POLICY ISSUE
NOTATION VOTE

June 22, 2004 SECY-04-0102

FOR: The Commissioners

FROM: Luis A. Reyes
Executive Director for Operations

SUBJECT: RESULTS OF THE STAFF'S EVALUATION OF POTENTIAL DOSES TO THE PUBLIC FROM MATERIAL AT THE KISKI VALLEY WATER POLLUTION CONTROL AUTHORITY SITE IN LEECHBURG, PENNSYLVANIA

PURPOSE:

To obtain Commission approval to take no further decommissioning action at the Kiski Valley Water Pollution Control Authority (KVWPCA) site.

BACKGROUND:

KVWPCA operates a waste water treatment plant in Leechburg, Pennsylvania, about 40 kilometers (25 miles) northeast of Pittsburgh on the flood plain of the Kiskiminetas River. From 1976 to 1993, KVWPCA treated sewage sludge by incineration. KVWPCA disposed of the resulting sewage sludge ash by mixing it with water to form a liquid slurry and pumping this material into an onsite lagoon. Discharges to the lagoon ceased in 1993 and plans for closure were developed in 1994.

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In the course of site closure, the Pennsylvania Department of Environmental Resources notified the U.S. Nuclear Regulatory Commission (NRC) that elevated uranium concentrations had been found in an ash sample. Subsequent analyses revealed that subsurface uranium contamination was present at concentrations of up to 34 becquerels per gram(Bq/g) [923 picocuries per gram(pCi/g)] total uranium, and that the material was enriched to approximately 4 percent uranium-235. Further characterization revealed that the volume of the contaminated ash is approximately 9000 cubic meters (320,000 cubic feet) and that the total uranium inventory is approximately 32-41 gigabecquerels (0.85-1.1 Ci), resulting in an average total uranium concentration of approximately 3.0 Bq/g (80 pCi/g). The contaminated ash is highly heterogeneous and the highest levels of contamination are found over a relatively small area, at a depth of 2 to 3 meters (m) [7 to 10 feet (ft)]. Nuclides other than uranium are also present, but at much lower concentrations. The contamination is believed to have resulted from the re-concentration of uranium-contaminated effluents released from the sanitary sewers and laundry drains of the Babcock & Wilcox (B&W) Apollo facility. There is no evidence suggesting that the discharges from the B&W Apollo facility exceeded permissible levels in effect during operation.

NRC and the Pennsylvania Department of Environmental Protection (PADEP) have held numerous interactions on the decommissioning of the KVWPCA site. During a February 12, 2003, meeting between NRC and PADEP, NRC staff took the position that it is KVWPCA's responsibility to identify an appropriate option for decommissioning the site in accordance with NRC regulations. If KVWPCA's preferred option were to leave the material in place, it would need to perform a dose analysis to demonstrate that this option meets the dose criteria of the License Termination Rule (LTR), 10 CFR Part 20, Subpart E.

Conversely, in an April 23, 2003, letter to KVWPCA, PADEP stated that it believes that "...the chemical constituents of the ash, the construction of the lagoon, the threat of groundwater contamination, the location of the lagoon within the 100 year flood plain of the Kiskimenetas River, and other factors, make closure in place undesirable...because closure in place is neither an appropriate nor lawful option under Pennsylvania law." PADEP classified the material in the ash lagoon as "low level radioactive waste" and stated that "...permanent placement of this ash in the Authority's lagoon would constitute unlawful shallow land burial of low level radioactive waste...(and that) portions of the waste with concentrations of uranium and other radioactive materials exceeding 30 picocuries per gram...(would be placed) under license on order of the Department, pursuant to the Department's authority under the Low Level Waste Act, the Solid Waste Management Act...and the Radiation Protection Act...such waste would, therefore, be prohibited from disposal in a municipal waste landfill under Section 273.201(1)."

Differences in the approaches between the two agencies prompted Senator Rick Santorum (Pennsylvania) to send a letter, dated May 15, 2003, to NRC, requesting that NRC and PADEP develop a single consistent recommendation for treatment of the contaminated ash at KVWPCA. Through a series of PADEP/NRC coordination telephone conferences and monthly coordination reports to PADEP, NRC staff developed its plan to conduct an independent dose assessment for the KVWPCA site.

By letter dated November 7, 2003, NRC staff informed KVWPCA and PADEP that it would be conducting its own dose assessment, and that KVWPCA should not submit its decommissioning plan (DP) until the NRC staff completed its assessment and determined whether submittal of a DP is necessary. The letter also noted that PADEP has taken the

position that under Pennsylvania's Solid Waste Management Act, the ash in the lagoon should be removed and properly disposed of per the Commonwealth's jurisdiction over the material as solid waste. Therefore, the NRC staff's dose assessment would also include potential dose to workers in an excavation scenario.

Contrary to the directions in the staff's November 7, 2003, letter, KVWPCA submitted a DP on April 8, 2004. KVWPCA indicated, that based on its dose analyses, no further remediation activities were necessary to meet the LTR criteria for unrestricted use.

DISCUSSION:

NRC staff conducted dose assessments for a range of potential scenarios consistent with the realistic scenarios approach discussed in SECY-03-0069 "Results of the Licence Termination Rule Analysis," May 2, 2003. These scenarios include a removal scenario, in which the contaminated ash is excavated and removed to an offsite disposal facility, and an onsite no-action scenario, in which the lagoon is abandoned in place with no remedial actions performed. The onsite scenarios included a reasonably foreseeable future land use case and a pair of less likely cases used as assessment tools to bound the uncertainty associated with future land use. In all of the scenarios, doses from the groundwater pathway are expected to be significantly limited by the relatively non-leachable form of uranium in the ash as determined by leaching tests.

A likely disposition of the contaminated ash is that it will be removed, and that the site will continue to be used as a waste water treatment plant. The critical group in the removal scenario is the workers who excavate the contaminated ash and are exposed through inhalation of resuspended fine contaminated ash particles and direct irradiation. In addition, to address the possibility that the ash may be removed to a RCRA-permitted landfill, potential impacts of more aggressive leachate chemistry (low or high pH conditions) on uranium mobility were considered and the range of doses to a hypothetical individual residing near the landfill was qualitatively evaluated.

The total effective dose equivalent to workers who excavate and remove the ash is expected to be approximately 0.15 mSv (15 mrem). As any removal operation would take considerably less than one year, this total effective dose equivalent constitutes the total annual dose in the year of removal. Doses to ash removal workers are dominated by the inhalation of uranium-234 and uranium-238 with a small additional dose from external exposure. Doses to the ash removal workers are limited by the relatively low average concentration of these isotopes, the limited exposure time during excavation of the ash, and the limited respirability of the ash particles.

The potential annual dose to residents near a landfill disposal site was not quantitatively evaluated, but is expected to be bounded by the results of the onsite agricultural and intrusion cases discussed below. Therefore, the dose is expected to remain well below 0.25 mSv (25 mrem). Protection of a future resident would arise from both the relatively non-leachable form of uranium in the ash as determined by leaching tests and the engineered and institutional features for groundwater protection that are required for both municipal and hazardous waste landfills. Although disposal of the ash under alkaline, carbonate-rich conditions may result in higher uranium mobility than predicted in the analysis of the onsite scenarios, these facilities are designed with engineered and institutional features to limit contamination of groundwater.

Three cases of the onsite no-action scenario, in which the ash is assumed to be left in place without any remedial action, were also evaluated. These include a recreational use case, in which the property is converted into a riverside park; an agricultural use case; and an intrusion case, in which it is assumed that a volume of ash is excavated for the construction of a basement and the excavated ash is spread on the land surface. These cases, while less likely, were evaluated because they are useful assessment tools. As they comprise a range of future land use and include all exposure pathways, they can be used to bound other scenarios and, therefore, provide an evaluation of the uncertainty associated with future land use.

In the event that the contaminated ash remains onsite with no remedial action taken, the assumption of a recreational exposure case results in a peak mean annual total effective dose equivalent of approximately 0.01 mSv (1 mrem) over the next few centuries, eventually rising to approximately 0.02 mSv (2 mrem) at 1000 years. This result is approximately an order of magnitude lower than either the agricultural case or the intrusion case because no crop intake is assumed in the recreational case.

The results of analysis of the agricultural case indicate that the peak mean annual total effective dose equivalent within the 1000-year compliance period is predicted to be less than 0.2 mSv (20 mrem) and to occur at 1000 years after the present time. Results of the analysis of the intrusion case indicate that the peak mean annual total effective dose equivalent within the 1000-year compliance period is also expected to be less than 0.2 mSv (20 mrem) and to occur at 1000 years after the present time.

In the agricultural and intrusion case, it was assumed that a member of the critical group would site their well or cultivated field at a random location within the 4000 m^2 (1 acre) site. In the unrealistic case that a farmer were to occupy the site and place a home in the most contaminated 200 m^2 (0.05 acre) area on the site, the peak mean annual total effective dose equivalent would be expected to be slightly above the dose constraint, but well below the public dose limit and thus this scenario is not given further consideration in the staff's evaluation. The staff's detailed dose assessment is attached.

As the site meets NRC criteria for unrestricted release under the scenarios considered, the staff's position is that no further action is necessary regarding decommissioning of the KVWPCA site. In accordance with Part 51 and the National Environmental Policy Act, the staff will prepare an Environmental Assessment (EA) and Federal Register Notice detailing the environmental impacts associated with each of the scenarios. The EA also would evaluate the impact of transporting the material from KVWPCA to a disposal site. The EA would be published for a 30-day comment period. Pending comment resolution, the staff would send a letter to KVWPCA, PADEP, and Senator Santorum detailing the NRC staff's dose assessment and conclusion that the site meets the LTR requirements for both the onsite and removal scenarios, and that therefore, NRC intends to take no further action regarding the site.

CONCLUSION:

The NRC staff's dose assessment indicates that under the likely removal scenario, the dose to the workers who excavate the ash is expected to be approximately 0.15 mSv (15 mrem), and the dose to the individuals residing near the landfill is expected to be well below 0.25 mSv/yr (25 mrem/yr). For the onsite scenario, three cases were evaluated. The patron of a park, in the recreational use case, is expected to receive a dose of approximately 0.02 mSv/yr

(2 mrem/yr). The dose to the less likely resident farmer in both the agricultural case and the intrusion case is expected to be less than 0.20 mSv/yr (20 mrem/yr). Regardless of whether the ash is left in place or excavated pursuant to Pennsylvania State law, the NRC staff concludes that the KVWPCA site is acceptable for unrestricted use, and no further remedial action under NRC authority is required. In addition, the on site scenarios analyzed in the DP submitted by KVWPCA are consistent with, and bounded by, the NRC staff's analyses.

RECOMMENDATION:

The Commission approved the staff's proposed action to prepare and publish an EA and FRN regarding the KVWPCA site. Pending resolution of comments on the EA, the staff would send letters to KVWPCA, PADEP, and Senator Santorum explaining that the KVWPCA site meets the LTR requirements for both onsite and removal scenarios and that the NRC; therefore, intends to take no further action.

COORDINATION:

The Office of the General Counsel has reviewed this paper and has no legal objections.

/RA Martin Virgilio Acting For/

Luis A. Reyes
Executive Director
for Operations

Attachment:
Dose Assessment Related to Kiski
Valley Water Pollution Control Authority
Incinerator Ash Lagoon

OFFICE OF NUCLEAR MATERIALS SAFETY AND SAFEGUARDS
DOSE ASSESSMENT
RELATED TO
KISKI VALLEY WATER POLLUTION CONTROL AUTHORITY
INCINERATOR ASH LAGOON

Executive Summary

The Kiski Valley Water Pollution Control Authority (KVWPCA) operates a waste water treatment plant in Leechburg, Pennsylvania, about 40 km (25 miles) northeast of Pittsburgh. The facility is located on a floodplain adjacent to the Kiskiminetas River. From 1976 to 1993, the KVWPCA treated sewage sludge by incineration. The resulting sewage sludge ash was disposed by mixing it with water to form a liquid slurry and pumping the slurry into an onsite lagoon of approximately 4000 m^2 (1 acre) area and 3 m (10 ft) depth. Discharges to the lagoon ceased in 1993. Ash from current operations is temporarily accumulated onsite and shipped in batches to a municipal waste landfill for final disposal.

In 1994, plans were made to remove the ash from the lagoon. In the course of the site closure, the Pennsylvania Department of Environmental Resources (PADER) notified the U.S. Nuclear Regulatory Commission (NRC) that a uranium concentration above the then-applicable Branch Technical Position* limit (1.1 Bq/g, or 30 pCi/g) for unrestricted disposal had been found in an ash sample. Subsequent analyses revealed that subsurface uranium contamination was present at concentrations of up to 34 Bq/g (923 pCi/g) total uranium, and that the material was enriched to approximately 4% uranium-235. Further characterization revealed that the volume of the contaminated ash is approximately 9000 m^3 (320,000 ft^3) and that the total uranium inventory is approximately 32-41 GBq (0.85-1.1 Ci), resulting in an average total uranium concentration of approximately 3.0 Bq/g (80 pCi/g). Contamination of the ash is highly heterogeneous and the highest levels of contamination are found over a relatively small area at a depth of 2 to 3 m (7 to 10 ft). Approximately 400 m^2 (0.1 acre) contains subsurface peak concentrations of uranium above 18.4 Bq/g (500 pCi/g). Other technogenic nuclides are present, but at much lower concentrations. The contamination is believed to have resulted from the reconcentration of contaminated effluents released from the sanitary sewers and laundry drains of the B&W Apollo Borough facility.

Dose assessments for a range of potential scenarios were carried out consistent with the realistic scenarios approach discussed in SECY-03-0069. These scenarios include both a removal scenario, in which the contaminated ash is excavated and removed to an offsite disposal facility, and an onsite no-action scenario, in which the lagoon is abandoned in place with no remedial actions performed. A suite of different land-use cases were evaluated for the onsite no-action scenario to evaluate the uncertainty in doses arising from future land use.

