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TO : Director, Centers for Disease Control

FROM : Chronic Diseases Division
Bureau of Epidemiology

SUBJECT: Biological Assessment After Uranium Mill Tailings Spill, Church Rock, New Mexico

SUMMARY

On July 16, 1979, a tailings pond dam broke near Church Rock, New Mexico, spilling liquid and solid wastes contaminated with radioactive uranium, thorium, radium, lead, and polonium. Subsequent evaluation of 6 Navajos potentially exposed to the materials spilled in the Rio Puerco River showed them to have no detectable increase in radioactivity by whole-body counting and no increase in urinary radionuclides.

Since food-chain analysis is a more sensitive indicator of possible radiation dose, 8 local and 3 control animals were autopsied to determine radionuclide concentrations in edible tissues. Calculations indicate that human doses which would result from consumption of such animals would be higher from local than from control animals; however, the data also suggest that exposure from chronic ingestion of uranium mine dewatering effluent may be responsible for the elevated radionuclide concentrations found in tissue of local animals. Even though no state or federal regulations were violated, radionuclide concentrations found in animals and calculated human ingestion doses fell in a range that justifies both further surveillance of radionuclides in animals and the natural environment and further efforts to reduce the amount of radiation to which humans and animals are exposed. Navajos in the area have been advised that their exposure to radionuclides may be reduced by not eating kidney and liver from local animals.

INTRODUCTION

Early on the morning of July 16, 1979, there was a breach in the earthen retaining dam of one of the tailings ponds of the United Nuclear Corporation's (UNC) Church Rock uranium mill. The water (approximately 94 million gallons of acidified effluent) and tailings slurry (approximately 1,100 tons) spilled through the damaged portion of the retaining wall into an arroyo that is a tributary to the Rio Puerco river system. The Rio Puerco runs through Gallup, New Mexico, and eventually crosses the New Mexico-Arizona border. On its way to Gallup, the Rio Puerco and its tributaries pass through land with a checkerboard

pattern of ownership, with portions owned or leased by the Navajo, individuals, the Bureau of Land Management, and the state.

In terms of tailings liquid volume, the UNC spill ranks as one of the largest spills (1). The mass of solids released in the slurry appears to be close to the median for accidents of this kind, however. Table 1 provides an overview of the concentrations of radionuclides associated with uranium mill tailings, and Table 2 lists tailings liquid heavy metals concentrations and water quality parameters (1,2).

The New Mexico Environmental Improvement Division (NMEID) was notified of the dam break on July 16 and responded to the incident by collecting environmental samples for radionuclide analysis and by establishing criteria for the cleanup of the contaminated arroyos and river. Officials from the Indian Health Service (IHS), U.S. Environmental Protection Agency (EPA), Nuclear Regulatory Commission (NRC), and Navajo tribe were also apprised of the situation. Because of the uncertainty surrounding the degree of environmental contamination, residents along the river were asked to keep themselves, their children, and their animals away from the Rio Puerco and the contaminated arroyos.

On August 6, 1979, the Center for Disease Control was contacted by Philip J. Landrigan, M.D., National Institute for Occupational Safety and Health, regarding the concerns of an IHS physician, George Winterer, M.D. Dr. Winterer thought that Navajo children, adults, and livestock may have been exposed to dangerous levels of radiation. Henry Falk, M.D., Chronic Diseases Division, Bureau of Epidemiology, initiated inquiries into the UNC situation, contacting the Environmental Analysis Division of the EPA in Region VI, the NMEID, the IHS, and the Bureau of Radiologic Health of the Food and Drug Administration. The NMEID subsequently requested CDC assistance to clarify the possible radiation exposure to humans as a result of the spill. On August 16, CDC personnel from the Chronic Diseases Division (Henry Falk, M.D., Kathleen Kreiss, M.D., A. James Ruttenber, Ph.D., and Erica Schiffman, epidemiology-elective student) met with officials from the IHS and the New Mexico Health and Environment Department. This group subsequently went to Gallup and Church Rock to work with the Church Rock Action Committee and the Navajo Tribal Task Force on Radiation to select appropriate individuals for whole-body counting and urinalysis for radionuclides. CDC researchers also reviewed the exposure histories of animals purchased by IHS for autopsy and radionuclide analysis, and they studied environmental data.

BACKGROUND

In 1977 the United Nuclear Corporation opened its Church Rock uranium mill in an area adjacent to land belonging to the Navajo tribe. The mill, which is next to the previously existing UNC Church Rock mine, is located approximately 10 miles northeast of Gallup, New Mexico, on state road 566 (Figure 1). Gallup, a town of 18,000 people, is the closest population center. The region surrounding the plant site is sparsely populated by Navajos at a density of approximately 15 persons per square mile. The UNC mill and mines currently employ approximately 650 persons, and the adjacent Kerr-McGee uranium mine employs around 300.

