WHO UNEP IAEA

INTERNATIONAL SYMPOSIUM ON HEALTH IMPACTS OF DIFFERENT SOURCES OF ENERGY

Nashville, Tennessee, USA, 22–26 June 1981

IAEA-SM-254/ 30
MLM-2829 (OP)
CONF-810652--1

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ASSESSMENT OF THE IMPACT OF RADIONUCLIDES IN COAL ASH

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*Mound Facility is operated by Monsanto Research Corporation for the U.S. Department of Energy under contract No. DE-AC04-76-DP00053.

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ABSTRACT

An assessment of the potential environmental and health impacts of radionuclides in the coal fuel cycle is being conducted at Mound. This paper describes our studies evaluating the potential for migration of radionuclides from ash disposal sites.

Studies at a power plant burning western U. S. coal dealt with an assessment of potential radiation doses from coal ash ponds and leachate discharges of radionuclides from the ponds. Emanation of radon-222 from the ash is relatively low. The emanation of radon-222 from the ash pond (radium-226 at 4.5 pCi/g) is predicted to be about six times less than from soil (radium-226 at 1 pCi/g). Ash with radium-226 at 25 pCi/g would approximate emanation of radon-222 from soil. At 1000 m from the center of the ash pond area, radon-222 from the ash pond is predicted to be 1000 to 6000 times less than background (0.1 to 0.5 pCi/liter).

Pathways exist for transport of radionuclides leached from ash into the aquifer beneath the holding ponds, but concentrations of radionuclides in water leaving the pond are lower than concentrations in groundwater which is upgradient of the ponds. Leachability of the ash is quite low, on the order of 0.002% in one month, and flow of ash sluicing water (3% of the volume of the ponds each day) has actually diluted normal background concentrations of radionuclides in the aquifer between the ponds and the adjacent river.

INTRODUCTION

Coal is expected to play an increasingly important role in meeting energy needs of the United States as we move to reduce our dependence on imported fossil fuels. The combustion of coal releases trace elements, including naturally occurring radionuclides, to the atmosphere as vapors and particles; and these particles have relatively greater concentrations of certain trace constituents than the feed coal [1-4]. Although the environmental impact of these residuals on ecosystems is not certain, control of these residuals appears to be a reasonable goal within acceptable cost limits.

A project was initiated at Mound Facility to assess the fate of radionuclides in coal and their associated health and environmental effects. Phase I included in the experimental design a plan to broadly survey pathways of radionuclides in the coal fuel cycle for western U. S. coal.

Samples of coal collected in Phase I from 19 active western mines that produce 65% of the coal mined in the province had an average concentration (95% confidence limits) for uranium-238, uranium-234, and radium-226 of 0.31 (±0.10) pCi/g [5]. The data for uranium-238 are somewhat below the national average (0.60 pCi/g) [6]. Pathways of radionuclides in a coal-fired steam

electric generating plant (1000 MWe) were investigated through analysis of coal, bottom ash, fly ash, stack effluents, airborne particulates, soil, and vegetation. Bottom ash and fly ash contained relatively higher concentrations of uranium-238, uranium-234, thorium-230, lead-210, and polonium-210 than did the feed coal. Although some small fraction of the radionuclides apparently bypassed the electrostatic precipitator as vapors or in association with very fine particles as do other constituents of coal, the electrostatic precipitator effectively controlled emission of radionuclides associated with fly ash. Atmospheric dispersion calculations, using data on stack effluents, indicated maximum depositions over a 20-yr period to be 0.1 to 1.0% of measured background concentrations [7].

Coal ash is presently accumulating in the United States at a rate in excess of 60 million tons annually [8]. Roughly 10% of this is being used in a variety of products such as concrete, aggregate in stabilizing roadways, and a filler in putty, paint, and wallpaper. Fly ash may also be used as a "dewatering" agent for waste slurries from flue gas desulfurization systems. The vast majority of waste products from flue gas desulfurization systems will be directly disposed of in ponds or used in landfill [9]. The potential of such practices for enhancement of radiation doses to man warrants further evaluation.

