195,000

^aRef. 4.

The authors of the BEIR Report considered inherent inadequacies in estimating absorbed dose in rads from a given exposure in WLM and made the following statement (p. 148):

"On the basis of the present evidence, 1 rad/WLM is probably close to the upper limit for a reasonably uniform dose to the basal cell layer of the epithelium of the larger bronchi on a probabilistic basis. In the presence of existing chronic bronchitis, the dose factor may well be substantially lower, owing to increased thickness of the mucous layer as well as of the epithelium, and thus a figure of 0.5 rad/WLM has been adopted for this report. It should be emphasized that uncertainties in this value are probably greater than for the working level measurements themselves in determining risks per rad for the mining populations."

On the basis of this discussion, we estimated that an exposure of 1 WLM/year could be considered, in a very conservative manner, to result in a lung dose of 10 rems/year to members of the general population. Thus, continuous exposure to 1 WLM/year will result in approximately 180 deaths per year per 10^6 persons exposed under these conditions. The reader should realize that the absolute risk estimator was obtained from epidemiological studies of uranium and other mineral miners, many of whom were cigarette smokers. Thus, the risk estimator is not precise and should not be regarded as an absolute number.

Health effects which we postulate as attributable to the Vitro tailings pile are given in Table 19. These were calculated from exposures given in Tables 16, 17, and 18. Health effects which may be attributable to background ^{222}Rn progeny levels are shown in Table 19 for comparison. It may be noted that in every case effects due to the tailings pile are about 10% of the standard deviation associated with the background estimate.

Table 19. A comparison of potential health effects attributable to ^{222}Rn progeny exposure from the Vitro tailings pile and from background from the population residing continuously within 7 miles of the Vitro tailings pile

Population	Lung cancer deaths/year in the population attributable to:	
	Vitro tailings pile	Background ^a (0.38 ± 0.26 pCi/liter)
Present (400,000)	0.4	7 \pm 5
Extrapolated (1,850,000)	3.	32 \pm 22
Extrapolated (3,000,000)	4.	51 \pm 35

^aRef. 26.

Exposure to Gamma Radiation

The plot of gamma exposure as a function of distance from the Vitro pile given in Fig. 12 was used to estimate the annual exposure shown in Table 20. In the preparation of this table, it was assumed that 1R in air is equivalent to 1 rem in tissue. An absolute risk estimator for death from all cancers, including leukemia, as given in Table 3-1 of the BEIR Report was equated to 101 deaths per year for a continuous radiation dose equivalent of 1 rem/year to a population of 10^6 .

An average background terrestrial gamma exposure of $8 \pm 2 \mu\text{R/hr}$ was measured at the sampling locations shown in Fig. 8. Although net gamma exposures attributable to the Vitro tailing pile could not be measured or interpolated accurately beyond 2 miles, a population exposure to 7 miles was conservatively estimated to be 500 person-rem/year.

Potential health effects due to pile gamma rays were estimated for the present population distribution. It was assumed that individuals in the population were subject to continuous outdoor exposure to gamma rays. This results in a conservative estimate but represents, at the most, about 10% of the potential effects due to radon diffusion. However, the estimate of effects due to radon progeny assumed continuous indoor exposure in a poorly ventilated structure. Because the same population cannot be outdoors and indoors at the same time, the effects of gamma exposure are calculated only to provide a perspective for ^{222}Rn exposure and should not be construed as being additive to the potential effects due to exposure to ^{222}Rn daughters.

Table 20. Potential health effects attributable to external gamma irradiation resulting from the Vitro tailings pile

Sector	Exposure (μ R/hr)	Exposure (mR/year)	Population exposure (person-rem/year)	Effects (deaths/year)
0-1	12	105	160	
1-2	1	9	180	
2-3	<1			
0-7			500	0.05
Background	8 ± 2	70 ± 17	$28,000 \pm 7,000$	2.8 ± 0.7

Health Effects Attributable to Other Pathways

Exposures attributable to other pathways have been considered in Chapters 3 and 5. Such potential exposures of the population are probably inconsequential when compared to ^{222}Rn diffusion.

7. SUMMARY AND CONCLUSIONS

An assessment was made of the radiological hazards associated with the inactive uranium tailings pile in Salt Lake City, Utah. An evaluation was made of various pathways by which man is exposed to radioactivity from the tailings. Pathways considered were: 1) radon diffusion from the pile, 2) radioactivity which becomes airborne due to wind, 3) radioactivity which is ingested through the intake of food grown on contaminated land, 4) radioactivity which enters the body directly or indirectly through aquatic paths, and 5) exposure to direct gamma radiation.

Measurements were made to determine the extent to which radioactivity had spread to areas away from the site and to evaluate existing exposure conditions. Radium contamination was found down to a depth of three feet below the tailings; however, due to the error associated with sampling techniques and the possibility of some degree of cross contamination in regions where samples of soil were collected, the actual downward migration of radium and other nuclides was probably less than indicated by radioactivity profiles. Soil samples were taken on the surface and at a depth of one foot at 60 locations outside of the perimeter of the mill property. Radioactivity had been blown or otherwise moved from the pile out to a distance of several hundred meters (tailings have been moved for private use to distances of several miles from the site). Surface samples collected on the west side of 900 West Street had activities ranging from less than 1.0 pCi/g to 34 pCi/g, while samples at a depth of one foot had activities ranging from about 2 pCi/g to 6.6 pCi/g. Surface samples collected on the south side of 3300 South Street

had activities ranging from 1.3 pCi/g to 51 pCi/g, while at a depth of one foot most activities ranged from 1 to 8 pCi/g but in two locations values were higher, 20 and 289 pCi/g, respectively. In areas where tailings had been used in construction at the sewage treatment plant, values as high as 570 pCi/g were found.