The UNC mill normally processes 3,500 tons per day of uranium ore, depositing the acidified tailings slurry in a series of 3 holding ponds, each of earthen construction. The tailings ponds are located along the pipeline arroyo that feeds into the Rio Puerco River approximately 1.5 miles from the southernmost tailings dam. The liquid portion of the tailings slurry evaporates in the ponds, and there is no surface flow from the holding ponds to the arroyo under normal conditions. Runoff from the plant site after heavy rains may deliver radionuclides to the arroyo-river system, however. Percolation of tailings

TABLE 1
Comparison of Radionuclide Concentrations[a] in United Nuclear Corporation (UNC)
Tailings Liquid and River Water, Church Rock, New Mexico, 1979

Radionuclides	Typical Tailings Liquid Concentration[b]	Concentration in Water Sample from UNC Tailings Pond[c]	Maxiumum River Water Concentration[d]	Maximum Permissible Concentration (MPC) in Bodies of Water[e]	
				Soluble	Insoluble
U-238	5.4×10^3	3.9×10^3	----	4.0×10^4	4.0×10^4
U-234	5.4×10^3	----	----	3.0×10^4	3.0×10^4
Th-230	1.5×10^5	9.3×10^4	4.8×10^4	2.0×10^3	3.0×10^4
Ra-226	4.0×10^2	1.3×10^2	5.5×10^2	3.0×10^1	3.0×10^4
Pb-210	4.0×10^2	7.9×10^2	1.3×10^3	1.0×10^2	2.0×10^5
Po-210	4.0×10^2	2.2×10^3	1.3×10^3	7.0×10^2	3.0×10^4
Bi-210	4.0×10^2	----	----	4.0×10^4	4.0×10^4
U-nat	----	----	----	4.0×10^4	4.0×10^4
U-nat(ug/l)	----	----	1.3×10^4	5.9×10^4	5.9×10^4

[a] All concentrations in pCi/l above natural background unless otherwise specified
[b] (1, Vol. I, Sec. 7, P.6)
[c] Water sample collected on 8/1/79 and analyzed by New Mexico Environmental Improvement Division
[d] Unfiltered water samples analyzed for New Mexico Environmental Improvement Division and UNC
[e] (4); Solubility with respect to physiologic fluids; USNRC MPCs are identical to state MPCs.

TABLE 2
Nonradioactive Constituents of Uranium Mill Tailings Liquid,
Church Rock, New Mexico, 1979

Constituent	Typical Mill[a] (mg/l)	UNC Mill[b] (mg/l)
Aluminum	0.0	----
Ammonia	500	----
Arsenic	0.20	1.22
Barium	----	0.29
Calcium	500	----
Carbonate	----	----
Cadmium	0.20	0.11
Chloride	300	----
Copper	50.0	----
Fluoride	5.00	----
Iron	1.0×10^3	----
Lead	7.00	1.56
Manganese	500	----
Mercury	0.07	$<0.5 \times 10^{-3}$
Molybdenum	100	2.30
Selenium	20.0	0.53
Sodium	200	----
Sulfate	3×10^4	----
Uranium	----	11.25
Vanadium	0.10	46.94
Zinc	80.0	7.22
Total Disolved Solids	3.5×10^4	----
pH	2	----

[a](1, Vol. I, Sec. 5, P. 6)
[b]Representative grab sample from UNC tailings pond, analyzed by New Mexico Environmental Improvement Division

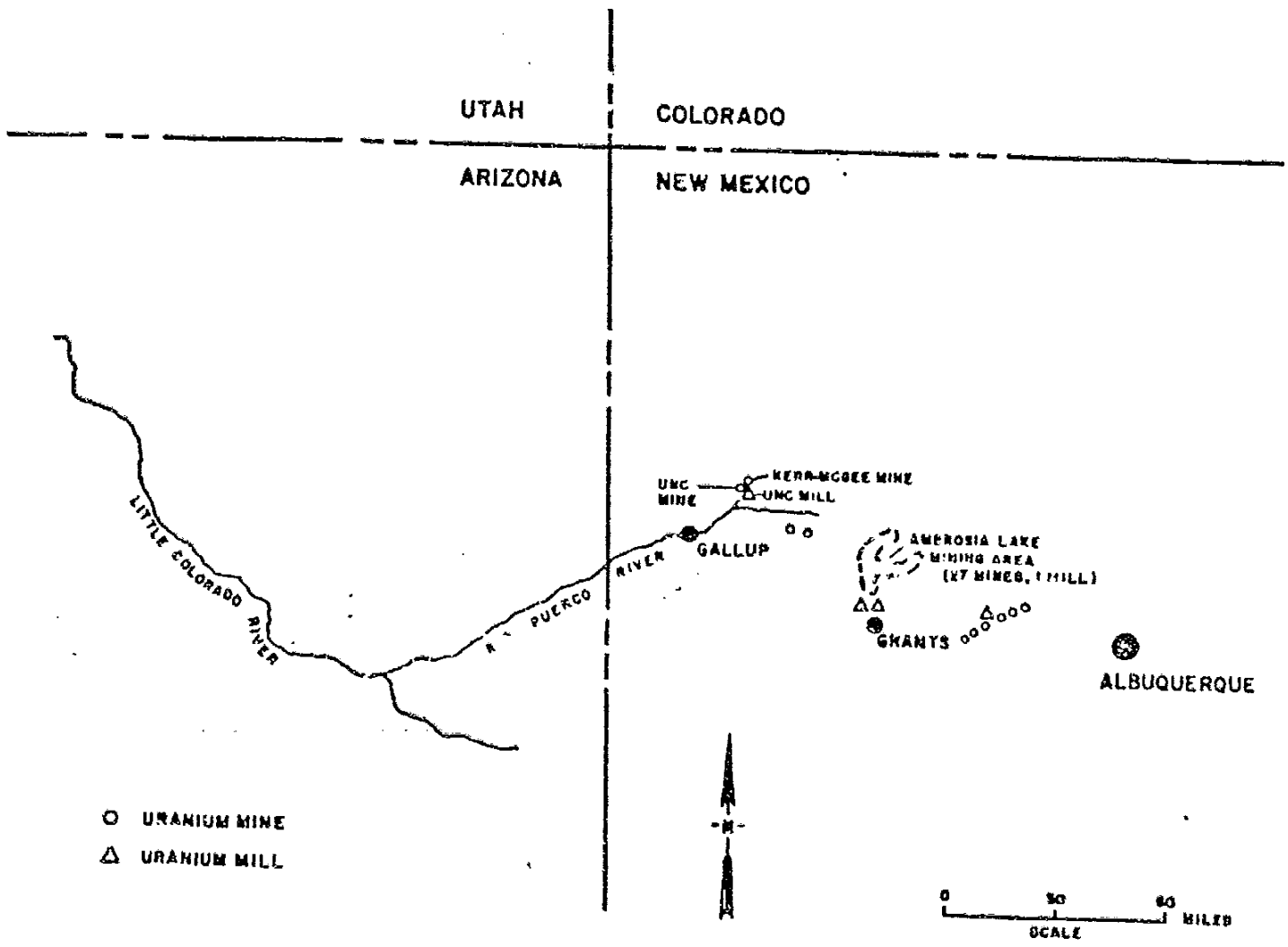
liquid through the underlying substrate and into the regional shallow aquifer has been suggested as a plausible source of contamination, but has not been adequately evaluated (3). UNC has also continuously released mine dewatering effluent into the pipeline arroyo at a rate of 1,400 gallons per minute since 1968. Before 1975 this effluent was not treated; after 1975 it received precipitation treatment for removal of Ra-226.