The scope of Mound's assessment of the radionuclide concentration in coal refuse and the potential migration of radionuclides from ash disposal sites has remained limited to western U. S. coal. Los Alamos National Laboratory is evaluating eastern U. S. coal. Mound's efforts are reported here for the following tasks:

1. Evaluation of potential for airborne radiation doses from coal ash.

2. Evaluation of leachate discharges of radionuclides from coal ash ponds to groundwater and surface water.

POTENTIAL RADIATION DOSES FROM COAL ASH

In this study of ash ponds at the George Neal Power Station near Sioux City, Iowa, U.S.A. (Figure 1), two pathways are considered which may potentially result in radiation doses to individuals in the vicinity of a site where ash has been deposited. The first pathway is the emanation and subsequent dispersion of radon-222 from the ash. Upon decay, radium-226 forms radon-222, which is an inert, radioactive gas. The radon can diffuse through voids within and between particles of ash, and then emanate from the surface of the ash pile. Upon dispersion, the radon and its short-lived decay products may be inhaled, and thus produce a radiation dose in the lungs of individuals in the vicinity of the ash pile. Also, the decay products of radon may deposit on the ground and subsequently decay to lead-210, bismuth-210, and polonium-210. These radionuclides may then be taken up by food crops grown in the area, and individuals may receive radiation doses from the ingestion of these foods. Because ash contains concentrations of radium-226 that are slightly greater than those generally found in soil, the doses resulting from the emanation and dispersion of radon-222 may be larger than those resulting from background radon-222.