A likely disposition of the contaminated ash is that that the ash will be removed consistent with the 1994 closure plan for final disposal offsite and that the site will continue to be used as a waste water treatment plant (Allard, 2003; Chester Environmental, 1994). The critical group in the removal scenario is the workers who excavate the contaminated ash and are exposed through inhalation of resuspended fine contaminated ash particles and direct irradiation. In addition, to address the possibility that the ash may be removed to a RCRA-permitted landfill, potential impacts of more aggressive leachate chemistry (low or high pH conditions) on uranium mobility were considered and the range of doses to a hypothetical individual residing near the landfill was qualitatively evaluated. Three cases of the onsite scenario were evaluated. All onsite scenarios were no-action scenarios in which the ash is assumed to be left in place without any remedial action. The onsite scenario includes a recreational use case, in which the property is converted into a riverside park; an agricultural use case; and an intrusion case, in

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* 46FR52061 “Disposal or Onsite Storage of Thorium or Uranium Wastes from Past Operations”, 23 Oct 1981, Option 1.

which it is assumed that a sub-volume of ash is excavated for the construction of a basement and the excavated ash is spread on the land surface. These cases were chosen to represent a range of potential land uses at the site. Agricultural and intrusion cases were evaluated because they comprise all exposure pathways and therefore bound other scenarios.

Analysis of the dose to a worker removing the ash under the removal scenario was performed using hand calculations with Microsoft Excel 2002 and the GNU Octave Version 2.1.36 mathematical modeling software package (Eaton, 2002). Analyses of onsite scenarios were performed using the probabilistic analysis capabilities of RESRAD 6.21 (Yu *et al*., 2001/2002) to evaluate the effect of uncertainties in key parameters and variability in contaminant distributions. Doses associated with the inhalation of radon were not evaluated for any of the scenarios.

The total effective dose equivalent[†] to workers who excavate and remove the ash is expected to be approximately 0.15 mSv (15 mrem). Doses to ash removal workers are dominated by the inhalation of uranium-234 and uranium-238 with a small additional dose from external exposure. Doses to a worker removing the ash are expected to be limited by the relatively low average concentration of these isotopes, the limited exposure time during excavation of the ash, and the limited respirability of the ash particles. Although potential annual doses to residents near a landfill disposal site were not quantitatively evaluated, they are expected to be bounded by the results of the agricultural and intrusion cases discussed below and are therefore expected to remain well below 0.25 mSv (25 mrem). Protection of a future landfill resident would arise from both the relatively unavailable form of uranium in the ash as determined by leaching tests and the engineered and institutional features for groundwater protection that are required for both municipal and hazardous waste landfills. Although disposal of the ash under alkaline, carbonate-rich conditions may result in higher uranium mobility than predicted in the analysis of the onsite scenarios, engineered and institutional features for groundwater protection are expected to be effective at limiting contamination of groundwater at disposal facilities.

In the event that the contaminated ash remains onsite with no remedial action taken, the assumption of a recreational land-use case results in an peak mean annual total effective dose equivalent of approximately 0.01 mSv (1 mrem) over the next few centuries, eventually rising to approximately 0.02 mSv (2 mrem) at 1000 years. This result is approximately an order of magnitude lower than either the agricultural case or the intrusion case because no crop intake is assumed. The results of analysis of the agricultural case indicate that the peak mean annual total effective dose equivalent within the 1000-year compliance period is predicted to be less than 0.2 mSv (20 mrem) and to occur at 1000 years after the present time. The mean dose during the 1000-year compliance period is due primarily to ingestion of crops contaminated by root and foliar uptake of uranium and its decay progeny (protoactinium-231, radium-226, and lead-210). The peak mean dose is associated with erosion of the relatively uncontaminated one meter overburden that provides significant shielding and isolation of plant roots from the

[†] The annual total effective dose equivalent comprises the sum of external irradiation for one year and committed effective dose equivalent arising from intake of contaminated media for the same one year period. Use of the term “dose” in this report indicates, unless otherwise indicated, a total effective dose equivalent. Because the removal operation is expected to take considerably less than one year, the total effective dose equivalent to the worker removing the ash constitutes the total annual dose in the year of removal.

more contaminated buried ash. Results of the analysis of the intrusion case indicate that the peak mean annual total effective dose equivalent within the 1000-year compliance period is also expected to be less than 0.2 mSv (20 mrem) and to occur at 1000 years after the present time. In all of the scenarios including a groundwater pathway, doses are expected to be significantly limited by the relatively unavailable form of uranium in the ash as determined by leaching tests.

In the onsite scenarios, it was assumed that a member of the critical group would site their well or cultivated field at a random location within the 4000 m^2 (1 acre) site. To implement this assumption for the no action case, heterogeneity of the contamination in the ash was modeled with a stochastic distribution of contaminant concentrations that was derived from the spatial distribution of contamination and the assumption that the randomly placed field has an area of 900 m^2 (0.22 acres). The source term geometry for the recreational case was assumed to be identical to that of the no action case. A similar approach was used to evaluate the intrusion scenario, except that it was assumed that a 200 m^2 (0.05 acre) house is placed randomly within the site, that the entire depth of the ash layer below the house is excavated for the construction of a basement, and that the excavated ash is spread on the surface of the site and used for the cultivation of crops. In the very unlikely case that a farmer were to occupy the site, place a home in the most contaminated 200 m^2 (0.05 acre) area on the site, and grow crops in the ash that was displaced by excavation for the construction of a basement, the peak mean annual total effective dose equivalent would be expected to be approximately 0.50 mSv (50 mrem).

1 Site Description

1.1 Facility Description and Process History

The KVWPCA sewage treatment plant is located on 14.6 hectares (36 acres) of former farmland. The property was purchased in 1972 and the sewage treatment plant has been in operation since December 29, 1975. Sludge solids are generated by separation of raw sewage in a primary settling tank and by an activated sludge process. The resulting sludge is then dewatered in a belt filter press. From 1976 to 1993, the KVWPCA treated sewage sludge by incineration in a seven-hearth furnace at 760 to 820 ºC (1400 to 1500 ºF) (Kossak, 2003) and disposed of the resulting sewage sludge ash by mixing it with water to form a liquid slurry and pumping the slurry into the onsite lagoon via a discharge pipe. Discharges to the lagoon ceased in 1993, and a closure plan was prepared in 1994 (Chester Environmental, 1994). The sludge ash from current operations is temporarily accumulated onsite and shipped in batches to a municipal waste landfill for final disposal. The ash lagoon is located in the east end of the KVWPCA property on a flood plain terrace approximately 120 m (400 ft) north and east of the Kiskiminetas River. The lagoon covers an area of approximately 4000 m² and the ash depth ranges from 2 to 3 m (7 to 10 ft). The base of the lagoon is at an elevation of approximately 239 m MSL (783 ft MSL), approximately 4.5 m (14.8 ft) above the mean water level of the river. Although there is not an engineered liner system, the lagoon was excavated into native floodplain silty clay and is surrounded by a 3 m (10 ft) berm.

The enriched uranium is believed to have been introduced by discharges of laundry water and sanitary sewage from the Babcock and Wilcox (B&W) Apollo facility (Chester Environmental, 1997). The B&W Apollo Borough facility was tied into the KVWPCA plant on March 9, 1977 (Williams, 1977) when PADER revoked the B&W permit for discharge into the Kiskiminetas River. In December, 1982, shower drains from the Parks Township Plant were tied into the

KVWPCA (Giotto, 1982). Discharges from the Apollo plant laundry continued until February 8, 1984. Records of the laundry discharges indicate a total discharge of approximately 44 GBq (1.2 Ci) of uranium between 1977 and 1982 (Bores, 1995).

1.2 Geology and Hydrology

In 1992, four monitoring wells were installed around the periphery of the ash lagoon. Boring logs indicate the presence of typical floodplain deposits, with a silty clay unsaturated layer ranging in thickness from 3.5 to 7 m (12 to 23 ft) under the ash lagoon (IT Corp., 2002). Interbedded silty sands, presumably horizontally bedded, are located within the unsaturated zone. Organic layers and fragments and coal fragments were frequently encountered within the unsaturated silty clay layer during the installation of the wells. A sand and gravel saturated zone in hydraulic communication with the river underlies the clay. Although depth to groundwater may fluctuate depending upon the location of the well and the river stage at the time of measurement, groundwater typically is found at depths from 6.0 to 8.0 m (20 to 26 ft) below the ground surface in the sand and gravel layer. Results from monitoring wells indicate that the alluvial aquifer has a saturated thickness ranging between 2 to 3 m (7 to 10 ft). Although the groundwater from the alluvium is potable, the well yield from the monitoring wells is less than 3.8 L/min (1 gpm)[‡], and one well (MW2), which penetrates only a small portion of the alluvial aquifer, failed to recover for sampling on at least two occasions. Typical sources of groundwater for domestic wells in the region rely on the deeper bedrock aquifers (Astwood *et al*., 1997; IT Corp., 2002). The site is located within the 100-year floodplain. Analyses of water levels in the monitoring wells indicate that the water level in the aquifer under the lagoon is affected by the river stage of the Kiskiminetas River. The river is controlled by a series of impoundments constructed between 1942 and 1971 (USGS, 2004). The nearest gauging station on the Kiskiminetas River is located at Vandergrift. Gauge zero at Vandergrift is 235 m (769.4 ft) above sea level (asl), and corresponds to approximately 233 m (765 ft) asl at the site. The annual average flow at Vandergrift is 87.7 m^3/s (3100 ft^3/s), ranging from 50.3 to 128 m^3/s (1780 to 4520 ft^3/s). Flood stage is approximately 7.6 m (25 ft) asl.

1.3 Radiological Characterization

Several studies of the lagoon have been performed. In 1995, the ORISE Environmental Survey and Site Assessment Program carried out a radiological characterization of the lagoon (Payne, 1995). A surface scan and exposure rate survey revealed no areas with exposures rates above the background exposure rate of 2.1 to 3.1 nC/kg (8 to 12 ìR/h). Fifty surface samples were collected from a 10 m by 10 m (32.8 ft by 32.8 ft) grid covering the lagoon, and 25 boreholes were augured to collect samples from the surface, a depth of 1 meter (3.3 ft), and the bottom of the lagoon. In addition, soil and sediment samples from outside the lagoon, as well as water samples from the four monitoring wells, were collected. Five samples of ash from the 1995 site characterization study were selected for more detailed characterization (ESSAP, 1996). The detailed characterization describes the environmental availability of the uranium as determined with batch leaching tests and measurement of the respirable activity in the ash. In June 1998, Koh Associates carried out additional sampling to provide a more complete characterization of the contamination in the lagoon. One hundred eight locations were sampled and 155 samples were analyzed by gamma spectroscopy for ^{235}U, ^{238}U, and ^{60}Co. The results of the 1995

[‡] However, well yields from alluvium can be quite variable depending upon the characteristics of the formation. There is insufficient data to conclusively eliminate the alluvium as a source of potable water for a residential family based only on aquifer yield.

ESSAP survey and the 1998 Koh survey were combined into a geostatistical dataset by the NRC in 1999 (Stirewalt, 2000).

The 9000 m³ (320,000 ft³) lagoon contains approximately 0.031-0.041 Bq (0.85-1.1 Ci) total enriched uranium with an average concentration of 3.0 Bq/g (80 pCi/g) and average isotopic distribution of 79% ^{234}U, 4.0% ^{235}U, and 17% ^{238}U. The highest observed uranium concentration was 34 Bq/g (923 pCi/g) total uranium. Approximately 2000 m³ (22%) of the ash has a uranium concentration exceeding 3.7 Bq/g (100 pCi/g), and approximately 1200 m³ (13%) of the ash has a uranium concentration exceeding 7.4 Bq/g (200 pCi/g) (Figure 1).

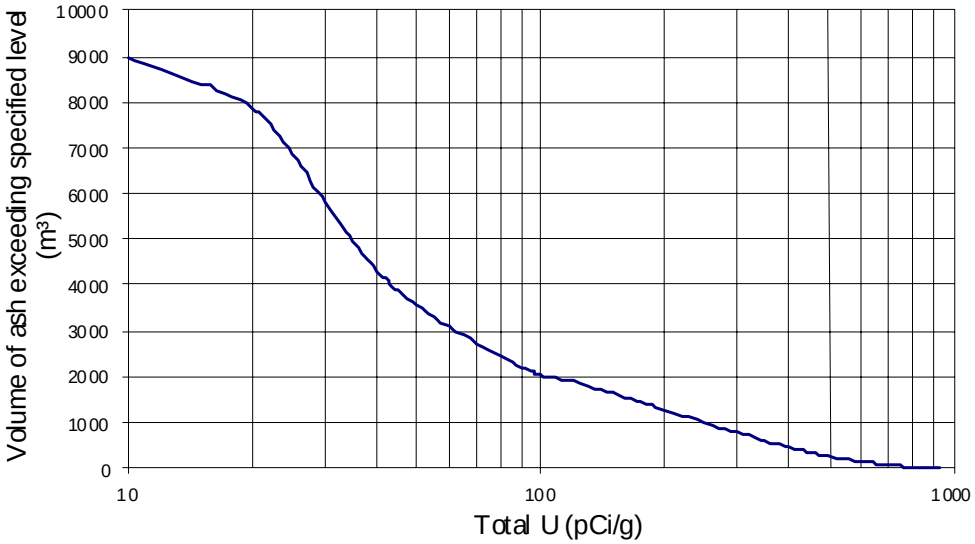

Figure 1: Distribution of Uranium Contamination in the Kiski Valley WPCA ash lagoon

Limited groundwater sampling has been performed. The 1995 ESSAP survey measured gross alpha and beta activity in standing water in the lagoon and in four monitoring wells around the periphery of the ash lagoon. Additional sampling was performed in October 2001 by IT Corporation (Table 1). All of the 2001 IT Corporation samples had less than the minimum detectable activity (MDA).