Sediments collected in a drainage ditch which runs through the tailings area were found to contain above background levels of ^{226}Ra , ^{238}U , ^{230}Th , and ^{210}Pb . Sediment samples taken from the Jordan River both upstream and downstream from the mouth of this drainage ditch were found to contain radionuclides at approximately background level. However, analysis of a sample collected within 100 m of the discharge into the Jordan River indicated that activity in the sediment was significantly less than in samples closer to the pile. The activity of this sample was on the order of three times background. Samples taken from Mill Creek, which runs along the northern boundary of the mill site, contained low levels of radionuclides. If a remedial action program is conducted at this site, consideration should be given to dredging Mill Creek and the drainage ditch and containing the spoil in order to reduce the concentration of radionuclides in the sediment of those streams.

Except for water which was collected in holes drilled through the tailings, the levels of radioactivity measured were not distinguishable from background.

Short-lived radon daughters were sampled in several structures in the vicinity of the pile. Places were chosen on the basis of 1) being close to the pile but having no tailings in intimate contact with the

structure, 2) having been constructed over a layer of tailings, and 3) having been constructed with tailings close to the structure but with no tailings under the floor. Results for these three situations are presented in Table 11. It should be pointed out that these radon daughter concentrations are representative of short term measurements and extrapolations to annual averages should not be attempted. One should note in Table 11 that the concentration of radon measured in a closed store room of one of the commercial buildings built over tailings (Bld. C, Fig. 11) ranged to about 550 pCi/liter with daughters in approximately 42% equilibrium. In most cases, the ventilation rate of a structure is such that the level of radon in the structure will be significantly reduced for much of the year but during cold months when most structures are closed, radon concentrations should be near maximum, and the degree of equilibrium between radon and its daughters will also be maximum.

This tailings pile represents a large, steady-state source of radon. It was estimated that the radon source term is 2.6×10^8 pCi/sec. In Chapter 5, a description was given of the technique used to estimate dispersion of radon through the environment. These estimates indicated that radon concentrations had decreased to < 10% of background in all directions at 4 miles from the pile. Calculations were carried out to a distance of 7 miles from the center of the pile. Beyond this distance, concentrations of radon were less than 1% of natural background and the additional population represented only a small fraction of the total within 7 miles. Also the diffusion model used to make these calculations loses validity when air masses reach the mountain range to the east.

Comparisons were made of calculated values of radon concentration as a function of distance from the pile with measurements of radon at several places in the vicinity of the Vitro site. Except for one station, the averages of the calculated and measured values were well within the stated limits of error for both sets of data.

A set of airborne radioactivity measurements made over a period of one year by the EPA were examined carefully in this study. These measurements indicated that average annual airborne activities were less than 3% of maximum permissible concentrations in air (MPC_a) and exposures due to airborne activity were less than errors associated with radon exposure. Furthermore, the presence of all airborne radionuclides except ^{222}Rn could be considered insignificant.

Exposure rates due to gamma rays from the pile were measured out to a distance of 2 miles. These measurements were used to calculate annual exposures in person-rem.

In Chapter 6, it was estimated that the annual incidence of lung cancer due to continuous exposure to the pile in its current condition was 0.4 cases per year. In addition, it was estimated that the mortality rate due to all cancers as a result of exposure to gamma rays from the pile will be 0.05 deaths per year. The health effects due to exposure to airborne activity (resuspended radionuclides) were not estimated in detail because the measured concentration of these nuclides was only 3% of the MPC_a (at a location where ^{222}Rn greatly exceeded its MPC_a) and were less than the error associated with radon activity estimates.

In three commercial structures bordering the tailings pile on 900 West Street, short term measurements were made of external gamma ray

exposure rates and of the concentration of airborne radon daughters.

Based on these short term measurements, and an average annual working time of 2000 hours, gamma ray exposure rates range from 15 to 400 mrem/yr, and radon daughter concentrations range from 0.24 to 83 working level months (WLM). According to information presented in Chapter 6 (page 80), conservative estimates of the potential lung dose to individuals exposed to the above cumulative radon daughter concentrations are 2.4 rems/yr and 830 rems/yr respectively.

It should be stressed that during this survey, time did not permit the determination of average annual radon daughter concentrations in accordance with guidelines recommended by the U. S. Surgeon General, which serve as the basis for the Grand Junction Remedial Action Criteria (10CFR712).⁽⁶⁴⁾ In these criteria, "indoor radon daughter concentration level" is stated to mean "that concentration of radon daughters determined by averaging the results of six air samples each of at least 100 hours duration, and taken at a minimum of four week intervals throughout the year in a habitable area of a structure". The short term samples collected during this study should be supplemented by a series of samples taken in accordance with the method outlined in 10CFR712. Estimates of potential health effects to the working population in these commercial structures should not be made until the average annual radon daughter concentration has been determined.

Estimates of the incidence of health effects are for the tailings pile in its present condition. If the pile were stabilized in place by covering it with earth, airborne activity and terrestrial pathways might be eliminated and the radon emanation pathway would be reduced in proportion

to the thickness of earth cover. This remedial action would not permit construction of residences on the property, and perpetual control would be necessary. Removal of the tailings would accomplish several things: 1) a large in situ source of natural radioactivity would be removed from near the center of a major metropolitan area, 2) concentrations of radon in the vicinity of the pile would be reduced to background, 3) it would be possible to release the property for productive utilization in future years without the need for continued surveillance, and 4) tailings could be moved to a permanent disposal location where the radionuclides involved would be removed from biospheric contact in a practical manner.

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