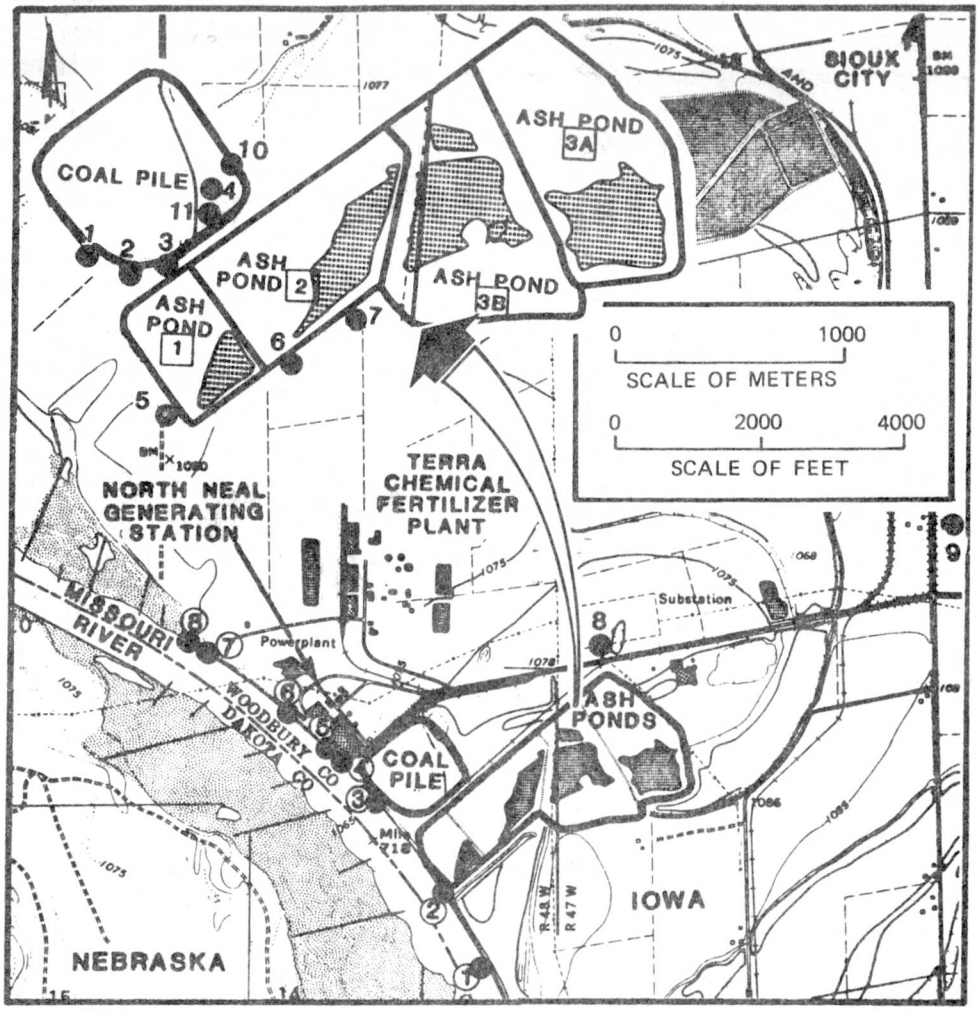

Figure 1 - The 11 groundwater monitoring wells (●), the 8 surface water sampling locations (○), and the 4 ash ponds (□). The scale refers to the base map only; its values must be divided by 2 for use with the enlargements of the coal pile and the ash ponds.

The second pathway is the suspension and dispersion of the ash itself. The dried areas of the ash ponds at the Neal Station seemed to have formed a crust on the surface that eliminated or retarded the suspension of the ash by wind. However, it seems likely that, if the ash were to remain unstablized over a long period of time, this crust would break up as the result of various weathering processes, and the ash could then be suspended and dispersed by wind. Thus, the ash is a potential source of radiation doses to individuals in the vicinity, through inhalation of suspended ash and ingestion of foods which may take up deposited radionuclides. It should be pointed out that the intent here is to estimate the potential impact of the ash if it is left in place over a long period of time in an unstabilized condition (so that suspension is possible). Dispersion of ash resulting from short-term disturbances, such as digging and moving the ash, is not considered here.

Physical characteristics of the ash, emanation of radon-222, and derivation of source terms for dispersion modeling are described elsewhere [10].

Atmospheric dispersion calculations were performed to estimate ground-level air concentrations and deposition rates of radionuclides as well as doses from inhalation and ingestion pathways, resulting from the release of 1.42 Ci of radon-222 per year from the ash piles at the Neal Station, the ash containing radium-226 at 4.5 pCi/g. A computer code [11], which is currently being used by the Tennessee Valley Authority for uranium mining and milling licensing calculations, was used to perform the calculation. This computer code uses a sector-average Gaussian dispersion plume model and uses methodologies consistent with U.S.N.R.C. Regulatory Guides 1.109 [12] and 1.111 [13]. Further, the code explicitly calculates decay and growth of all radionuclides in the uranium-238 decay series.

The meteorological data used in the dispersion calculations were obtained from the National Climatic Center in Asheville, North Carolina. These data were based on observations covering the 5-yr period, January 1970 through December 1974, at the Sioux City, Iowa, weather station.

For each receptor location, the following were calculated: ground-level air concentration of radon-222, polonium-210, lead-214, bismuth-214, and polonium-214; ground-level air concentration of the short-lived decay products of radon-222 in working levels (WL); deposition rates of polonium-218, lead-214, bismuth-214, and polonium-214; annual average inhalation dose to the bronchial epithelium; and annual average ingestion dose to bone, total body, kidneys, liver, and G.I. (gastrointestinal) tract.

The location for which the calculated doses were the largest was at 1000 m in the WNW direction, hereafter referred to as "1000 m WNW." A summary of the calculated concentrations, deposition rates and doses for this location is presented in Table 1.

There are no regulatory guidelines or limits which directly apply to release of radionuclides from coal-fired power plants. However, the concentrations of radon-222 (6.9×10^{-5} pCi/liter)

Table 1 - ANNUAL AVERAGE CONCENTRATIONS, DEPOSITION RATES, AND DOSES AT 1000 M WNW

Parameter	Calculated Value
Rn-222 conc.	6.9×10^{-5} pCi/liter
Po-218 conc.	4.9×10^{-5} pCi/liter
Pb-214 conc.	5.4×10^{-6} pCi/liter
Bi-214, Po-214 conc.	5.9×10^{-7} pCi/liter
Combined Radon Decay Products	8.0×10^{-8} WL
Po-218 dep. rate	3.0×10^{-10} $\mu Ci/m^2$ -sec
Pb-214 dep. rate	2.6×10^{-11} $\mu Ci/m^2$ -sec
Bi-214, Po-214 dep. rate	2.3×10^{-12} $\mu Ci/m^2$ -sec
Inhalation dose to bronchial epithelium	8.5×10^{-3} mrem
Ingestion dose to:	
Bone	3.9×10^{-4} mrem
Total Body	1.6×10^{-5} mrem
Kidneys	3.8×10^{-4} mrem
Liver	1.3×10^{-4} mrem
G. I. Tract	3.4×10^{-6} mrem