Table 1: Uranium concentration in water samples collected during site characterization

Location	ESSAP, 1995		IT Corp., 2001		
	Gross Alpha (pCi/L)	Gross Beta (pCi/L)	Gross Alpha (pCi/L)	Gross Beta (pCi/L)	Total U (pCi/L)
Standing water	<2.4	8.4±1.4	Not sampled		
MW 1	<1.6	3.0±1.1	<0.6	<2.2	<0.15
MW 2	3.5±1.1	4.9±1.1	Insufficient recovery		
MW 3	<1.5	2.5±1.0	<0.71	<1.8	<0.16
MW 4	<1.3	<1.6	<0.92, <0.73	<2.0, <1.9	<0.14,0.15

1.4 Geochemical Characterization

An initial characterization of the environmental availability of the uranium was performed by the Environmental Survey and Site Assessment Program (ESSAP) of Oak Ridge Institute for Science and Education (ORISE) (ESSAP, 1996). The Environmental Survey and Site Assessment Program measured readily available uranium in five samples of ash from the KVWPCA ash lagoon (ESSAP, 1996). The extraction protocol recommended by Amonette and co-workers (1994) was used. The results of the analysis indicated that only approximately 3% of the uranium in the ash was in a form that would be readily available for transport out of the ash. A site-specific distribution coefficient of 1400 mL/g was estimated from the average ratio of leachate uranium concentration to solid sample uranium concentration and was used in a previous dose analysis conducted by the NRC (NRC 1996).

Initial sensitivity analyses indicated that the dose was sensitive to the distribution coefficient of uranium in the contaminated ash. Because the chemical characteristics of the ash are different than typical soil characteristics, the release of uranium from the ash under different chemical conditions was unknown. To eliminate the resulting uncertainty in the dose, additional leaching tests were performed in 2004 (ESSAP, 2004). Release of uranium from the ash was tested with three different leaching solutions: deionized water that had been pre-equilibrated with uncontaminated ash, acetic acid, and an oxidizing sodium bicarbonate solution. Five ash samples that each had a different uranium activity in the range from 2.7 to 30 Bq/g (74 to 800 pCi/g) were tested with each of the three leaching procedures. Ash samples with different uranium concentrations were tested so that the relationship between uranium in the contaminated ash and uranium in the leachate could be evaluated. Samples with different concentrations were selected based on the results of uranium measurements previously conducted by ORISE (Payne, 1995).

Leaching tests were used because previous measurements (ESSAP, 1996) indicated the uranium in the ash was relatively environmentally unavailable. Because the release of the uranium from the ash rather than the mobility of uranium in the ash layer is expected to control the flux of uranium out of the contaminated layer, it was preferable to test the release of the uranium from the ash rather than to test the mobility of dissolved uranium in the ash. For implementation with RESRAD, uranium release was modeled with a linear distribution coefficient (K_d). Distribution coefficients were determined as the slope of a line through the origin fit to the concentration of uranium left on each of the five ash samples after the extractions as a function of the concentration of uranium in the leachate. Although it is possible that uranium is leached from the ash slowly but moves quickly through the remaining ash layer once released from its original solid form, the use of a K_d value to represent leaching of uranium from the ash is consistent with the implementation of the source term with RESRAD because RESRAD does not take credit for contaminant transit time in the contaminated layer.

Tests with acetic acid and pre-equilibrated deionized water were performed as a series of four batch extractions of 18 h each. Each of these series was followed by a 6 h extraction with an oxidizing bicarbonate buffer. The extractions performed with acetic acid and the bicarbonate buffer were conducted as described in the Readily Available Uranium (RAU) and Slowly Available Uranium (SAU) protocols, respectively, described in NUREG/CR-6232 (Amonette *et al.*, 1994). The extraction with pre-equilibrated deionized water was conducted with the same procedure as the RAU test, with the exception of the extractant used.

The extractant believed to be most representative of water infiltrating through the ash lagoon was the pre-equilibrated deionized water. The water was pre-equilibrated with uncontaminated ash before being used as an extractant with contaminated ash samples so that it would better represent the chemical composition of water that had infiltrated through the top of the ash layer. For implementation with RESRAD, a Kd of 18,800 mL/g was determined from results of the extractions with pre-equilibrated deionized water. Extractions performed with pre-equilibrated deionized water were essentially similar to the standard leaching tests ASTM D 3987-85 and ASTM D 4793-93 (*i.e*., 18 h batch extractions with a 20:1 water to solid ratio), which are suggested methods for predicting soil pore water contaminant concentrations (Amonette *et al.*, 1994).

The fraction of uranium in the ash that would be expected to be readily available for transport out of the ash was tested with an acetic acid extraction (Amonette *et al*., 1994).
The results of this test also were used to predict the extent of leaching that could be expected in an acidic environment, such as a municipal waste landfill. In agreement with the results of the previous measurement of readily available uranium (RAU) (ESSAP, 1996), the results of the RAU test indicated that only approximately 3% of the uranium was in a form that could be readily available to transport from the ash. For evaluation of a hypothetical case in which the ash was disposed of in a municipal landfill, a K_d of 1790 mL/g was determined from the results of the RAU test. The RAU test was repeated in 2004 because the 1996 RAU test only included samples in a narrow concentration range and the results were difficult to interpret because they did not indicate a clear relationship between leached uranium and uranium in the solid samples.

The fraction of uranium that could become environmentally available over an extended period of time was evaluated with a test of Slowly Available Uranium (SAU) (Amonette *et al*., 1994). The SAU test, performed with an oxidizing bicarbonate buffer, also indicates the extent of leaching that could be expected in a basic, carbonate-rich environment. The SAU extraction is aggressive because U (VI) forms stable complexes with carbonate. Because the concentration of carbonate species in the leaching solution (0.1 M) is much higher than the concentration expected in infiltrating water in the ash lagoon, the results from the SAU tests were interpreted not as a likely partitioning coefficient but as a bounding value representing leaching under aggressive chemical conditions. An extraction with the bicarbonate buffer solution was conducted on samples used in the acetic acid leach test and on samples used in the water leach test. The results of the SAU test performed after acetic acid extraction indicated that approximately 21% of the uranium in the ash would be expected to become environmentally available over an extended period of time. The distribution coefficient calculated from the results of the SAU extraction conducted with samples used in the water leach test was slightly higher (340 mL/g) than the distribution coefficient calculated from the results of the SAU extraction conducted with samples used in the acetic acid extraction (67 mL/g). The distribution coefficients calculated from the results of the alkaline extractions confirmed the higher mobility of uranium in alkaline, carbonate-rich environments as compared to neutral or weakly acidic environments. The release of uranium from the ash under alkaline conditions was considered because water in contact with sewage sludge ash typically develops an elevated pH and because $CaCO_3$ has been observed in samples of sewage sludge ash from other sites (Gray and Penessis, 1972). However, the pH of the water extraction test leachate was approximately 6 (ESSAP, 2004), suggesting that alkaline, carbonate-rich conditions will not evolve in the ash in the lagoon.

In summary, two series of extractions were conducted according to the protocols described in Amonette *et al*. (1994) to test the leachability of uranium from the ash under different chemical conditions. In the first test a sequence of four batch extractions with acetic acid (pH 2.9) was used to extract the readily available uranium and a high pH (8.3) oxidizing sodium bicarbonate solution was then used to extract the slowly available uranium (SAU). The low fraction of readily available uranium (*i.e*., 3%) was in agreement with the results from the previous RAU test (ESSAP, 1996). The results of the SAU test indicated that a limited fraction of the uranium (*i.e*., 21%) would be expected to become environmentally available over an extended period of time. The second series of tests used a synthetic infiltrate (deionized water pre-equilibrated for 18 hours with a low activity ash sample) as the extractant for a sequence of four extractions and used the alkaline SAU extractant on the solid sample left after the water extraction. The results of the water leach test indicated that under expected conditions at the site, the uranium in the ash is expected to be relatively unavailable to environmental transport.

The extractions conducted at pH 8.3 were conducted with an extractant containing 0.1 M sodium bicarbonate which is expected to lower the calculated distribution coefficient significantly as compared to distribution coefficients calculated from the results of extractions performed without added carbonate species.

2 Dose Assessment

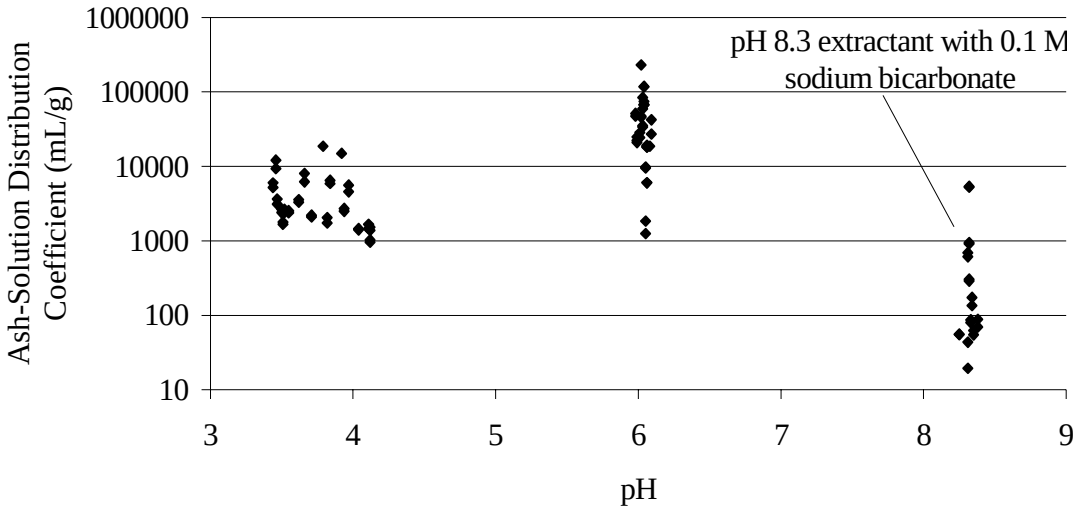

Figure 2: Relationship between leachability-based distribution coefficient and pH for KVWPCA ash (calculated using the final ratio method from results in ESSAP, 2004).

In order to capture an adequate range of future scenarios, dose assessments were performed for both an onsite no action scenario and an ash removal scenario. The onsite scenario was based on an unrestricted release (*i.e*., no credit was taken for institutional measures such as land use restrictions or groundwater monitoring). No remedial action (*e.g*., capping, partial removal of ash to an off-site location) is presumed to be performed in the onsite scenario. The onsite scenario includes a recreational use case, in which the property is converted into a

riverside park; an agricultural use case; and an intrusion case, in which it is assumed that a sub-volume of ash is excavated for the construction of a basement and the excavated ash is spread on the land surface and used for cultivation of crops. In the removal scenario, the ash is assumed to be excavated and removed to a land disposal facility. This represents a likely disposition of the contaminated ash in which the ash is removed consistent with the 1994 closure plan for final disposal offsite and the site will continue to be used as a waste water treatment plant (Allard, 2003; Chester Environmental, 1994).

Analyses of onsite scenarios were carried out using the probabilistic analysis capabilities of RESRAD 6.21 (Yu *et al*., 2001/2002) to evaluate the effect of uncertainties in key parameters and variability in contaminant distributions. Analysis of the removal worker dose scenario was carried out using hand calculations with Microsoft Excel 2002 and the GNU Octave Version 2.1.36 mathematical modeling software package (Eaton, 2002). Compliance is based on the peak value of the mean dose. Supplemental computations were carried out to ensure that this approach did not result in an unintended risk dilution.

2.1 Onsite (No-Action) Scenarios

The recreational case consists of abandonment in place and subsequent conversion of the property currently occupied by the sewage treatment plant into a riverside park. The agricultural case consists of abandonment in place and subsequent occupancy by a farmer. The intrusion case consists of abandonment in place followed by excavation of a sub-volume of ash for the construction of a basement, spreading of the excavated ash on the land surface, and raising of crops in the contaminated spoil. The recreational case is expected to be a reasonably foreseeable future land use. It is not possible to determine when, over the next millennium, agricultural use of the site could become plausible. Sufficient conditions for the plausibility of agricultural uses would include 1) loss of the knowledge that the site had been used for disposal of waste material; 2) slow weathering of the ash that would render it sufficiently indistinct from surrounding soils such that a future occupant would not recognize the nature of the material, and 3) suitability of the ash as a media for raising crops. Because staff did not have any information to make judgments on any of these conditions, agricultural and intrusion cases were constructed as an assessment tool to help bound the uncertainty associated with future land uses. Because these cases include all exposure pathways, they bound other scenarios. Alternate cases can be constructed by taking an appropriate subset of the exposure pathways for these two cases. For example, a residential gardener case could exclude the exposure pathways associated with animal husbandry and reduce consumption fractions of contaminated crops to reflect the more limited intake of home-grown produce. A light industrial scenario could exclude all agricultural pathways. In this analysis, the recreational case has been constructed by suppressing all agricultural and groundwater pathways and reviewing the parameters for the remaining pathways for applicability.

The three onsite scenarios have much in common and much of the analytical framework is shared between them. For example, the recreational scenario comprises a subset of exposure pathways of the agricultural case. The intrusion case is identical to the agricultural case in terms of exposure pathways; the difference is an assumption that excavation of a basement alters the contaminant geometry by bringing contamination to the surface. Because of the extensive common elements of these cases, they are discussed together rather than in three separate, redundant sections. Elements that are specific to a particular case are explicitly identified.