Radionuclides are also released into the river system through the dewatering of the Kerr-McGee uranium mine located 1 mile north of the UNC mill (Figure 1). During usual mining operations, approximately 3,600 gallons per minute are released into the pipeline arroyo and subsequently to the Rio Puerco. The continuous release of dewatering effluent began in January 1972, and--along with the UNC mine effluent--was responsible for transforming the downstream portion of the Rio Puerco from a sporadically dry riverbed into a continuously flowing stream. Radionuclides in dewatering effluent from both mines have contributed to the current levels of background radiation along the river system. The Kerr-McGee mine began Ra-226 precipitation treatment of its dewatering releases in 1974, but NMEID data indicate that treatment has been incomplete on many occasions.

The monitoring and regulation of the UNC mill falls under the jurisdiction of the NMEID by agreement with the NRC. As partial fulfillment of its license agreement, UNC is required to monitor radionuclides and other chemicals in river

water and plant effluent. NMEID also routinely monitors these parameters in samples from local wells and from the Rio Puerco downstream from the plant site. Because the Kerr-McGee mine is not associated with a mill, its effluent is not regulated by NMEID, but by the EPA. However, EPA's attempt to regulate releases has been contested in court since 1974, and Kerr-McGee has treated its effluent for Ra-226 on a voluntary basis. In 1980 the courts supported EPA's regulatory power, and plans for regulation of the Kerr-McGee dewatering effluent are in progress.