and its decay products (8.0×10^{-8} WL) are small in comparison with the limits listed in 10CFR20 for release to uncontrolled areas: 1 pCi/liter for radon-222 or 1/30 WL (0.033WL) for the short-lived decay products of radon-222. In fact, the radon-222 concentration in Table 1 is small in comparison with background concentrations which are typically on the order of 0.1 to 0.2 pCi/liter [14]. Also, the doses are small in comparison with proposed limits under 40CFR190 for uranium fuel cycle facilities of 25 mrem to total body or any organ other than thyroid.

The effects resulting from radon-222 emanating from the ash piles at the Neal Station are insignificantly small, even if they were an addition to background. However, the ash is physically replacing the soil that would be there if the ash were not. According to Harley [14], a reasonable average value for radon-222 emanation from soil is 1600 pCi/cm^2-yr or about 0.5 pCi/m^2-sec. If it is assumed that this value is applicable to the soil in the vicinity of the Neal Station, and that the derived value for ash of 0.09 pCi/m^2-sec is accurate, then the ash piles at the Neal Station may be emanating less radon-222 than the soil without the ash. Another way of stating this is that if the radium-226 concentration in the ash were increased to about 25 pCi/g, then the radon-222 emanating characteristics of the ash would be essentially the same as those of soil, and the effects of radon-222 emanation would be indistinguishable from background.

The value of 0.09 pCi/m^{2}sec was derived considering only the ash as a source of radon-222; therefore, the actual radon flux from the surface of the ash pile may be somewhat larger than this value because of radon-222 emanating from the soil beneath the ash piles. However, the radon-222 flux from the surface of the ash piles should be lower than that from the surrounding soil. It is therefore concluded that radon emanation from the ash piles at the Neal Station is of no present or potential environmental concern.

Atmospheric dispersion and dose calculations were also performed for the suspension of ash [10] in the same manner as was described in the previous section for the dispersion of radon-222.

The largest doses calculated were for location 1000 m SSE. The largest inhalation dose was approximately 1.8 mrem to bone. The breakdown of doses by radionuclide indicates that, for kidneys, liver, bone, and lungs, the isotopes of thorium contribute a very large percentage of the total dose. For total body and the G.I. tract, radium-226 is the largest contributor to the total dose, with the thorium isotopes making up a large percentage of the remainder.

The largest ingestion dose was approximately 40 mrem to bone. The breakdown of doses by radionuclide indicates that, for every organ, radium-226 and lead-210 contribute a very large percentage of the total dose.

It must be reemphasized that the methodology used to calculate suspension source terms is based on models which have not been verified. The intent was to arrive at a rough, order-of-magnitude estimate of potential effects should the ash ponds be

allowed to be suspended and dispersed by the wind. One should keep this in mind when interpreting the above results.

The largest calculated dose was to bone and resulted almost entirely from deposition of radium-226. The concentration of radium-226 in the ash was measured to be about 4.5 pCi/g. The average concentration of radium-226 in soil is reported to be on the order of 0.7 pCi/g [15]. Data presented in the Phase I report [1] indicate that the radium-226 concentration in soil in the vicinity of the Neal Station should be in the range of about 0.6 to 2.0 pCi/g. Thus, application of the models used here to the suspension, dispersion, and deposition of soil would lead to predicted ingestion doses on the order of 10 to 45% of those predicted for ash. Therefore, the dose methodology used here would predict that background ingestion doses may be an appreciable fraction of the 25-mrem limit proposed under 40CFR190. Further, it was assumed that the radionuclides in the ash were available for uptake by plants. Data presented in Table 2 indicate that the radionuclides are not easily leached from the ash and, therefore, may not be taken up by plants. Thus, the ingestion doses would be greatly reduced. For these reasons, it is considered that the potential impact through the ingestion pathway resulting from the suspension of ash is not significant. However, it is also recommended that a prudent approach should be taken for the long-term storage of the ash, and that the ash should be stabilized to preclude suspension and dispersion.