2.1.1 Exposure Pathways

As required under 10 CFR 20.1402, expected doses were evaluated for the average member of the critical group, which is not necessarily the same as the maximally exposed individual. The use of the "average member of the critical group" acknowledges that any hypothetical "individual" used in the dose assessment is based, in some manner, on the statistical results from data gathered from groups of individuals. Calculating the dose to the critical group is intended to bound the individual dose to other possible exposure groups because the critical group is a relatively small group of individuals who, due to their habits, actions, and characteristics, could receive among the highest potential dose at some time in the future. It is unlikely that any individual would actually receive doses in excess of that calculated for the average member of the critical group.

For the onsite recreational case, the critical group is assumed to be a park patron engaging in such recreational uses as picnics or light athletic activity. The exposure pathways for the park patron include exposure to external irradiation, inhalation of resuspended contaminated dust, and incidental ingestion of contaminated ash. The major difference between the recreational case and the agricultural and intrusion cases is the elimination of all agricultural uses of the site (plant, milk, or meat ingestion) coupled with the elimination of the drinking water pathway. The parameter values for the remaining pathways were reviewed in the process of constructing the recreational case. The most significant parameters for this case are those associated with occupancy. Although there were no data to establish these parameters rigorously, it was determined that an appropriate parameter set for these pathways would not be substantially different from those used in the agricultural or intrusion cases. There are therefore no differences in the input parameters between the recreational and the agricultural or intrusion cases for the external irradiation, inhalation, and soil ingestion pathways. The use of a light industrial scenario, with a critical group comprising workers at the current sewage treatment plant, was also considered. It was determined that the source term geometry, exposure pathways, and exposure parameters would be very similar to those of the recreational user (i.e., no agricultural or drinking water pathways). The doses for this case would be expected to be approximately the same as those for the recreational scenario and were therefore not explicitly evaluated.

The critical group for the onsite agricultural and intrusion cases is defined as a residential farm family that drinks water from a well that is affected by the contaminated ash and consumes agricultural products raised on site. The size of the farm family is equivalent to three adults for purposes of determining the consumption of produce and for determining the necessary aquifer yield for groundwater extraction. Because the contamination is spatially heterogeneous and because compliance is based on the average member of the critical group, a distribution of uranium concentrations was used for the subsurface source term in the agricultural case to reflect the uncertainty in the exact location of a household well or garden (see Section 2.1.2.3). An additional analysis of the intrusion case was performed to bound the mean predicted dose to the critical group in the case that a house is sited in the most contaminated area on the site.

Exposure pathways include ingestion of contaminated groundwater, ingestion of contaminated food, external exposure, inhalation of contaminated dust, and ingestion of contaminated soil. Each of these pathways is included for both simulations (*i.e.*, surface and buried contamination) for the agricultural case. In the analysis of the dose due to the buried contamination in the

onsite scenario intrusion case the external gamma, inhalation, and soil ingestion pathways were excluded, in accord with the recommendation of NUREG/CR-5512 Volume 3 (Beyeler *et al*., 1999). In the analysis of the dose due to ash that had been excavated and spread on the land surface in the intrusion case, the drinking water, milk ingestion, and meat ingestion pathways were excluded. The drinking water pathway was excluded in accord with the recommendation of NUREG/CR-5512 Volume 3. The milk and meat ingestion pathways were excluded for the analysis of the dose due to surface contamination in the intrusion case because the area of the surface layer (700 m^2) is expected to be too small to support livestock and a substantial garden. A summary of the pathways included in each RESRAD simulation is provided in the Appendix.

Neither ingestion of contaminated aquatic food nor inhalation of radon were considered. Ingestion of aquatic food grown in groundwater pumped from the site was not considered because well yields were considered to be insufficient to supply both the needs of a farm family and a fish pond. The presence of enriched uranium gives rise to the potential for doses from radon, because ^{234}U decays to ^{226}Ra via ^{230}Th. However, the radon pathway is highly sensitive to the details of construction of dwellings. It is not possible to predict details of construction at points up to 1000 years in the future. Also, preliminary analysis suggested that ingrowth of radium would not yield radium concentrations above 0.56 Bq/g (15 pCi/g) for at least the next 1000 years. For these reasons, doses due to the inhalation of radon were not included in this analysis.

2.1.2 Source Term and Site Geometry

2.1.2.1 Identification of Contaminants

The major dose-producing nuclides are ^{238}U, ^{235}U, and ^{234}U and their decay progeny. It is assumed that the level of enrichment of the uranium is homogeneous, with a ^{234}U: ^{235}U: ^{238}U activity ratio of 79%:4.0%:17%. This assumption is expected to be slightly conservative because lower activity samples may contain primarily naturally occurring uranium, which has a lower fraction of ^{234}U. Cobalt-60, a technogenic gamma-emitting radionuclide, also is present in the subsurface at the site and is correlated to the presence of elevated total uranium concentrations. Cobalt-60 decays with a 5.27 year half-life and is primarily of concern due to external irradiation. The ratio of cobalt-60 to total uranium was determined to be 52.5 (*i.e*., a uranium concentration of 100 pCi/g was associated with a cobalt-60 concentration of 1.9 pCi/g) (ESSAP, 1996). Cobalt-60 concentrations used in RESRAD simulations were based on the ratio of cobalt-60 to total uranium and were corrected for radioactive decay that has occurred since the time of sampling.

Other radionuclides that were identified in the 1996 ESSAP survey included ^{226}Ra (1.6 pCi/g), $^{228/232}Th$, (3 pCi/g), ^{241}Am (0.55 pCi/g), and ^{137}Cs (0.64 pCi/g). Analysis of these nuclide concentrations did not show a correlation with the elevated uranium concentrations. Because there are other natural or anthropogenic sources of these radionuclides, and because these values were typical of both currently generated ash at the site and from ash across the country (ISCORS, 2003), it was determined that these are present as the result of natural processes and do not represent regulated material. ^{137}Cs concentrations demonstrated a weak correlation with elevated uranium, but screening analyses indicated that introduction of the correlation would not significantly affect the total dose. These nuclides were therefore not included in the dose assessment.

2.1.2.2 Contaminated Zone Geometry

The contaminated zone geometry for both the agricultural case and the recreational case comprises a layer of less-contaminated ash covering a layer of more contaminated ash. The source term for the onsite scenario agricultural case and the recreational case is modeled as a 3 m deep, 4000 m² rectangular area in which the top meter has a uniform uranium concentration equal to the average uranium concentration in the top meter of ash (0.93 Bq/g, or 25 pCi/g) and the bottom 2 m contains a distribution of uranium concentrations reflecting the spatial distribution of uranium at the site. Because RESRAD is not capable of modeling more than one contaminated layer, two RESRAD simulations were performed and their results were combined. Simulation 1 was used to evaluate the potential dose from a 2 m thick contaminated layer covered by a 1 m thick uncontaminated cover layer. Simulation 2 was used to evaluate the potential dose from a 1 m thick slightly contaminated (0.93 Bq/g, or 25 pCi/g) layer without a cover layer. The mean dose results from Simulation 1 were added to the mean dose results from Simulation 2 to determine the combined peak mean dose.

The intrusion case is modeled with a top layer of contaminated ash composed of the excavated ash that has been brought to the surface, a middle layer composed of the less-contaminated ash currently in the top 1 m of the ash lagoon, and a bottom layer composed of the more contaminated ash currently in the bottom 2 m of the lagoon. A procedure was used to develop the source term that was analogous to the procedure used for the agricultural case. It was assumed that a 200 m² (0.05 acre) basement is excavated and the resulting spoil is spread over an area of 700 m² (0.17 acres) at a depth of 0.90 m (3.1 ft). The contamination in this layer was developed as a distribution to represent the uncertainty in the location of the basement excavation. Contaminant concentrations in the layer formed from spreading the excavated ash on the surface are represented with the vertically averaged distribution described below. The buried subsurface layer has an area of 4000 m^2 (1 acre), a thickness of 2 m (6.6 ft), and is covered by a 1 m (3.3 ft) cover layer (the original layer of less contaminated ash currently in the top 1 m of the lagoon). To avoid double counting the dose due to the ash with the highest uranium concentrations (i.e., using the highest uranium concentration for both the surface layer and for the concentration in the buried layer) the uranium concentration in the buried layer was represented with an average value in the intrusion case (3.4 Bq/g or 92.9 pCi/g). The average uranium concentration used to represent uranium in the buried layer in the intrusion case was calculated by dividing the upper bound of the total uranium inventory at the site (41 GBq or 1.1 Ci) by the volume of ash in the bottom 2 m of the lagoon (8000 m^3 or 283,000 ft^3). This procedure results in a slight overestimate of the average uranium concentration in the bottom 2 m of the lagoon but ensures that the uranium in the lightly contaminated upper 1 m of ash is included in the source term. Analysis of the doses due to the lightly contaminated upper 1 m of ash is not included as a separate simulation in the intrusion case because all agricultural activities and external exposure at the site are assumed to occur in the 700 m^2 (0.17 acre) area on which the excavated ash is spread. As in the agricultural case, the predicted dose associated with the intrusion scenario is the sum of the doses resulting from the buried and surface layers of contaminated ash.

2.1.2.3 Analysis of Spatial Variability of Contamination

Uranium contamination in the KVWPCA ash lagoon is highly heterogeneous and the most contaminated ash is located at depths of one to two meters. However, the simple models used in the dose assessment require a homogenous soil contamination with a simple vertical

structure (*i.e*., clean cover layer – contaminated layer – unsaturated layers – saturated layer) in order to determine the level of contamination in foodstuffs and well water. Methods for addressing spatial variability in contaminant concentrations for site-specific dose assessments for unrestricted release are discussed in the Consolidated NMSS Decommissioning Guidance Volume 2 (Schmidt *et al*., 2003) Sections I.2.3.2 and I.3.3.5. While the use of area factors to account for the more limited production of crops and livestock on smaller areas are acceptable, the guidance on the use of area factors pertains to cases with homogeneous contamination more clearly than to cases with highly heterogeneous contamination. In this dose assessment, usage factors for agricultural pathways were set to 100% in accord with the recommendation in NUREG/CR-5512 Volume 3 (Beyeler *et al*., 1999). This represents a potentially significant conservatism in the analysis.

The effect of spatial variability is included in this analysis by assuming that there is an uncertainty about the exact location of a field, a well, or a basement excavation. It is assumed that these are located somewhere on the 4000 m^2 (1 acre) site. The selection of a single soil concentration to represent heterogeneous contaminant concentrations requires consideration of the appropriate averaging area over which sample heterogeneity may be neglected. As indicated by Schmidt and others (2003), this is a function of the pathway under consideration. Because the averaging area may be considerably less than the area of the site, there may be several representative areas within a site. If the representative area is considerably less than the area of the site, a degree of uncertainty is introduced into the assessment because of the lack of knowledge of the location of a future agricultural field, well, or basement excavation. The variability in the contaminant levels therefore induces an uncertainty in the average contaminant concentration to which the critical group is exposed. This uncertainty is a function of two factors: the location of the exposure site and the appropriate averaging area. The method used to vertically average the contamination is another source of uncertainty. Doses can vary considerably for sites with the same vertically averaged concentration but with different contaminated zone depths and thicknesses. The detailed method for deriving the distributions to represent these uncertainties is discussed in the following paragraphs.

The primary consideration in determination of an averaging area is the exposure pathway. For drinking water pathways, the selection of an appropriate averaging area to determine an averaged, homogeneous, soil concentration was based on setting the infiltration volume (excluding irrigation) equal to the dilution volume, as defined by Equation 4.14 of NUREG/CR-5512, Volume 1 (Kennedy and Strenge, 1992). No assumptions are made about depth over which the well is screened because the mass balance model was used in the analyses. Water is withdrawn from the well at a rate of 214 m^3 (7,600 ft. ^{3}) per year, considered to be adequate to supply the needs of a farm family based on three times the average per capita annual water use in Pennsylvania (Beyeler *et al*., 1999)[§]. The use of parameters consistent with the site results in an averaging area of 1390 m^2 (0.34 acre). For pathways arising from plant consumption, the area required for crop consumption can be estimated in a similar way. The size of the representative volume for agricultural pathways is set equal to the area required to raise sufficient crops for three adults. The use of the default parameters in RESRAD for crop yield and food consumption (excluding animal pathways, which would require a much larger

[§] The well yields in the alluvial aquifer are sufficiently low (<1 gpm) that it would probably not be an adequate source of irrigation water. The use of a storage tank with an intermittently operated pump would be able to supply 214 m^3 (7,600 ft^3) per year for domestic use.

area) results in an area of 714 m² (0.17 acre). These values are both below the 2000 m² (0.50 acre) averaging area used by RESRAD for crop consumption. The averaging area was therefore set to 900 m² (0.22 acre) to provide a prudently conservative estimate of the appropriate averaging area.

In order to perform the averaging, the vertical heterogeneity of the contaminant profile also was considered. For groundwater pathways, the peak groundwater concentration will reflect the breakthrough of the contaminated water arising from the most contaminated layer. The use of a vertical average therefore would be potentially non-conservative. In order to account for the potential non-conservatisms associated with the use of a vertical average, the distribution used the average taken over the maximum contaminant concentration at each x, y point sampled. In order to simplify the analysis, the same subsurface contaminated zone was used for both the agricultural case and the recreational use case, and the area of the site was conservatively set at 4000 m^2 (1 acre) (i.e., it was assumed that the contamination level represented by a 900 m² (0.22 acre) average extends over an area of 4000 m² (1 acre). This avoided the need to compute a joint distribution representing the negative correlation between the contamination in a particular 900 m^2 (0.22 acre) area and the remaining 3100 m² (0.76 acre) of the site.