Fig. 1 LOCATION OF UNITED NUCLEAR CORPORATION (UNC) MINE AND MILL, NEW MEXICO, 1979

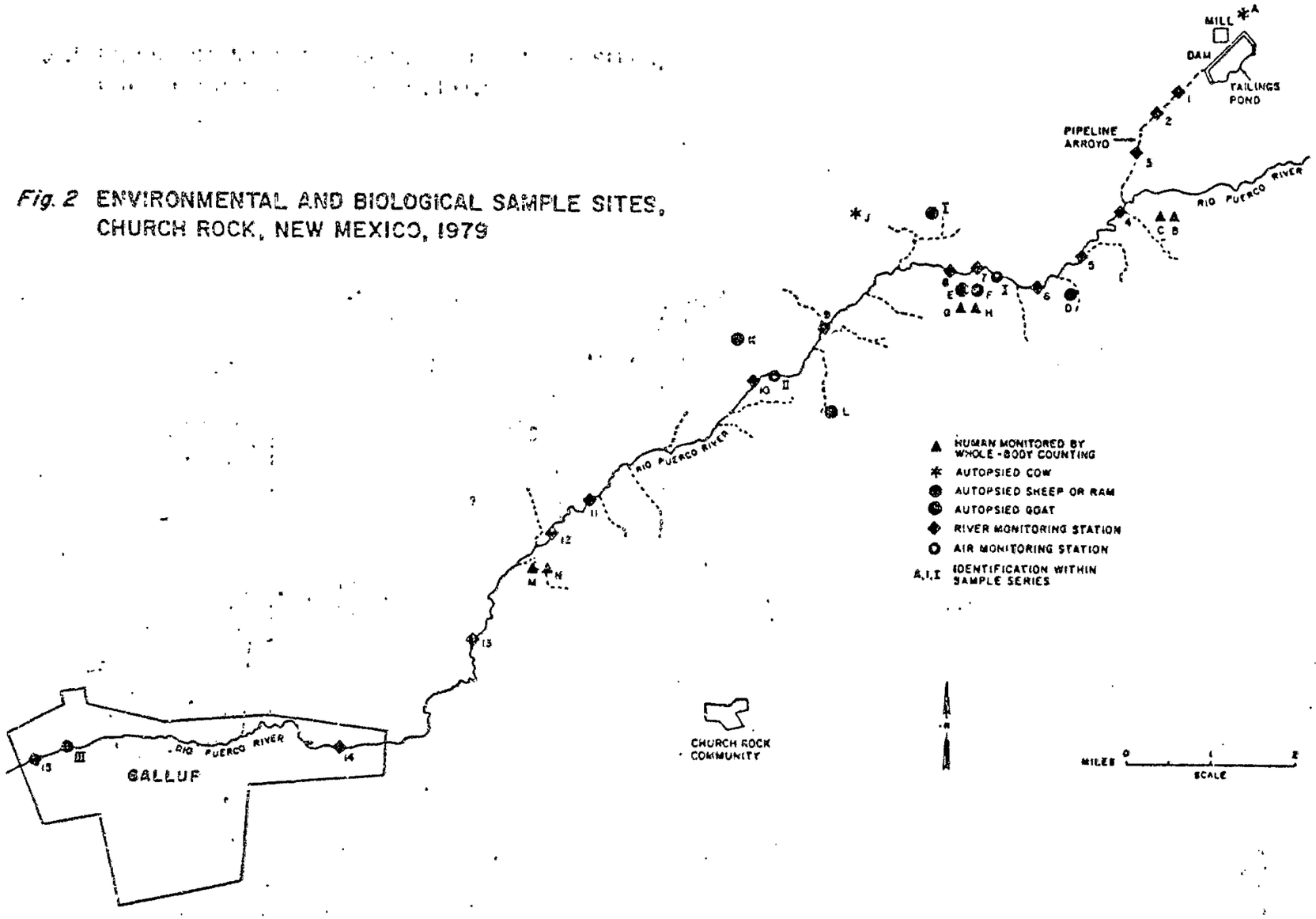


METHODS

Water Monitoring

Immediately after the tailings dam breach, the NMEID began a program of environmental monitoring to determine the extent of contamination by radionuclides, heavy metals, and other chemical compounds (2). Surface water sample sites were located along the Rio Puerco River and its tributary arroyos (Figure 2). Similar locations were also chosen by UNC personnel. Both UNC and NMEID periodically collected samples from river water, sediment, and soil along the river bank between July 16 and August 24, 1979. NMEID and the IHS sampled water from public wells located within 200 feet of the Rio Puerco and NMEID sampled from 5 shallow test wells on the Rio Puerco bank. NMEID samples of river and well water were filtered through a 0.45μ filter and acidified. Water in stagnant pools that were isolated from the main channel was processed in a similar manner. Filtered water samples were assayed for natural uranium (U-nat), Ra-226, and gross alpha emissions by the New Mexico Scientific Laboratory Division. All analyses of other radionuclides and heavy metals were performed by the Eberline Instrument Corporation (EIC), Albuquerque, New Mexico.

Fig. 2 ENVIRONMENTAL AND BIOLOGICAL SAMPLE SITES, CHURCH ROCK, NEW MEXICO, 1979



River Sediment Monitoring

Several groups (NMEID, EPA, NRC, and UNC) performed surveys of soils and sediments to evaluate the extent of tailings slurry deposition and river water leaching into river substrates. The first set of samples (July 19-21, 1979) was taken by NMEID at variable intervals between the UNC mill and Gallup. Cores of various depths were extracted from sediment adjacent to the flowing water and from sites along the first terrace above the active riverbed. Samples of crystalline precipitates were also taken from isolated pools adjacent to the active riverbed. All samples were analyzed for long-lived radionuclides in the U-238 decay series by either the EPA Las Vegas Laboratory (Field Studies Branch, Office of Radiation Programs) or the U.S. Department of Energy's Radiological and Environmental Sciences Laboratory at the Idaho National Engineering Laboratory.

Another set of soil samples was taken by the NMEID (July 31-August 2) from 3 stations between the UNC mill and Gallup. Sediment was collected in 4-foot cores from a variety of streambed and terrace locations. The cores were separated into 1-foot sections and analyzed for Th-230, Ra-226, Pb-210, and U-nat. One core sample was analyzed by the EIC laboratory, and the other samples were processed in a mobile field laboratory operated by Battelle Pacific Northwest Laboratory under contract with the NRC.

Between September 27 and October 5, approximately 1,300 samples were collected by UNC and NMEID personnel. Composite surface soil samples from the first and second terraces above the active riverbed were collected at 1,000-foot intervals from the UNC dam to the New Mexico-Arizona border. Two 3-foot core samples were taken every 5,000 feet over the same length of river. Areas suspected of tailings contamination, such as tributary arroyos, pools, and depressions, were also sampled. Background samples (not contaminated by the spill) were taken from sites 50 to 100 yards from the riverbed, from the pipeline arroyo above the UNC mill, and from the Rio Puerco upstream from the pipeline arroyo. All sediment samples collected by UNC and NMEID were analyzed by Battelle Pacific Northwest Laboratory for Th-230, Ra-226, Pb-210, and U-238.