LIQUID DISCHARGES FROM ASH PONDS

The potential for release of radionuclides from coal ash ponds was evaluated through a series of ash leachate studies and a survey of geohydrology at the George Neal Steam Electric Generating Station, Sioux City, Iowa (Figure 1).

Leaching experiments were performed on material from Ash Pond #3B (Figure 1) of the Neal Station and coal from the Neal Station.

Distilled water and 75- to 150-g samples were placed in 1-liter Erlenmeyer flasks and shaken on a Burrell Wrist-Action Shaker or stirred with a magnetic stirrer. Replicate leachates were combined and divided into two equal samples for duplicate analyses [10].

Measured concentrations of radionuclides leached from ash and coal are given in Table 2. The data show that very little of the elements of interest was found in the leachates. However, radionuclide concentrations were significantly higher in coal leachates than in ash leachates. The data also suggest that uranium-234 is more leachable from most of the samples than uranium-238 is. This suggests that the radionuclide decay process may cause activities of daughters to be less bound to the sample matrix than the parent radionuclide.

The field study involved an assessment of surface and groundwater transport of selected radionuclides from ash ponds of the Neal Station to the environs. Specific objectives included:

Table 2 - LEACHABILITY OF NEAL STATION COAL AND ASH WITH WATER

Sample	Radionuclide	Concentration (pCi/g)	Total Activity In Samples (pCi)	Total Activity In Leachates (pCi)	Percent Leached
Neal Station	U-238	0.54 ± 0.02	41	0.36	0.88
Coal	U-234	0.53 ± 0.02	40	0.65	1.6
	U-235	0.022 ± 0.003	1.7	0.025	1.5
	Th-231	0.28 ± 0.006	21	< 0.05	< 0.2
	Th-230	0.63 ± 0.10	47	0.06	0.13
	Pb-210	0.40 ± 0.12	30	4.7 ± 0.3	16
Neal Station	U-238	3.93 ± 0.09	2950	0.035	0.0012
Ash, Pond #3b	U-234	3.93 ± 0.09	2950	0.066	0.0022
(32302)	U-235	0.18 ± 0.02	135	< 0.015	< 0.01
	Th-232	2.04 ± 0.11	1530	0.057	0.0037
	Th-230	4.67 ± 0.11	3500	0.11	0.0031
	Pb-210	2.98 ± 0.23	2230	< 1.5	< 0.07
Neal Station	U-238	4.01 ± 0.13	2400	0.035	0.0015
Ash, Pond #3b	U-234	4.03 ± 0.13	2420	0.047	0.0019
(32303)	U-235	0.22 ± 0.02	132	< 0.012	< 0.01
	Th-232	1.86 ± 0.10	1120	0.017	0.0015
	Th-230	4.44 ± 0.19	2660	0.11	0.0041
	Pb-210	3.08 ± 0.35	1850	< 1.2	< 0.07
Neal Station	U-238	4.41 ± 0.14	2200	0.023	0.0010
Ash, Pond #3b	U-234	4.44 ± 0.14	2220	0.044	0.0020
(32304)	U-235	0.22 ± 0.03	110	< 0.012	< 0.01
	Th-232	1.93 ± 0.11	965	0.015	0.0016
	Th-230	4.57 ± 0.15	2300	0.055	0.0024
	Pb-210	3.62 ± 0.23	1810	< 1.3	< 0.07

(1) identify radionuclide levels in the surface water and groundwater regimes at the site, (2) assess the hydrologic and geologic conditions at the Neal Station from available historical information, and (3) formulate a preliminary working model describing the operating mechanism for movement of radionuclides from the ash pond system. Samples of water from ash ponds, test wells (Figure 1), and the Missouri River (Figure 1) were analyzed for radionuclides as described in Styron et al. [10] for stable trace elements by Inductively Coupled Argon Plasma Emission Spectroscopy.

Samples were collected from nine groundwater monitoring wells (Figure 1) and eight river sampling locations (Figure 1) on July 16-18, 1979 and June 23-24, 1980. The following data were gathered at each monitoring well: date, time, weather, well number, water level before and after pumping, depth to well point, pH and general appearance of the water. All samples, except for fractions designated for gross alpha and gross beta analyses, were filtered and acidified at the power plant.