In addition to the agricultural case, an intrusion case was considered to evaluate the potential impact of the location of a 200 m^2 (0.05 acre) house on the ash and excavation of a 600 m^3 (21,200 ft^3) sub-volume of the ash for the construction of a basement (see Section J.4 of Schmidt *et al*., 2003). Doses due to two contaminant layers are evaluated in the analysis of the intrusion case. The surface layer is a 700 m^2 (0.17 acre) area, 0.9 m (3.1 ft) deep layer representing the ash that was displaced for the excavation of the basement and spread on the land surrounding the house. In the intrusion case it is assumed that all external exposure and agricultural activities take place in the 700 m^2 (0.17 acre) surface layer. Because a house could be located at any location on the site, the uranium concentration in the surface layer composed of excavated ash is represented by a distribution of uranium concentrations based on the average uranium concentration in 600 m^3 (21,200 ft^3) sub-volumes of ash (Figure 3). Because it is assumed that the ash is mixed when it is excavated, the uranium concentration in the ash is averaged over the 200 m^2 (0.05 acre) area and 3 m (10 ft) depth of the ash layer. In order to assess the dose that would result from the highly unlikely case that the most contaminated ash is excavated for the construction of a basement and that crops are cultivated in the excavated ash, an additional intrusion case was considered in which the concentration of uranium in the surface layer is not represented by a distribution of values but by the uranium concentration in the most contaminated 600 m^3 (21,200 ft^3) sub-volume.

In order to develop distributions of uranium concentrations, the results of the three-dimensional geospatial model (Stirewalt, 2000) were converted to a two-dimensional grid. For the agricultural case, the dimensional reduction was based on taking the maximum concentration in the subsurface at each location. This is conservative in that a) the most contaminated layers do not all occur at the same depth, b) setting the concentration equal to the peak concentration will overestimate the amount of activity available for leaching due to the assumption that the contaminated zone thickness is 2 m (7 ft) rather than 25-75 cm (0.82 to 2.5 ft). At each grid location, a horizontal average was made of an approximately 900 m² (0.22 acre) area centered on the grid point. For the intrusion scenario, the dimensional reduction was based on a vertical average to reflect the dilution resulting from mixing of the cleaner overlying layer with the more contaminated layer at depth, and a smaller area (200 m² or 0.05 acres) was used for determining the horizontal average to reflect the footprint of a typical house. The procedure is

based on that described in section J.4 of Schmidt *et al.* (2003). Credit for edge effects (*e.g.*, averaging over uncontaminated soil for points close to the sides of the lagoon or vertically averaging to the full 3 m (10 ft) depth recommended in section J.4 of Schmidt *et al*. (2003)) was not taken. The distribution of these values, shown in Figure 3, was used to develop an input distribution for RESRAD.

This averaging procedure is appropriate due to the very limited size (approximately 4000 m², or 1 acre) of this site. For sites encompassing large areas, the use of the expected value of a heavy tailed or multi-modal distribution may represent an inappropriate averaging area and in effect be a form of mathematical dilution. A more careful definition of the critical group would be necessary to avoid an inappropriate averaging area. In this case, because the site is only 4000 m^2 (1 acre) in area, such considerations are not as significant and the use of the expected value arising from the use of a distribution of concentrations for both agricultural and drinking water pathways is considered appropriate.

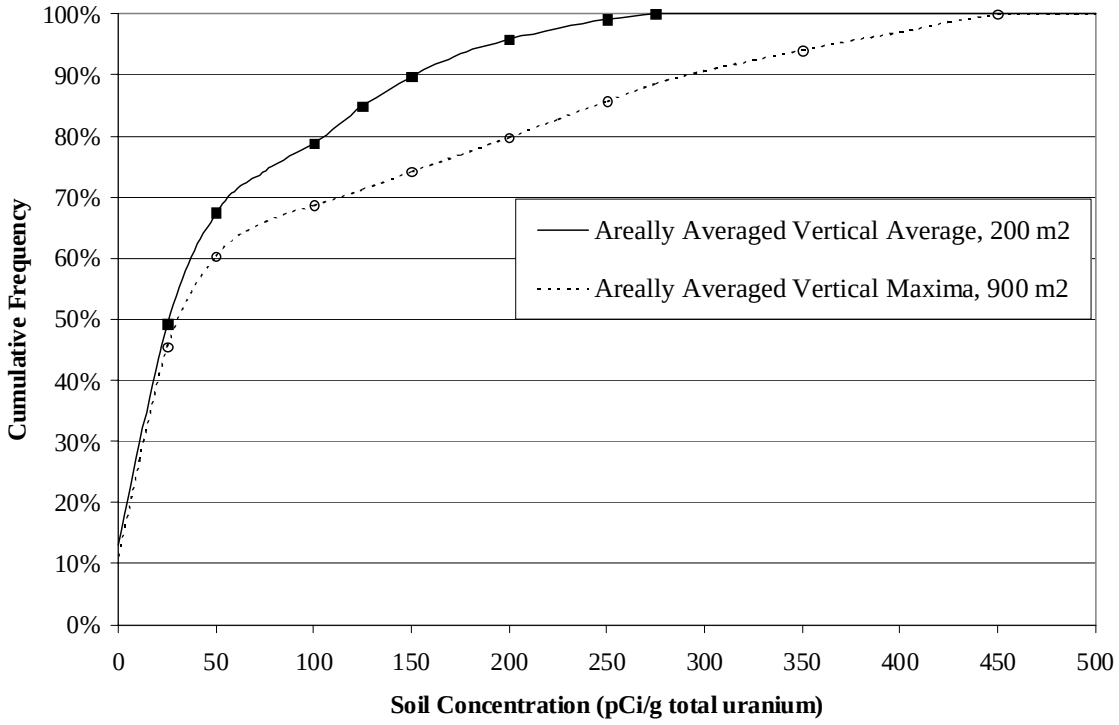

Figure 3: Computed and Approximate Areal Average Distributions of Total Uranium

2.1.2.4 Site Stratigraphy

In all of the onsite scenarios, the contaminated layer is assumed to be underlain by a 3.5 m (11.5 ft) unsaturated layer consisting of silty clay. The unsaturated zone underlying the KVWPCA ash lagoon is a natural alluvial deposit of silty clay with some sand. Because the clay is a natural alluvial deposit, it may contain sand lenses that could result in high conductivity areas in the unsaturated zone. However, because alluvial lenses typically are deposited horizontally, and because a degree of anisotropy would be expected in such a layer, it was considered unlikely that sandy interbeds in the unsaturated zone would create a significant high

permeability channel through the entire 3.5 to 7 m (11.5 to 23 ft) thickness of the unsaturated zone.

2.1.3 Geochemical Considerations

Because the agricultural and intrusion cases include a drinking water pathway, estimation of potential groundwater contamination is necessary. The release of uranium from the contaminated ash to the underlying aquifer was modeled with site-specific parameters calculated from the results of leaching tests performed by the Environmental Survey and Site Assessment Program (ESSAP) of Oak Ridge Institute for Science and Education (ORISE) (see Section 2.4). Uranium mobility in the underlying unsaturated zone was represented with linear distribution coefficients, or K_d values, that were based on literature values. Site-specific soil characteristics were considered in the selection of distribution coefficients to represent uranium mobility in the unsaturated zone.

Distribution coefficients in the ash layer were represented with a triangular distribution. Because deionized water that had been pre-equilibrated with ash was the extractant expected to be most representative of water that infiltrates through the ash lagoon, results from the extractions conducted with pre-equilibrated deionized water were used to establish the most probable value of the distribution (*i.e*., 18,800 mL/g). The most probable value also was used as the maximum of the distribution to avoid an inappropriate risk dilution associated with the use of a triangular distribution. Results from the bicarbonate buffer extraction conducted on samples used in the acetic acid leach test (*i.e.*, 67 mL/g) were used as a lower bound of the triangular K_d distribution. These results were used instead of the results from the bicarbonate buffer extraction performed on samples used in the water leach test (*i.e*., 340 mL/g) because 1) they indicated a clearer relationship between the concentration of uranium extracted and the concentration of uranium in the solid samples and 2) because choosing a lower bound of 67 mL/g included the result of 340 mL/g in the K_d distribution. It is extremely unlikely that chemical conditions in the ash could become more aggressive than the oxidizing bicarbonate buffer extractant.

A distribution of K_d values representing the mobility of uranium in the unsaturated layer underlying the ash lagoon was based on literature values and consideration of site-specific soil conditions. Water chemistry can have a significant effect on uranium mobility in the unsaturated zone. Because the ash in the lagoon was formed by incineration in contact with atmospheric concentrations of oxygen, and because the ash should not contain any organic matter capable of reducing U(VI) to U(IV), uranium in the ash is assumed to be U(VI). U(VI) forms stable aqueous complexes with carbonate anions that can increase its mobility in carbonate-rich environments (see, *e.g*., Davis *et al.*, 2001). The possibility of elevated carbonate concentrations in water infiltrating through the ash layer was considered because water in contact with ash typically develops an elevated pH and because $CaCO_3$ has been observed in samples of sewage sludge ash from other sites (Gray and Penessis, 1972). However, deionized water in contact with samples of ash from the site for 18 h had a pH of 6.0 (ESSAP, 2004). The relatively low pH of deionized water in contact with the ash sample indicates that, although the possibility exists for sewage sludge ash to have an elevated carbonate concentration and pH that could increase uranium mobility, water infiltrating through the ash from the KVWPCA ash lagoon is not expected to have an elevated carbonate concentration or pH.

The distribution coefficient of uranium in the unsaturated layer was represented with a triangular distribution. The minimum value was chosen to represent uranium mobility in a clay layer with a relatively high fraction of sand, but without an interconnected high conductivity pathway through the unsaturated zone. Three K_d values for "clayey sand" (i.e., 58 to 78% sand, 6 to 8% silt, and 19 to 36% clay) from the data compilation of Thibault *et al.* (1990) were averaged to yield a K_d of 680 mL/g (data originally from Neiheisel, 1983). This value is believed to be a conservative lower bound because the unsaturated layer at the KVWPCA is primarily clay whereas the samples of "clayey sand" for which the K_d values were measured were primarily sand. A distribution coefficient of 1600 mL/g was used as the upper end of the distribution to represent uranium adsorption in clay (Sheppard and Thibault, 1990). No independent information was available to develop a central tendency of the distribution, which was estimated as the mean of the upper and lower bounds (*i.e.*, 1140 mL/g).

2.1.4 Selection of Input Parameters

The sensitivity of the predicted dose to the input parameters was tested using the probabilistic features of RESRAD 6.2 with the RESRAD default parameter distributions. Initial sensitivity analyses indicated that the dose was sensitive to the distribution coefficient of uranium isotopes in the contaminated zone. To reduce the uncertainty in this parameter, leaching tests were used to determine the partitioning of uranium in the ash (Section 3.1.3). A range of distribution coefficients for uranium in the unsaturated zone was determined from literature values based on site-specific soil type information.

After ranges were determined for the distribution coefficient of uranium in the contaminated and unsaturated zones, an additional sensitivity analysis was conducted with the new parameter distributions for the uranium distribution coefficients and with default RESRAD distributions for the remaining parameters. The results of the second sensitivity analysis indicated that the dose was most sensitive to the erosion rate of the cover layer, the erosion rate of the contaminated zone, and the thickness of the unsaturated zone. Because no site-specific information was available for the erosion rate of the cover layer or the contaminated zone, the recommended RESRAD probabilistic distributions were used to represent these parameters (Yu *et al*., 1993). The thickness of the unsaturated zone was conservatively chosen at the lower end of the range of unsaturated zone thicknesses recorded in well boring logs from the site (IT Corp., 2002). Because of the importance of the plant and milk ingestion pathways to the predicted dose and the lack of site-specific information about plant and milk transfer factors, distributions were used to represent the plant and milk transfer factors (Yu *et al*., 1993).

Because the dose was relatively insensitive to other parameters, the recommended values in NUREG/CR-5512 Volume 3 (Beyeler *et al*., 1999) or the mean values of distributions recommended in NUREG/CR-5512 Volume 3 were used. In cases in which no parameter recommendation was available in NUREG/CR-5512 Volume 3, the mean of the RESRAD probabilistic distribution (Yu *et al*., 1993) was used. In the few cases in which recommended values were not available in either of these sources, the RESRAD default values were used. The input parameter values and the bases for parameter selection are summarized in the Appendix.

2.1.5 Model Results

The assumption of a recreational exposure scenario results in an peak mean annual total effective dose equivalent of approximately 0.011 mSv (1.1 mrem) over the next few centuries,

eventually rising to approximately 0.017 mSv (1.7 mrem) at 1000 years (Figure 4). This result is approximately an order of magnitude lower than either the agricultural case or the intrusion case and is due to the lack of agricultural activity at the site. The primary difference between the expected dose in the agricultural case and the intrusion case is that the mean dose is slightly higher in the intrusion case at earlier times and slightly higher in the agricultural case at later times (Figure 4). The results of analysis of the agricultural case indicate that the peak mean annual total effective dose equivalent to an average member of the critical group within the 1000-year compliance period is predicted to be approximately 0.18 mSv (18 mrem) and to occur at 1000 years after the present time. Results of the analysis of the intrusion case indicate that the peak mean annual total effective dose equivalent within the 1000-year compliance period is expected to be 0.17 mSv (17 mrem) and to occur at 1000 years after the present time. In both the agricultural case and intrusion cases, the peak mean dose is associated with erosion of the relatively uncontaminated one meter overburden that provides significant shielding and isolation of plant roots from the higher uranium concentrations in the lower 2 m of ash. The mean dose during the 1000-year compliance period is due primarily to ingestion of crops contaminated by root and foliar uptake of uranium and its decay progeny (protoactinium-231, radium-226, and lead-210). Potential groundwater contamination is expected to be limited by the relatively unavailable form of uranium in the ash, evidenced by low fractions of Readily Available (approximately 3%) and Slowly Available (approximately 21%) Uranium in the ash as determined with leaching tests. Doses from buried cobalt-60 are limited because the less-contaminated overlying ash provides significant shielding while the activity rapidly decays over a period of several decades. In the highly unlikely case that a hypothetical future farmer were to site their home in the most contaminated 200 m^2 (0.05 acre) area on the site and grow crops in the ash that was displaced by excavation for the construction of a basement, the peak mean annual dose would be expected to be approximately 0.50 mSv (50 mrem) (Figure 4).