On September 28-29, a survey for Th-230 was conducted by NMEID along the Rio Puerco between the mill site and Navajo, Arizona. Sediment samples were removed from the first and second terraces, areas suspected of contamination, and background locations.

Air Monitoring

Airborne particulates were collected for radionuclide analysis with a high-volume air sampler by NMEID. The sampler, located on a bank of the Rio Puerco near Gallup, New Mexico (Figure 2), was operated continuously between August 22 and September 28, 1979, and between November 21, 1979, and January 4, 1980. Analysis of the filters (99.9% efficient at 0.3μ) was performed for U-nat, Th-230, Ra-226, Pb-210, and Po-210 by EIC and NMSLD Laboratories. NMEID also collected a single 7-hour sample with a cascade impactor to measure the particle size distribution of the airborne particulates. The corrected 50% efficiency cutoff for particle size in the 4 stages was 17.0μ, 6.9μ, 4.5μ, and 3.3μ, and it was 0.3μ in the backup glass fiber filter. The cascade impactor air sample was taken from a location downwind from a UNC cleanup crew that was working in the Pinedale Bridge area near station 6 (Figure 2).

Assessment of Human Exposure

To clarify the extent of danger the tailings spill posed to humans, we--together with the Church Rock Action Committee and the Navajo Tribal Task Force on Radiation, which were conducting a survey of all residents and livestock near the contaminated waterways--selected 6 Navajos for in vivo assessment of radionuclide concentrations. No human consumption of river water was documented, as

most residents used water hauled from wells nearby. It was concluded, therefore, that acute human exposure to spill materials would come primarily from inhaling particulates as a result of living near the river, herding livestock to the river for watering, and playing or working near the river shortly after the spill. Aerial photographs and information from representatives of the Church Rock community were used to select individuals who were most likely to have been exposed to the tailings contamination. Six persons underwent whole-body counting for 1/2 hour each at the Los Alamos Scientific Laboratory (LASL) on August 23. A 24-hour urine specimen was collected the day following whole-body counting. The residences of the subjects are marked in Figure 2. A description of whole-body counting and urinalysis procedures appears in Appendix 1.

Internal exposure from radionuclides was possible through the ingestion of local livestock that watered in the Rio Puerco. However, internal doses from the animal ingestion pathway are more sensitively evaluated through the measurement of radionuclide concentrations in animal tissue, rather than through human whole-body counting. Radiation exposure caused by the ingestion of food crops irrigated with spill effluent was also considered but deemed unlikely, as only a few gardens were found along the Rio Puerco banks at the time of the spill. The NMEID had also restricted Rio Puerco water use immediately after the spill.

Assessment of Animal Exposure

Sheep, cattle, and goats grazing along the Rio Puerco and its tributaries were the domestic animals most likely to have been exposed to the tailings pond effluent. Several Navajo herds have depended upon this river system as their sole source of drinking water, even during the months following the dam break. The IHS purchased 2 cows, 4 sheep, and 2 goats from the Church Rock area in cooperation with the Church Rock Action Committee. As controls, 1 cow and 2 sheep were purchased from outside the Church Rock area. Because of the difficulty the IHS had in convincing herd owners to sell animals for autopsy, the exposure histories of the Church Rock animals varied considerably (Table 3). However, most Church Rock animals received both acute exposure from the spill and chronic exposure from the mine dewatering effluent. Six Church Rock animals and the 3 control animals were autopsied in an Albuquerque slaughterhouse in late August under the supervision of the New Mexico State Veterinarian and consultants from LASL and the U.S. Department of Agriculture. The other 2 animals were autopsied in a Gallup slaughterhouse by different personnel on September 26.

Tissue samples of muscle, liver, kidney, and bone were removed from each animal that was autopsied. The samples were analyzed for concentrations of U-238, U-235, U-234, Th-232, Th-230, Ra-226, Pb-210, and Po-210. Radionuclide analysis was performed by the EPA Las Vegas Laboratory. A summary of the analytical techniques employed by this laboratory appears in Appendix 2. Concentrations of heavy metals were also measured in 1 exposed goat and 1 exposed sheep by the Veterinary Diagnostic Services branch of the New Mexico Department of Agriculture.