Data are grouped according to the particular types of environment found in the study area. The environments identified and data points (sampling locations) representing these environments include:

Environment	Data Points
Missouri River	River sample locations 1 - 8
Coal pile - groundwater	Groundwater monitoring wells 1, 2, 3, 4, 10, and 11
Ash pond - groundwater	Groundwater monitoring wells 5, 6, and 7
Background - groundwater	Groundwater monitoring wells 8 and 9

Data on concentrations of radium-226, gross alpha, gross beta, and trace elements are summarized in Table 3. Groundwater samples from background wells (8 and 9) had the highest concentration of radium-226. Groundwater down gradient of the ash ponds and coal pile had concentrations of radium-226 that fell between the concentrations found in background groundwater and surface water of the Missouri River. Gross alpha activity was also lower in ash pond and coal pile groundwater than background groundwater in 1980. The trend for gross beta activity was somewhat different. Background wells had the lowest value; the river, next lowest; and the coal pile and ash pond wells, the highest. Except for sodium, phosphorous, and potassium, most stable elements that were measured were found at the same or lower concentrations in groundwater from the ash pond environment than in water from background wells.

In order to examine the groundwater gradients near the ash ponds, water levels were measured in nine shallow monitoring wells. Water levels measured on July 16 and July 18, 1979 and June 23, 1980, indicate that the predominant groundwater gradient is toward the river. Groundwater gradients are slightly steeper near the river. However, they do not exceed one-half percent.

Table 3 - COMPARISON OF CONCENTRATIONS OF TRACE ELEMENTS IN GROUND WATER AT THE NEAL STATION - JULY 1979 AND JUNE 1980

Parameter	Environment[a] River 1979	River 1980	Coal Pile 1979	Coal Pile 1980	Ash Pond 1979	Ash Pond 1980	Background 1979	Background 1980	EPA Drinking Water Standard
Radium-226[b]	<0.10	0.07	<0.10 to 0.11	0.11	<0.10 to 0.14	0.14	0.17	0.19	5.0
Gross Alpha[b]	2.73	1.74	6.90	3.18	1.76	1.5	3.08	5.75	15.0
Gross Beta[b]	4.11	6.21	13.4	9.40	3.17	9.10	3.30	5.00	50.0
Silver	0.01	0.02	0.06	0.04	0.13	0.21	0.14	0.16	0.05
Aluminum	0.06	0.53	0.22	0.69	0.25	1.64	0.39	1.14	
Barium	0.05	0.06	0.06	0.06	0.07	0.11	0.19	0.18	1.0
Beryllium	0	0	0.01	0	0	0	0	0	
Boron	0.14	0.14	0.40	0.38	0.55	0.56	0.22	0.21	
Calcium	56.2	58.5	259.0	201.0	40.0	76.0	157.0	139.9	
Cadmium	0	0.01	0	0.01	0	0.04	0	0.03	0.01
Cobalt	0.01	0.02	0.50	0.02	0.02	0.05	0.04	0.03	
Chromium	0.02	0.05	0.07	0.10	0.03	0.14	0.06	0.12	0.05
Copper	0.01	0.05	0.05	0.06	0.05	0.16	0.05	0.11	1.0
Iron	0.05	0.09	0.62	2.30	0.22	0.35	1.55	1.04	0.3
Magnesium	23.3	23.9	86.4	68.9	10.0	15.6	46.9	41.4	
Manganese	0.02	0.02	4.46	3.67	0.53	1.18	2.72	2.40	0.05
Molybdenum	0.02	0.04	0.05	0.08	0.09	0.15	0.07	0.09	
Sodium	60.4	63.9	87.1	82.7	94.6	61.6	26.9	22.1	
Nickel	0.14	0.64	0.53	1.16	0.45	1.95	0.63	1.61	
Lead	0.03	0.18	0.14	0.29	0.17	0.64	0.20	0.40	
Phosphorus	0.08	0.40	0.27	0.79	0.80	1.28	0.47	0.90	
Antimony	0.02	0.14	0.09	0.24	0.14	0.47	0.16	0.34	
Silicon	3.54	1.50	10.3	0	8.42	0	13.3	0	
Tin	0.03	0.12	0.12	0.22	0.06	0.18	0.12	0.24	
Strontium	0.59	0.49	2.09	1.39	0.39	0.59	1.40	1.12	
Titanium	0	0.01	0.01	0.01	0.01	0.04	0.01	0.03	
Vanadium	0.13	0.20	0.48	0.53	0.08	0.25	0.27	0.38	
Zinc	0.01	0.02	0.04	0.36	0.01	0.36	0.03	0.34	5.0
Potassium	4.78	5.62	7.43	7.53	5.41	7.74	6.72	7.27	

[a]Values for River, Coal Pile, Ash Pond, and Background are averages.