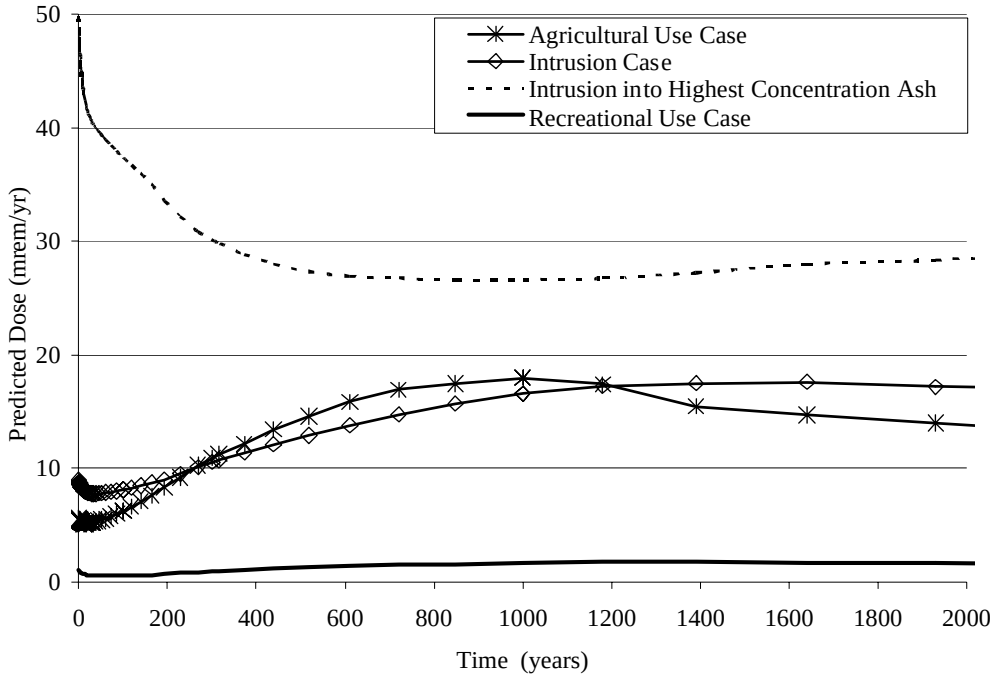

Figure 4: Predicted mean doses for the recreational case, agricultural case, intrusion case, and case in which a house is located on the 200 m^2 (0.05 acre) area with the highest uranium concentration.

2.1.6 Uncertainty Analysis

Because preliminary analyses indicated that contamination of groundwater is the only pathway with the potential to cause doses that significantly exceed 0.25 mSv/yr (25 mrem/yr), laboratory tests were performed to constrain uncertainty in the dose due to uncertainty in the availability of uranium to transport from the contaminated ash (see Section 1.4). Characterization surveys and geospatial modeling provided an adequate characterization of the contaminant distribution in the lagoon. Of the remaining parameters, the one that is expected to have the largest impact on dose is the erosion rate of the ash. Because no site-specific data were available to predict the erosion rate, a distribution of values was used (Yu *et al*., 1993). Because flood plains are expected to experience deposition of sediment rather than a significant amount of erosion, the distribution of values used to represent erosion at the site is expected to be conservative.

In the absence of significant groundwater contamination, future land use is the dominant uncertainty in the dose analysis. It is not possible to predict with confidence the use of the site over the next several centuries. For this reason, different cases were chosen to help bound the uncertainty in potential doses associated with future land uses. One process that was not included in the analysis that may be a source of uncertainty is site flooding. This process was not considered because scenarios in which flooding resulted in increased doses were not considered likely. The most likely outcome of flooding is expected to be a reduction in the dose as compared to the results presented in this analysis because flooding would be expected to deposit additional sediment on the ash, resulting in deeper burial. In the event that extreme flooding resulted in scour in the area of the lagoon, the tremendous dilution and high sediment loads associated with flood stage discharges in the Kiskiminetas River are expected to result in substantial dilution and dispersion of any contaminated ash that might be remobilized and deposited downstream.

2.2 Removal Scenarios

Staff has considered a removal scenario because it represents a likely disposal alternative for the ash at the site. It is being evaluated to ensure that no unacceptable consequences will arise as a result of future excavation and off-site disposal of the ash. Staff considers this scenario to represents a likely disposition of the material at the site because:

1) Representatives of the Pennsylvania Department of Environmental Protection have indicated that the State of Pennsylvania will require removal of the ash under state solid waste legislation (Allard, 2003), and
2) the management of the KVWPCA has indicated that excavation and disposal of the ash at a landfill is their preferred alternative (Chester Environmental, 1994).

Staff has therefore considered the potential impacts of excavation of the ash and disposal in an unspecified landfill. Evaluation of doses to workers during excavation has been performed quantitatively. In conjunction with the evaluation of removal worker doses, exposure of a landfill neighbor to the contaminated ash at a landfill also was evaluated. A formal quantitative assessment of the impact of disposal at a landfill is not possible, because the landfill has not yet been identified. Possible options include an in-state municipal waste landfill (RCRA subtitle D), an out-of state hazardous waste landfill (RCRA subtitle C), or a licensed low-level waste disposal facility. Therefore, the landfill neighbor scenarios were not analyzed quantitatively. However, insight into the potential doses to a resident near a landfill can be gained by evaluating the pathways and parameters used in the onsite scenario and determining the extent to which the doses computed under the onsite agricultural and intrusion scenarios can be considered to bound the doses to a resident near a disposal landfill.

2.2.1 Evaluation of Removal Worker Scenario

The most likely procedure for removing the ash is specified in the 1994 Closure Plan (Chester Environmental, 1994). That document specifies that "ash and visibly impacted soils will be excavated with the aid of a front-end loader...mixed with cement kiln dust or sawdust to stabilize the ash to a minimum of 20% solids…placed into lined 20 ton trailers…and covered prior to transportation". The ash removal worker scenario consists of the exposure of non-radiological workers during ash removal. For purposes of the analysis, it was assumed that a worker is present during the excavation and loading of the ash. The source term consists of the entire ash volume of the landfill. The exposure pathway is inhalation of contaminated ash. A sensitivity analysis has been performed to identify the dose reduction associated with less than complete lagoon excavation.

Mechanical disturbance of the ash during excavation of contaminated material can result in resuspension of fine ash particles. This can result in an enhanced potential for doses due to inhalation of radioactive material. The process for computing doses associated with resuspended surface dust is given in NUREG/CR-5512 (Kennedy and Strenge, 1992) Section 6.3.1. The dose due to inhalation is computed by determining the intake by inhalation and multiplying this intake by an intake-to-dose conversion factor. For this analysis, the dose was computed on a unit basis by determining the committed dose factor associated with handling a cubic meter of ash. The committed dose factor per cubic meter of ash is multiplied by the volume of ash in each contamination fraction and the upper bound of contamination for that contamination interval to yield the total dose, as shown below:

$$\mathrm{CEDE} = \mathrm{IR} \cdot \mathrm{ML} \cdot \mathrm{DF} \cdot \left(\frac{\mathrm{ED}_{\mathrm{Total}}}{\mathrm{V}_{\mathrm{Total}}} \right) \cdot \sum_{C_{soil}} \mathrm{V} \cdot \mathrm{C}_{\mathrm{dust}}$$

where
CEDE: Committed Effective Dose Equivalent (mSv or mrem)
ED: Exposure Duration (hours)
IR: Inhalation Rate (m^3/hr)
ML: Mass loading of respirable particles (g/m^3)
C_{dust}: Concentration of radionuclide in respirable dust (Bq/g or pCi/g)
DF: Dose factor for inhalation (mSv/Bq or mrem/pCi)

For the case in which only a partial excavation takes place, the limits of summation can be changed to account for removal of only part of the ash. The justification for the parameters selected as input are provided below, followed by a presentation of the computational results in tabular and graphical form.

The inhalation rate, based on the ventilation rates of males, is assumed to be 1.7 m^3/hr (Anderson et al. 1985). Values for males were chosen as they have higher ventilation rates (due to greater body mass and lung volume) than females. Males engaged in light activity are reported to breathe at a rate of 0.078-1.7 m^3/hr, with a mean of 0.84 m^3/hr. Males engaged in moderate activity are reported to breathe at a rate of 0.84-4.7 m^3/hr with a mean of 2.5 m^3/hr. We assume that an individual exposed during excavation spends 50% of their eight hour day engaged in light activity and 50% engaged in moderate activity and use the average of the two mean values. This number is therefore slightly higher than the RESRAD default, which is appropriate because this is a worker scenario rather than a residential scenario.

In contrast to residential scenarios with continual exposures, the exposure for the removal scenario lasts only as long as the time required for ash removal. The closure plan (Chester Environmental, 1994, p. 6-2) had estimated 30 days from award of the excavation contract until

removal of the ash was complete. Assuming that each of these days represents one eight-hour work day, this yields a total excavation time of 240 hr, or an excavation rate of approximately 40 m³/hr. In terms of the time required to excavate a cubic meter of ash, this corresponds to 1.5 min/m³ (0.026 hr/m³).

The OSHA limit for respirable inert or nuisance dust is 5 mg/m³. It is presumed that the removal operations will be performed in compliance with OSHA** regulations, thereby limiting respirable dust concentrations to below 5 mg/m³. Oztunali *et al*. (1981) reports mass loadings of 0.6 mg/m³ for construction activities and 0.3 mg/m³ for agricultural activities, approximately ten times lower than the conservative upper bound of 5 mg/m^3 used in this analysis.

The dose conversion factor for inhalation is based on the supplement to Part 1 of ICRP Publication 30 (ICRP, 1979b). Because most of the respirable dust is larger than the 1 micron AMAD used for the standard dose factors in ICRP Publication 30, a size correction factor has been applied to account for a reduced dose associated with greater probability of capturing larger particles in the naso-pharyngeal region and consequent reduced probability of deposition of large, insoluble particles in the lung. The procedure for correcting the DCF to account for larger particle sizes is given in ICRP 30 Part 1 (ICRP, 1979a) Section 5.5.

$$\frac{H_{50}(\mathrm{AMAD})}{H_{50}(1\mu m)} = f_{\mathrm{NP}} \frac{D_{\mathrm{NP}}(\mathrm{AMAD})}{D_{\mathrm{NP}}(1\mu m)} + f_{\mathrm{TB}} \frac{D_{\mathrm{TB}}(\mathrm{AMAD})}{D_{\mathrm{TB}}(1\mu m)} + f_{\mathrm{P}} \frac{D_{\mathrm{P}}(\mathrm{AMAD})}{D_{\mathrm{P}}(1\mu m)}$$

For insoluble uranium of clearance class Y, the lung is the critical organ. For the lung, f_{NP} and f_{TB} are zero and f_P is 100% (ICRP 30, Supplement to Part 1, pp 364, 368, and 378). Values for percent deposition as a function of the airborne median aerodynamic diameter are given in Figure 5.1 of ICRP 30 Part 1. Using the correction factors derived from Figure 5.1 of ICRP 30 Part 1 together with the information on the activity fraction in different size ranges from ESSAP (1996), we obtain the following factors:

Table 2: Computation of Removal Worker Inhalation Dose Conversion Factors

Particle Size Range (microns)	Activity Fraction in Size Range (ESSAP (1996))	Correction Factor (ICRP, 1979a)	Weighted Contribution to Correction Factor
9-10	0.39	0.21	0.083
5.8-9	0.25	0.25	0.063
4.7-5.8	0.21	0.32	0.067
3.3-4.7	0.1	0.38	0.038
2.1-3.3	0.04	0.50	0.020
1.1-2.1	0.01	0.72	0.0072
0.43-1.1	0	1	0

** The limit for airborne uranium (as U) present in the form of soluble compounds is 50 ìg/m³, and for insoluble compounds is 250 ìg/m³. Assuming that all of the respirable ash was contaminated at a level of 2000 pCi/g, the total U concentration in air at a nuisance dust loading of 5 mg/g total dust would be 5 ìg/g, well under the most restrictive OSHA limit for uranium dust. Much higher levels of enriched uranium contamination would be required to violate the OSHA airborne uranium standards.

Summation of the weighted correction factors in the last column yields an overall particle size correction factor of 0.28 for airborne insoluble uranium characterized by the particle size distribution measured in the sludge ash. Therefore, DCF(ash)=0.28DCF(1 ìm).

The inhalation DCF also must be adjusted for the ratio of uranium isotopes in the ash. The activity ratio for the five samples analyzed yield a mean activity ratio of 81% uranium-234, 4.0% uranium-235, and 15% uranium-238. Therefore, inhalation of 1 pCi of U will result in the inhalation of 0.81 pCi uranium-234, 0.04 pCi uranium-235, and 0.15 pCi uranium-238. Adjusting these activities by the respective inhalation dose conversion factors (ICRP, 1979b), the total activity-weighted DCF for a one micron particle is calculated to be 0.129 mrem/pCi. If this DCF is adjusted by the particle size correction factor computed above, an inhalation DCF is given by:

$$0.129 \text{ mrem/pCi} * 0.28 = 0.036 \text{ mrem/pCi } U_{enriched}$$

The results from the ESSAP survey (ESSAP 1995) were evaluated to determine the characteristics of the respirable activity in the ash. The smaller ash particles that constitute the respirable size fraction tend to be characterized by a higher specific activity (Bq U per gram ash) than the bulk ash. This enhancement factor can range from 1.2 to 3.1, as shown in the table below:

Table 3: Relationship between Bulk Specific Activity (SpA) and Specific Activity in the Respirable Fraction

Sample Location	35N15E 150-170	35N15E 200215	30N15E 160175	45N25E 150170	45N05E 115130
Total U in bulk sample, pCi/g	721.7	699.9	833.4	659.4	922.9
Respirable U in sample, pCi	35.86	56.47	36.66	14.41	40.02
Total Respirable activity, pCi	35.9	56.5	36.7	14.4	40.0
Total Respirable mass, mg	28.1	25.8	23.1	11.1	36.2
Total Respirable U, pCi/g	1276.2	2188.8	1587.0	1298.2	1105.5
Ratio of Respirable SpA to Bulk SpA	1.77	3.13	1.90	1.97	1.20

Based on these results, we set the mass activity concentration in the respirable particle size fraction at twice the mass activity concentration in the bulk sample.