RESULTS

Water Monitoring

Table 1 provides a synopsis of the maximum concentrations of selected radionuclides in unfiltered samples from the Rio Puerco in the days following the spill. With the exception of U-nat, all concentrations were maximal within 48 hours of the dam break. The maximum uranium concentration was detected 16 days after the tailings pond release. Gross alpha concentrations were also highest (1.28×10^5 pCi/l) on the day following the tailings release. Most of the radionuclide concentrations in Rio Puerco water returned to background levels within approximately 20 days postspill. Late August samples revealed

TABLE 3
Animals Autopsied for Radionuclide Analysis, Church Rock, New Mexico, 1979

Animal Number	Slaughter Date	Species	Age (Yrs.)	Location (Figure 2)	Rio Puerco Drinking Water Source		Other Water Source Available
					Before Spill	After Spill	Before Spill
Church Rock							
1	8/22	Sheep	2	K	+	(1)[a]	-
2	8/22	Sheep	8	I	Seldom	Seldom	+[b]
3	8/24	Sheep	5	F	+	(2)	-
4	9/26	Sheep	5	D	+	(6)	-
5	8/24	Goat	1	E	+	(2)	-
6	9/26	Goat	3	L	+	+	-
7	8/22	Cow	2	J	+	+	+
8	8/22	Cow	11	A	-	-	+[c]
Control							
9	8/22	Ram	9				
10	8/22	Cow	3				
Martinez Camp[d]							
11	8/22	Sheep	8				

[a] ( ) Number of weeks Rio Puerco water consumed
[b] Primarily well water from old Church Rock uranium mine
[c] Mine dewatering stream above tailings ponds
[d] Area not exposed to Church Rock uranium mining and milling effluent, but noted for reportedly high levels of background radiation

above-background concentrations of Pb-210, however. Because NMEID surface water samples were taken sporadically from various locations along the Rio Puerco, it is impossible to establish a clear relationship between nuclide concentrations and either postspill elapsed time or distance from the dam breach; however, the surface water data from NMEID and UNC have suggested the following generalizations. Radionuclide concentrations at all sampling sites appear to have fluctuated in a similar manner with respect to time postspill. Likewise, concentrations of all nuclides decreased with time, but the rates of decrease differed among particular nuclides. Certain stations tended to have comparatively high concentrations of specific radionuclides, and these stations were not necessarily the ones closest to the tailings pond.

These general trends are consistent with the dynamics that would be expected in a river system that suspends and redeposits sediments as a function of changing water levels. The same dynamics are also suggested by fluctuations of sediment nuclide concentration in prespill data. This situation is exemplified at station 15 (Figure 2), where Th-230 concentrations rose from 760 pCi/l on August 7 to 1,800 pCi/l on August 8. This resuspension of sediments probably resulted from elevated water levels associated with heavy rains late in the day on August 7.

Five public wells located within 200 feet of the Rio Puerco were first sampled on July 18 and then at 2- to 3-month intervals. Samples taken on July 18 had gross alpha concentrations between 1.7 and 5.0 pCi/l, gross beta

concentrations between 2.8 and 8.9 pCi/l, and sulfate concentrations between 52 and 448 mg/l. These preliminary data indicate no early contamination of groundwater by the river system. Measurements taken in September 1979 from 1 NMEID test well suggested seepage from the contaminated river into groundwater, as sulfate and uranium concentrations showed increases over previous background levels. These concentrations of uranium and sulfate returned to background levels by October, however. Subsequent public well samples have shown no indication of groundwater contamination as of May 2, 1980. Gross alpha, gross beta, and sulfate concentrations for the more recent samples have remained within the ranges reported for July 18, 1979.

Table 4 provides a synopsis of chemical element concentrations in the Rio Puerco and its tributaries for periods before and after the tailings spill. The data indicate significant postspill increases in all elements but barium. Measurements made at station 8 (Figure 2) approximately 3 weeks after the accident suggest that all elements returned to prespill levels relatively quickly--with the exception of pH, which appears to have remained depressed for a comparatively longer period.

Sediment Monitoring

Like radionuclide concentrations in river water, sediment radionuclide levels varied widely over the length of the river and over time postspill. The first set of 40 sediment samples (July 19-21) was collected from 21 stations located over a 38-mile stretch of the Rio Puerco. Concentrations of U-nat showed no consistent pattern with respect to sampling distance from midstream, and they ranged from 0.7-8.4 pCi/g in the active streambed and from 1.4-8.2 pCi/g on the bank between the active streambed and the vertical wall of the river. Th-230 levels were generally higher in samples from the bank than from the streambed, ranging from 18-55 pCi/g in the streambed and from 1.4-160 pCi/g on the bank. Concentrations of both U-nat and Th-230 were highest in areas of crystalline precipitation found along the river bank (U-nat: 30-36 pCi/g; Th-230: 290-490 pCi/g). Concentrations of Ra-226 and Po-210 were measured in only a few of the sediment samples (Ra-226:0.8-4.4 pCi/g, stream and 1.0-1.6 pCi/g, bank; Po-210:1.1-5.7 pCi/g, stream and 1.8-3.8 pCi/g, bank).

Fourteen 4-foot core samples were collected (August 31 - September 2) at stations 6, 10, and 13 (Figure 2) from a variety of locations between the active stream and terraces of the bank adjacent to the river. No major concentration differences were noted among stations, and no general trends were found for radionuclide concentrations in relation to location on stream bank, distance from tailings dam, or depth of core subsample. Radionuclide concentrations in core subsamples were not consistently stratified with respect to depth. Different radionuclide concentrations within each subsample tended to show similar changes with respect to depth, however. The data suggest that sediments are mixed to varying extents depending upon location in the river system and that all radionuclides monitored seem to have similar sedimentation dynamics. Ranges for radionuclide concentrations (pCi/g) in this set of samples are as follows: Th-230, 0.9-210; Ra-226, 0.11-1.6; Pb-210, 0.86-13.0; and U-235, 1.0-2.4.