[b]These parameters are given in pCi/l. All others are in mg/l.

During the site investigation, no discharge through the dikes or indication of surface runoff were observed. Additionally, it was reported by personnel from Iowa Public Service Company that discharge from the ash pond has never been observed. The conclusion is evident that, except for the small amount of evaporative loss, almost all the 3.19×10^7 liter/day (8,424,000 gal/day) input, i.e., 3.12×10^7 liter/day (8,240,000 gal/day), flows directly out the bottom of the ash pond and ultimately enters the groundwater regimes, which will discharge into the river. A diagramatic sketch representing the operative mechanism is shown in Figure 2.

In this context, we considered a possible scenario in which the burning of coal would tend to concentrate uranium and its daughter products in the ash. In this scenario, the ash disposed of in containment ponds may be selectively leached and "excessive" amounts of radionuclides made available to the groundwater for subsequent transport.

Groundwater samples representing the background environment had the highest concentration of radium-226 whereas the ash pond environment had the lowest. This observation is diametrically opposed to the proposed scenario.

Studies on leachability of radionuclides from Neal Station fly ash point strongly to limited solubility of radionuclides bound in fly ash particles. It is plausible that river water, which is relatively low in concentration of radium-226 (0.074 pCi/liter), used to sluice ash to the holding ponds, in passing from the ponds to the aquifer dilutes natural background (0.216 pCi/liter) to levels observed for the ash pond groundwater environment (0.112 pCi/liter). The short residence time of sluicing water in the pond and the low leachability of radionuclides from ash indicate that little of the radionuclides in ash is transferred to groundwater.

Comparison of concentrations of stable trace elements in the ash pond and background groundwater environments suggest that dilution by ash pond water may also be lowering natural levels of barium, calcium, iron, magnesium, manganese, and strontium. The concentration of sodium in water down gradient of the ash ponds is higher than in the background environment, but the levels are not greatly different from those found in river (sluicing) water. The highest concentration of boron is associated with ash pond groundwater.

In summary, pathways exist for the transport of radionuclides into the hydrologic regime, but concentrations of radionuclides in water leaving the ponds are less than concentrations in groundwater upgradient (Wells 8 and 9) of the ponds. Radionuclide levels observed in monitoring wells are not significantly different from levels observed in the background environment (Wells 8 and 9).