The sensitivity of the worker dose to the contaminant concentration removal goal was evaluated (Figure 5). Under the assumptions that the ash can be removed at a rate of 40 m^3/hr, that the total respirable dust load is 5 mg/m^3, and that an exposed worker breathes at a rate of 1.7 m^3/hr during handling operations, the effective dose commitment would be approximately 0.13 mSv (13 mrem) if all of the ash were removed. If only the ash with a concentration of uranium greater than 7.4 Bq/g (200 pCi/g) were to be removed, the total dose would be approximately 0.8 mSv (8 mrem).

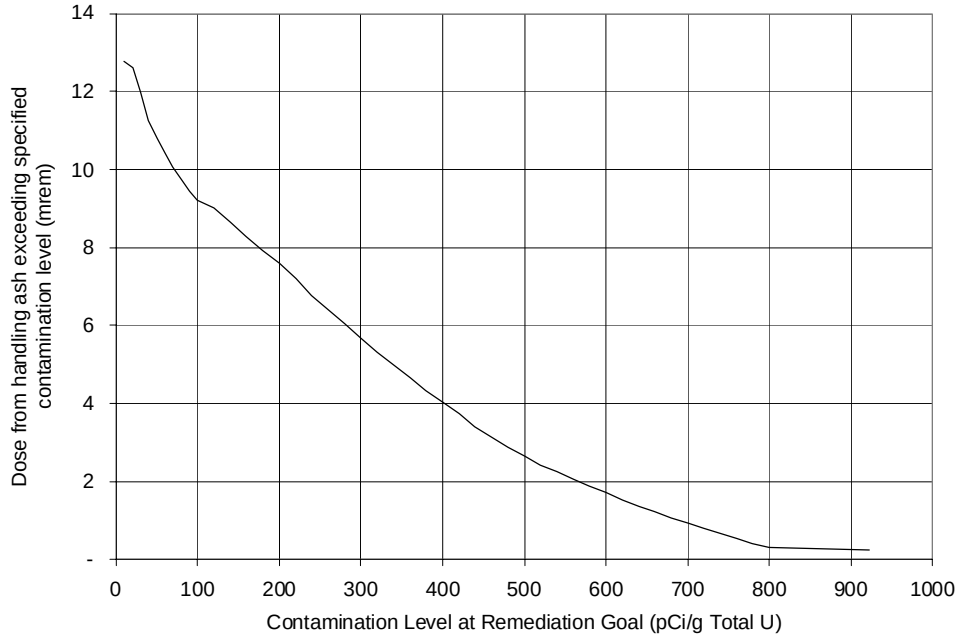

Figure 5: Predicted dose to an ash removal worker as a function of the contaminant concentration remediation goal.

External irradiation to excavation workers was computed by estimating the areal average concentration across the entire lagoon in each 50 cm layer (Figure 6).

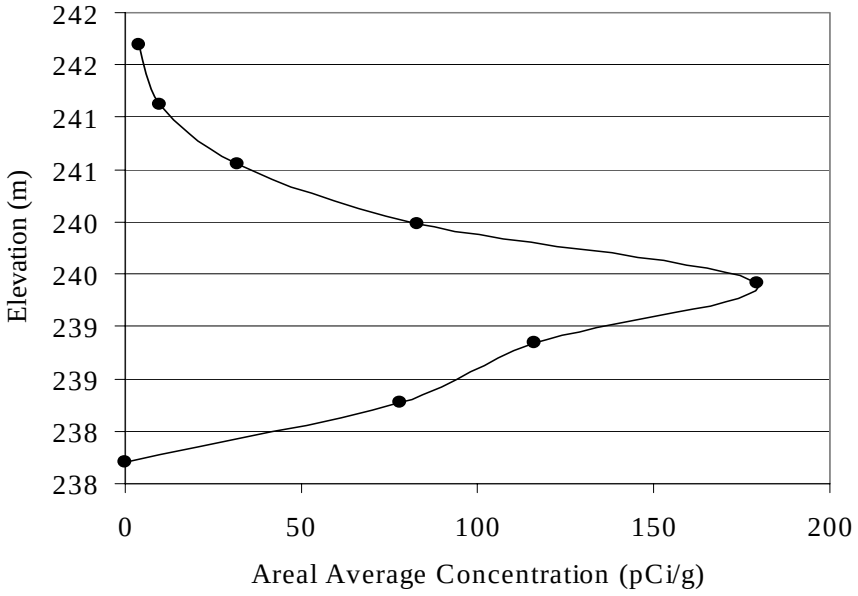

Figure 6: Horizontally Averaged Vertical Contamination Profile

The dose conversion factor from Federal Guidance Report 12 for an infinite plane source of infinite depth were used, and are given in the table below. Isotopes of uranium were assumed to be in equilibrium with their short lived decay products. A regression analysis of the results of the radiological characterization of the lagoon performed by ORISE (ORISE, 1995) indicated that cobalt-60 was typically a factor of 52.4 less than the value of total uranium at the time of sampling. The relative abundance (amount present in relation to total uranium) for cobalt was based on this factor multiplied by a decay correction to account for the ten years of decay that has elapsed since the measurements were made in 1995. The resulting total weighted DCF reflects the dose factor based on total uranium.

Table 4: Computation of Removal Worker Direct Radiation Dose Conversion Factors

Isotope	DCF (Sv/s per Bq/m3)	Relative Abundance	Weighted DCF (Sv/s per Bq/m3)
U234	2.15E-21	0.8	1.72E-21
U235	2.24E-17	0.05	1.12E-18
U238	2.36E-17	0.15	3.54E-18
Co60	8.68E-17	0.005	4.40E-19
Total			5.10E-18

Consistent with the 240 hr exposure time used in the inhalation analysis, each 50 cm layer is assumed to require 30 hours of excavation. The dose is derived by multiplying the DCF by the exposure time and the concentration for each level, and summing the results.

Table 5: External Doses Arising from Different Buried Layers

Elevation (m asl)	Areal Average U_{total}	External Dose from Excavation of Level (mrem)
240.6	32	0.091
240.0	83	0.24
239.4	180	0.51
238.8	120	0.33
238.3	78	0.22
237.7	0.0	-
Total		**1.4**

The resulting computed dose from external irradiation is on the order of a few mrem, bringing the total dose to less than 0.15 mSv (15 mrem).

The primary uncertainties in the worker dose in the ash removal scenario are associated with the mass loading during excavation, the ventilation rate of the exposed workers, and the

exposure duration. The most significant of these is the mass loading, which was set at a conservative upper bound. Doses could be higher than those computed if workers are engaged in heavy activity for the entire time, which is believed to be unlikely. However, moisture in the ash may result in significant dust suppression. Dust suppression or dust masks would significantly reduce inhalation doses by limiting the mass loading, which is set at a conservative upper bound in this analysis.

2.2.2 Evaluation of Landfill Disposal

In order to evaluate the potential doses to a farmer occupying the landfill site at some point in the future, or to an individual residing near the landfill, the source term is assumed to be similar to the source term for the onsite compliance scenario. Analysis of the potential impacts of landfill disposal is based on the following assumptions:

the ash would be homogenized by the process of removal and disposal at the landfill, thereby eliminating localized areas of high uranium concentrations and
the effects of landfill leachate chemistry on uranium mobility can be represented by the use of a distribution coefficient derived from the results of the RAU procedure.

A major difference between the onsite scenario and the landfill scenario is the predicted leachate chemistry. The use of the distribution coefficient derived from the RAU procedure, which is based on the EPA Toxicity Characteristic Leaching Procedure Test, is believed to be appropriate for a landfill because the acetic acid extractant used in the TCLP and RAU procedure is intended to simulate the environment of a municipal waste landfill (EPA/SW 846-Method 1311). Additional differences include the spatial homogeneity of the uranium contamination and the characteristics of the cover layer. Movement of the waste to a landfill is expected to homogenize the ash, justifying the use of an average 3.0 Bq/g (80 pCi/g) total uranium concentration. Instead of a slightly contaminated cover layer, a clean cap is assumed to be present because a landfill would be expected to be covered with a clean cap as a part of landfill decommissioning. The scenario is therefore modeled identically to the onsite scenario, with the exceptions that a uniform uranium concentration of 3.0 Bq/g (80 pCi/g) is used, only waste buried under a clean cap is considered, and a distribution coefficient of 1790 mL/g (Section 2.4) is used to represent uranium mobility in the contaminated layer.

Direct intrusion into the landfill is expected to be unlikely. Institutional measures to prevent direct intrusion are required as a condition of landfill closure, and any subsequent intrusion in the event of a landfill failure would most likely result in complex public health problems apart from the radiological considerations. The physical barriers to radionuclide transport at a landfill are expected to be more protective than the physical barriers present in the ash lagoon. Either a municipal or hazardous waste landfill would be expected to have an engineered barrier that would serve to limit radionuclide transport offsite, such as an engineered clay layer. Such an engineered barrier is expected prevent radionuclide transport at least as well as the natural clay barrier at the ash lagoon. In addition, either a municipal waste or hazardous waste landfill would be expected to have an engineered cap to limit infiltration at the site, which would further limit the possibility of contaminant transport. Information on reported infiltration rates at RCRA Subtitle D (municipal waste) landfills is presented in ISCORS (2003). Reported infiltration rates range from 40 m/yr to 0.16 m/yr, with computer analysis resulting in values ranging from 0.033 to 0.22 m/yr (ISCORS, 2003). Thus the highest infiltration rate that could be expected at a landfill is bounded by the value of 0.5 m/yr used in the onsite scenario for deep infiltration. For

these reasons, the physical barriers at a landfill are expected to be bounded in their performance by the physical barriers at the ash lagoon. A low-level waste landfill would be subject to NRC regulation and is therefore not analyzed further. However, infiltration at any plausible candidate sites would be expected to be quite low.

The results of a deterministic analysis of the effects of acidic leachate chemistry indicates that annual doses to an average member of the critical group would not be likely to exceed 0.25 mSv (25 mrem) within the 1000-year compliance period. The peak dose is less than 0.15 mSv/yr (15 mrem/yr) and occurs at approximately 2000 years, and is due, as in the onsite scenarios, to erosion of the cover layer. A smaller peak of about 0.05 mSv (5 mrem/yr) occurs at approximately 5000 years and is due to consumption of contaminated groundwater. These low doses are based on the experimental results indicating that the uranium is strongly associated with the ash particles and would not be expected to be effectively mobilized by contact with a weakly acidic leachate (see Section 2.4). Doses to a user of groundwater near a landfill, although not explicitly calculated due to the lack of site specific information, would be expected to be lower than the results computed above because of the provisions for reducing infiltration and maintaining institutional controls at regulated landfill sites. Furthermore, in the reducing conditions expected to exist in a municipal waste landfill, U(VI) is expected to be reduced to U(IV), which is a much less soluble and less mobile species. Thus the leachability of the uranium from the ash in a municipal waste landfill should be bounded by this analysis.

The chemical conditions at a hazardous waste landfill may be more alkaline because of the potential presence of significant quantities of concrete. In addition, the potential addition of cement kiln dust for ash stabilization, which is a possibility described in the original lagoon closure plan (Chester Environmental, 1994), may, depending on the amount and characteristics of the added dust, result in alkaline, carbonate-rich conditions in the ash. The results of the SAU protocol, following both the water extraction and the acetic acid extraction, suggest that a high pH, high carbonate environment would be more effective at mobilizing the uranium than a weakly acidic solution. Although the SAU extraction solution is expected to be more aggressive than leachate that would evolve in a landfill, the degree of conservatism in the use of distribution coefficients derived from the SAU test results cannot be evaluated without more specific information about the conditions in the landfill and the potential for co-disposal of the ash with cement kiln dust. In contrast to the scenario of onsite disposal, a demonstration of the safety of disposal at a landfill with high carbonate, alkaline conditions would most likely be based on the institutional measures for monitoring leachate chemistry and restricting water use near the landfill and by the engineered measures at the site to reduce deep infiltration of the leachate rather than by reliance on the low leachability of the uranium in the ash.

3 Summary and Conclusions

Based on the best available evidence, exposure to the contaminated ash in the disposal lagoon at the KVWPCA is not expected to result in doses exceeding 0.25 mSv/yr (25 mrem/yr). The primary factor limiting the dose to low levels, given the elevated uranium concentrations, is the limited environmental availability of the uranium in the ash, which significantly reduces the potential for groundwater contamination.

The assumption of a recreational exposure scenario results in an peak mean annual total effective dose equivalent of approximately 0.011 mSv (1.1 mrem) over the next few centuries,

eventually rising to approximately 0.017 mSv (1.7 mrem) at 1000 years. This result is approximately an order of magnitude lower than either the agricultural case or the intrusion case and is due to the lack of agricultural activity at the site. The results of analysis of the agricultural case for the onsite scenario indicate that the peak mean annual dose to an average member of the critical group within the 1000-year compliance period is predicted to be approximately 0.18 mSv (18 mrem) and to occur at 1000 years after the present time. The mean dose during the 1000-year compliance period is due primarily to ingestion of crops contaminated by root and foliar uptake of uranium and its decay progeny (protoactinium-231, radium-226, and lead-210). The peak dose is associated with erosion of the relatively uncontaminated one meter overburden that provides significant shielding and isolation of plant roots from the more contaminated buried ash. Results of the analysis of the intrusion case of the onsite scenario indicate that the peak mean dose within the 1000-year compliance period is expected to be 0.17 mSv (17 mrem/yr) at 1000 years. In both onsite scenario cases, potential groundwater contamination is expected to be limited by the relatively unavailable form of uranium in the ash, evidenced by low fractions of Readily Available Uranium (approximately 3%) and Slowly Available Uranium (approximately 21%) as determined with leaching tests.