Approximately 40% of the samples collected in the extensive sediment sampling program carried out between September 27 and October 5 were processed immediately. These samples were analyzed for specific radionuclides; concentration ranges are listed in Table 5. We have not had the opportunity to review the remaining results or the most recent measurements obtained after cleanup procedures. With the exception of U-235 concentrations, levels in both surface and core samples showed no consistent relationship between radionuclide concentration and location in either the first or second terrace away from the active streambed. U-235 concentrations were generally higher in samples taken from the first terrace than they were in those taken from the more distant second

TABLE 4

Chemical Composition[a] of Water Samples From Rio Puerco and Tributaries, Church Rock, New Mexico, 1979

	Before Spill				After Spill			
Date:	5/19/76	5/19/76	3/9/78-5/22/79[b]	4/13/78-5/22/79[b]	7/16/79	7/16/79[b]	7/16/79	8/7/79
Location[c]:	UNC Mine Discharge	Kerr-McGee Mine Discharge	Station 8	Station 15	Station 8	Station 15	Below UNC Dam[d]	Station 8
Elements								
As	50	60	6.1	5.3	--	187.3	1,200	<5.0
Ba	--	1,000	205.5	88.7	--	166.7	316	328.0
Cd	<5	<5	--	--	--	18.1	112	<1.0
Pb	--	--	--	--	--	11.5	1,550	<5.0
Mo	15	14	155.2	113.5	--	10.0	2,099	140.0
Se	16	9	25.2	17.4	--	119.7	505	26.0
U-Nat	1,800	460	620.0	475.0	--	6,974.3	11,355	390.0
V	45	25	--	--	--	4,002.7	47,645	<10.0
Zn	20	40	--	--	--	6,778.6	7,786	75.0
SO4	--	--	176.1	202.7	--	6,545.7	--	--
pH	8.9	9.3	8.3	8.2	1.44	3.3	--	6.8

[a]All concentrations in µg/l; water samples were filtered (0.45µ membrane) unless otherwise specified
[b]Average of multiple samples for specified time period
[c]Stations are identified in Figure 2
[d]Unfiltered water sample

terrace. Likewise, no clear trends existed for any of the radionuclides with respect to depth of sample or distance from the tailings pond dam. Concentrations of Th-230 and Pb-210 in the river system tended to be above levels measured in background samples, and certain samples exceeded background by a factor of 10. Less than 50% of the samples had levels of either Ra-226 or U-235 that exceeded background measurements, and samples that exceeded background concentrations did so by less than a factor of 2 for both radionuclides.

TABLE 5
Preliminary Radionuclide Concentration Ranges for Surface and Core Samples* of Rio Puerco Sediment, Church Rock, New Mexico, 1979

Radionuclide	Surface Composite Sample (pCi/g)	Core Sample (pCi/g)	Background Samples (pCi/g)
Th-230	15-270	5.3-160	19-28
Ra-226	0.23-2.7	1.1-6.8	1.2-1.7
Pb-210	1.8-44	0.84-59	3.0-4.6
U-235	0.18-2.5	1.6-4.0	1.7-2.4

*Samples collected between September 27 and October 5 by UNC and NMEID and analyzed by Batelle Pacific Northwest Laboratory. These concentrations represent only a portion of the samples taken during this period. Samples from areas with concentrated salts are not included in these data.

Sediment samples for Th-230 analysis were collected over a stretch of the Rio Puerco extending 75 miles from the dam into the state of Arizona (September 28-29). Six of the ten samples from the second terrace had higher concentrations than were in the first terrace (adjacent to the active streambed). At the time of sampling, concentrations were highest in a region 19.5-22 miles downstream from the dam. In this set of samples, Th-230 concentrations in river sediments ranged between ~3.5 and 110 pCi/g, with only 5 of the 40 samples exceeding the NMEID cleanup standard of 30 pCi/g.

Immediately after the dam break, UNC began sampling sediment from the river bottom of the Rio Puerco downstream from the dam. Their data for August 17 indicated maximum concentrations of Ra-226 and Th-230 in a region of 1.5-4.8 miles below the dam. Another area of comparatively high concentrations of Ra-226 and Th-230 was detected approximately 8 miles below the dam. Ranges of concentrations for these samples were 1.7-21.8 pCi/g Ra-226 and 11.3-115.7 pCi/g Th-230.

In the months immediately following the spill, UNC was ordered by NMEID to remove tailings from areas known to have high concentrations of Th-230. In the cleanup operation UNC workers shoveled the tailings from these areas. Sediment samples taken after the monitoring and cleanup indicate that Th-230 concentrations from most sample sites in the Rio Puerco have dropped to levels less than the established NMEID cleanup level; however, several areas of elevated Th-230 concentrations remained for months after the spill. Most of these areas showed evidence of raffinate pooling with deposition of white and yellow crystalline material.

Air Monitoring

The NMEID collected and analyzed air samples with a high-volume sampler located at station III (Figure 2) and with a cascade impactor placed near a river bank cleanup operation during conditions of high dust loading. Table 6 presents air sampler radionuclide concentrations and New Mexico's maximum permissible concentrations and Table 7 indicates radionuclide concentrations in particulate samples. One should note that the cascade impactor data in Table 6 indicate that the size of collected dust particles ranged between 0.3μ and 3.3μ --sizes considered to be respirable by humans.