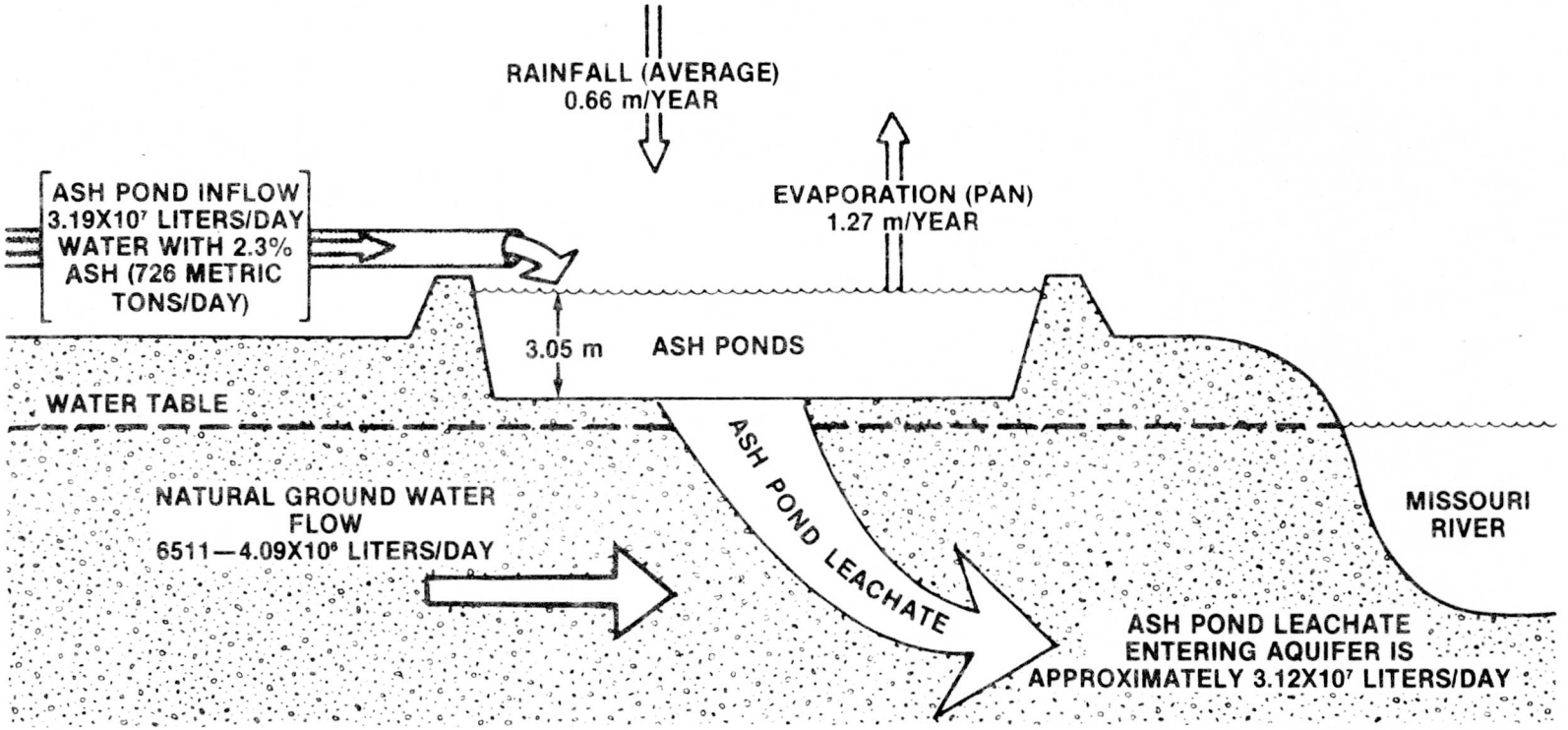

Figure 2 - Water balance for the ash ponds at the North Neal Generating Station. The surface area of the ash ponds (1, 2, 3A, and 3B) is 4.23 x $10^5 m^2$; the capacity of the ponds, which have an average depth of 3.05 m, is 1.29 x $10^6 m^3$.

CONCLUSIONS

1. Modeling of the emanation and dispersion of radon-222 from the ash ponds at the Neal Station indicated that there is no significant potential environmental impact. In fact, because of the very small radon emanation power of the ash, the radon flux from the surface of the ash pond is predicted to be less than from typical soil.

2. Modeling of the suspension and dispersion of the ash itself indicated no significant potential impact from the radionuclides in the ash. The modeling did indicate, however, that annual doses from ingestion of foods grown in the immediate vicinity of the ash ponds could conceivably be of the same order of magnitude as the 25-mrem limit proposed under 40CFR190 for uranium fuel cycle facilities. Therefore, it is recommended that the ash be stabilized for long-term storage in order to prevent the ash from being suspended and dispersed.

3. Pathways exist for transport of radionuclides leached from ash into the aquifer beneath holding ponds at the Neal Station, but concentrations of radionuclides in water leaving the ponds are lower than naturally occurring concentrations in groundwater upgradient of the ponds. Percolation of ash pond water into the aquifer appears to have diluted normal background concentrations of radionuclides.

ACKNOWLEDGEMENTS

The authors gratefully acknowledge the support of the Department of Energy for this study, the management and staff of the Neal Station and the Iowa Public Service Company for their cooperation and help in collecting environmental samples, and other members of the Environmental Assessment and Planning section at Mound for their help in collecting and analyzing samples.

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