In both the agricultural case and the intrusion case of the onsite scenario, it was assumed that a member of the critical group would site their home, well, or cultivated field at a random location within the 4000 m^2 (1 acre) site. In the unlikely case that a farmer were to site a home in the most contaminated 200 m^2 (0.05 acre) area on the site and grow crops in the ash that was displaced by excavation for the construction of a basement, the peak mean annual dose would be expected to be approximately 0.50 mSv (50 mrem).

Doses from inhalation of contaminated dust during excavation of the site also are expected to be less than 0.25 mSv (25 mrem). Inhalation doses are associated with the inhalation of uranium-234 and uranium-238 and are limited by the low average concentration of these isotopes, the limited exposure time, and the limited respirability of the ash. The effect of more aggressive leachate chemistry that may be present in a landfill was evaluated. Neutral to weakly acidic leachate chemistries are not expected to mobilize uranium in the ash to a significant extent. Furthermore, in the reducing conditions expected to exist in a municipal waste landfill, U(VI) is expected to be reduced to U(IV), which is a much less soluble and less mobile species. Doses to a user of groundwater near a landfill also would be limited by the provisions for reducing infiltration and maintaining institutional controls at regulated landfill sites. In contrast to the scenario of onsite disposal, the safety of disposal at a landfill with alkaline conditions would be based on the institutional measures for monitoring leachate chemistry and restricting water use near the landfill and by the engineered measures at the site to reduce deep infiltration of the leachate.

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5 Appendix: Input Parameters

The selection of input parameters is discussed in Section 3.1.5

Onsite Scenario Cases

	Agricultural case Recreational case[(i)]		**Intrusion case**		**Intrusion into Hotspot**	
	Surface	**Buried**	**Surface**	**Buried**	**Surface**	**Buried**
Total U (pCi/g)	25	Distribution [(ii)]	Distribution [(ii)]	92.9	275	92.9
Co-60 [(iii)] (pCi/g)	0.129	Distribution [(ii)]	Distribution [(ii)]	0.478	1.42	0.478
Area (m^2)	4000	4000	700	4000	700	4000
Thickness of Contaminated layer (m)	1	2	0.9	2	0.9	2
Cover layer (m)	0	1	0	1	0	1
Mass loading for foliar deposition (g/m^3)	0.0001	0.0001	0.0001	0.0	0.0001	0.0
Pathways						
External gamma	On	On	On	Off	On	Off

Inhalation	On	On	On	Off	On	Off
Plant ingestion	On (Off)	On (Off)	On	On	On	On
Meat ingestion	On (Off)	On (Off)	Off	On	Off	On
Milk ingestion	On (Off)	On (Off)	Off	On	Off	On
Aquatic foods	Off	Off	Off	Off	Off	Off
Drinking water	On (Off)	On (Off)	Off	On	Off	On
Soil ingestion	On	On	Off	On	Off	On
Radon	Off	Off	Off	Off	Off	Off

(i) Values for the recreational scenario, where different, are given in parentheses

(ii) See Section 3.1.2

(iii) Co-60 concentrations are set as a function of Total U. See Section 3.1.2

Subsurface Parameters

Zone	Thickness (m)	Bulk Density (g/cm^3)	Total porosity	Effective porosity [g]	Field capacity [h]	Hydraulic conductivity [f] (m/yr)	B parameter [f]
Cover	1[a]	1.48[c]	0.44[e]	0.20 (silt)	0.24	65 (silt)	3.8 (silt)
Contaminated	2 [a]						
Unsaturated	3.5 [b]	1.64[d]	0.38[f] (sandy clay)	0.06 (clay)	0.32	47 (sandy clay)	6.09 (sandy clay)
Saturated	not used	1.51[d]	0.43[f] (sand)	0.32 (sand)	0.11	10,850 (sand)	not used

[a] See section 3.1.1
[b] Groundwater investigation (Chester, 1992)
[c] Upper end of range of densities for sewage sludge ash (REA, 1980)
[d] Calculated from total porosity based on assumed particle density of 2.65 g/cm^3
[e] Calculated from bulk density based on assumed particle density of 2.65 g/cm^3
[f] Recommended value or mean of the recommended distribution (NUREG/CR – 5512 Volume 3) for the soil type (IT Corp., 2002)
[g] Mean of recommended distribution (Yu *et al*., 1993) for the soil type (IT Corp., 2002)
[h] Calculated from the effective porosity as demonstrated in Yu *et al*., 1993

Distribution Coefficients

Correlation coefficients of 0.99 were used to correlate distribution coefficients for uranium isotopes in the same layer

	Ash	**Unsaturated**	**Saturated**[iv]
U (cm^3/g)	Triangular distribution[i]	Triangular distribution[i]	35
Ac-227 (cm^3/g)	1740[ii]	2400[iii]	450
Co-60 (cm^3/g)	1000[ii]	550	60
Pa-231 (cm^3/g)	2040[ii]	2700[iii]	550
Pb-210 (cm^3/g)	2400[ii]	550[iii]	270
Ra-226 (cm^3/g)	3550[ii]	9100[iii]	500
Th-230 (cm^3/g)	5890[ii]	5800[iii]	3200

(i) See section 3.1.3
(ii) Mean of distribution recommended in NUREG/CR-5512 Volume 3
(iii) Recommended value for clay (Sheppard and Thibault, 1990)
(iv) Recommended value for sand (Sheppard and Thibault, 1990)

Onsite Scenario

Parameter	Input	Reference
Contaminated zone area (m^2)	4000	IT Corp., 2002
Length parallel to aquifer flow (m)	100	IT Corp., 2002
Cover depth erosion rate (m/yr)	distribution	Yu *et al*., 1993
Contaminated zone erosion rate (m/yr)	distribution	Yu *et al*., 1993
Average annual wind speed (m/s)	1.45	Mean of distribution (Yu *et al*., 1993)
Evapotranspiration coefficient	0.625	Mean of distribution (Yu *et al*., 1993)
Precipitation (m/yr)	0.96	30 year average for Pittsburgh (National Climatic Data Center)
Irrigation (m/yr)	0.5	Mean of irrigation rates for humid states (NUREG/CR-5512 V. 3)
Irrigation mode	overhead	Default
Runoff coefficient	0.45	Mean of distribution (Yu *et al*., 1993)
Watershed area for nearby stream or pond (m^2)	74,320	IT Corp., 2002
Inhalation rate (m^3/yr)	12,260	Median of distribution (NUREG/CR 5512 v.3)
Mass loading for inhalation (g/m^3)	1.45E-5	NUREG/CR 5512 v. 3
Exposure duration (y)	30	Default
Shielding factor, inhalation	0.4	Default
Shielding factor, external gamma	0.27	Weighted average of indoor and outdoor shielding factors (NUREG/CR 5512 v.3) based on mean indoor and outdoor exposure times (NUREG/CR 5512 v.3)
Fraction of time spent indoors	0.66	Mean of distribution (NUREG/CR 5512 v.3)
Fraction of time spent outdoors (on site)	0.11	Mean of distribution (NUREG/CR 5512 v.3)
Hydraulic gradient	0.01	IT Corp., 2002
Water table drop rate (m/yr)	0.0	Aquifer in communication with the river (see Section 2.2)
Well pump intake depth (m below water table)	not used	Not used in mass balance model
Model: Nondispersion (ND) or Mass-Balance (MB)	MB	More conservative than ND
Well pumping rate (m^3/yr)	214	Three times the mean annual per capita water consumption rate for PA (NUREG/CR 5512 v. 3)

Shape Factor Parameters for External Exposure Pathway for Onsite Scenario

Shape factor flag		Non circular
	Outer annular radius (m)	Fractions of annular areas within area
ring 1:	4.333E+00	100%
ring 2:	8.667E+00	100%
ring 3:	1.300E+01	100%
ring 4:	1.733E+01	100%
ring 5:	2.167E+01	100%
ring 6:	2.600E+01	100%
ring 7:	3.033E+01	79%
ring 8:	3.467E+01	59%
ring 9:	3.900E+01	40%
ring 10:	4.333E+01	17%
ring 11:	4.767E+01	9.2%
ring 12:	5.200E+01	2.7%

Storage times of contaminated foodstuffs (days)

Fruits, non-leafy vegetables, and grain	14	NUREG/CR 5512 v. 3
Leafy vegetables	1	NUREG/CR 5512 v. 3
Milk	1	NUREG/CR 5512 v. 3
Meat and poultry	20	Value for beef (NUREG/CR 5512 v. 3)
Fish	Not used	
Crustacea and mollusks	Not used	
Well water	1	RESRAD default
Surface water	1	RESRAD default
Livestock fodder	45	RESRAD default

Crop Parameters

	Non-Leafy	**Leafy**	**Fodder**
Wet weight crop yield (kg/m^2)	2.40[(a)]	2.89[(a)]	1.91[(a)]
Growing Season (years)	0.17[(b)]	0.25[(b)]	0.08[(b)]
Translocation Factor for Non-Leafy	0.1[(a)]	1.0[(a)]	1.0[(a)]
Dry Foliar Interception Fraction for Non-Leafy	0.35[(c)]	0.35[(c)]	0.35[(c)]
Wet Foliar Interception Fraction	0.35[(c)]	0.35[(c)]	0.35[(c)]
Weathering Removal Constant	36[(d)]	36[(d)]	36[(d)]

(a) Mean of distribution (NUREG/CR-5512 v. 3)
(b) RESRAD default
(c) Mean of distributions for leafy and non-leafy vegetables and beef cattle forage
(d) Mean of distribution (Yu *et al*., 1993)

Agricultural Pathway Parameters

Parameter	Value	Reference
Fruits, vegetables and grain consumption (kg/yr)	112	Sum of means of distributions for fruits, vegetables, and grains (NUREG/CR 5512 v. 3)
Leafy vegetable consumption (kg/yr)	21	mean of distribution (NUREG/CR 5512 v.3)
Milk consumption (L/yr)	233	mean of distribution (NUREG/CR 5512 v.3)
Meat and poultry consumption (kg/yr)	65	Sum of means of distributions for meat and poultry (NUREG/CR 5512 v. 3)
Soil ingestion rate (g/yr)	18.3	mean of distribution (NUREG/CR 5512 v.3)
Drinking water intake (L/yr)	410	mean of distribution (NUREG/CR 5512 v.3)
Contamination fraction of drinking water	100%	NUREG/CR 5512 v.3
Contamination fraction of livestock water	100%	NUREG/CR 5512 v.3
Contamination fraction of irrigation water	100%	NUREG/CR 5512 v.3
Contamination fraction of plant food	100%	NUREG/CR 5512 v.3
Contamination fraction of meat	100%	NUREG/CR 5512 v.3
Contamination fraction of milk	100%	NUREG/CR 5512 v.3
Livestock fodder intake for meat (kg/day)	44	Sum of beef livestock feed values (NUREG/CR 5512 v.3)
Livestock fodder intake for milk (kg/day)	67	Sum of milk livestock feed values (NUREG/CR 5512 v.3)
Livestock water intake for meat (L/day)	50	NUREG/CR 5512 v.3
Livestock water intake for milk (L/day)	60	NUREG/CR 5512 v.3
Livestock soil intake (kg/day)	1.11	Based on average of feed consumed by beef livestock and milk livestock and soil consumption fraction of 0.02 g soil/g feed (NUREG/CR 5512 v.3)
Mass loading for foliar deposition (g/m^3)	1.00E-4	
Depth of soil mixing layer (m)	0.15	NUREG/CR 5512 v.3
Depth of roots (m)	0.9	
Drinking water fraction from ground water	100%	Conservative assumption
Household water fraction from ground water	---	Not used
Livestock water fraction from ground water	100%	Conservative assumption
Irrigation fraction from ground water	100%	Conservative assumption

Concentration Ratios and Dose Conversion Factors

Nuclide	Inhalation[(i)] (mrem/pCi)	Ingestion[(i)] (mrem/pCi)	Plant/soil concentration ratio[(ii)]	Beef/livestock-intake ratio [(iii)], (pCi/kg)/(pCi/d)	Milk/livestock-intake ratio[(ii)], (pCi/L)/(pCi/d)
Ac-227+D	6.72E+0	1.48E-2	Lognormal distribution	3.4E-5	Lognormal distribution
Pa-231	1.28E+0	1.06E-2	Lognormal distribution	8.5E-6	Lognormal distribution
Pb-210+D	2.32E-2	7.27E-3	Lognormal distribution	1.0E-3	Lognormal distribution
Ra-226+D	8.60E-3	1.33E-3	Lognormal distribution	1.3E-3	Lognormal distribution
Th-230	3.26E-1	5.48E-4	Lognormal distribution	1.7E-4	Lognormal distribution
Co-60			Lognormal distribution	5.1E-2	Lognormal distribution
U-234	1.32E-1	2.83E-4	Lognormal distribution	1.0E-3	Lognormal distribution
U-235+D	1.23E-1	2.67E-4	Lognormal distribution	1.0E-3	Lognormal distribution
U-238+D	1.18E-1	2.69E-4	Lognormal distribution	1.0E-3	Lognormal distribution

(i) Federal Guidance Report 11 (Eckerman, 1988)
(ii) Yu *et al*., 1993
(iii) Mean of distribution recommended in Yu *et al*., 1993