The high-volume composite sample data were used to calculate inhalation doses to the human tissues receiving the highest doses according to the technique of Dunning et al (5). The most conservative clearance classes were used to calculate doses: D for U-238 and Po-210; W for Th-230, Ra-226, and Pb-210 doses to all target organs except respiratory lymph nodes, and class Y for U-238 and Th-230 doses to respiratory lymph nodes. Inhalation rate was assumed to be 15 m^3/d, and duration of inhalation, 1 year; quality factors were selected to give the most conservative dose estimates. Fifty-year dose commitments (in mrem) of 17, 3, 14, and 139 were computed for bone, spleen, endosteum, and respiratory lymph nodes, respectively. Th-230 contributed the greatest single radionuclide dose to endosteum and respiratory lymph nodes, while Pb-210 provided the greatest contribution to bone, and Po-210 to spleen.

Human inhalation doses based on cascade impactor data were calculated in a similar manner. Fifty-year dose commitments (in mrem) of 191, 2, 766, and 3,619 were calculated for bone, spleen, endosteum, and respiratory lymph nodes, respectively. The dose to all target organs from Th-230 was much higher than from all other radionuclides. It should be noted that the inhalation dose from Th-230 is sensitive to the solubility of this radionuclide. If clearance class W (indicating more rapid physiological clearance) is used for the Th-230 dose calculations, doses to the endosteum and respiratory lymph nodes are much lower Likewise, it must be remembered that all doses were calculated for annual continuous inhalation of the measured air concentrations--a situation that would be unlikely, since Church Rock residents do not spend all their time near the banks of the Rio Puerco.

Human Exposure

All 6 of the subjects who underwent whole-body counting were found to have normal amounts of radioactivity, primarily K-40. LASL specifically searched for the presence of radionuclides of uranium, thorium, and their daughters, and found no detectable activity of any of these isotopes in any of the subjects. In addition to the whole-body counts performed on the Church Rock residents, 4 children from Los Alamos were monitored in a similar fashion on the day before the counting of the Church Rock subjects. These particular analyses were performed for control purposes, as LASL had limited experience with the monitoring of children.

Because negative spectroscopy results indicate only the absence of concentrations above the counting system's limits of detection, these limits were computed for the deposition of Th-230 in the lung. The Th-230 detection limit can then be converted to the smallest organ dose that can be detected by the counting system. This minimum detectable dose is thus the largest dose that could be incurred but not detected. LASL estimated the limit of detection for Th-230 deposited in the lungs of a 10-year-old child to be 10 nCi and calculated a minimum detectable dose rate to lung of 4.0 rem/yr and a minimum detectable total lung dose of 7.9 rem. The lung dose from Th-230 was assumed to be the highest organ dose expected from an acute exposure to materials from the tailings pond spill.

Table 6
Radionuclide Concentrations in Air Samples,
Church Rock, New Mexico, 1979

Sample Number	Sample Type*	Collection Date	Total Suspended Particulates ($\mu g/m^3$)	Air Concentration (pCi/m^3)				
				$U(x10^{-4})$	Th-230($x10^{-5}$)	Ra-226($x10^{-4}$)	Pb-210($x10^{-2}$)	Po-210($x10^{-3}$)
1	A	8/22/79	--	--	<13	6.0±4	2.8±1.8	--
2	B	8/23-8/30	119	<4.0	<4.1	2.0±1.6	1.2±0.2	--
3	B	8/30-9/7	80	<2.7	<3.5	1.4±1.4	1.3±0.2	--
4	B	9/7-9/13	107	<3.6	<4.8	2.4±1.9	1.3±0.2	--
5	B	9/13-9/21	80	<2.7	--	--	--	--
6	B	9/21-9/28	141	<4.8	--	--	--	--
7	C	12/79-1/80	119	3.8	19±13	0.71±0.21	24±0.2	5.9±1
8	D	9/12/79	794	42	1300±800	24±2	6.7±1.2	--

NMEID Maximum Permissible Concentration (pCi/m^3)(4)[a]	U-238	U^b-234	Th-230	Ra-226	Pb 210	Po-210
Soluble	3.0	$2.0x10^1$	$8.0x10^{-2}$	3.0	4.0	$2.0x10^1$
Insoluble	5.0	4.0	$3.0x10^{-1}$	2.0	2.0	7.0

*A - One day high-volume sample
B - One week high-volume sample
C - Six-week composite high-volume sample
D - Seven-hour cascade impactor sample (all material deposited on beck-up filter)

[a]Annual average above-background concentrations.
[b]Uranium in atmosphere is assumed to be predominantly the U-234 isotope, as indicated by concentrations in animal organs. In the case of low concentrations of U-235, U mixtures are more toxic via chemical pathways than as ionizing radiation (4).
The MPC for chemical concentrations of U is $7.0x10^{-3}$ $\mu g/m^3$.

TABLE 7
Radionuclide Concentrations in Atmospheric Samples,
Church Rock, New Mexico, 1979

Sample Number	Radionuclide Concentration* (pCi/g)			
	U	Th-230	Ra-226	Pb-210
1	<3.4	--	--	--
2	<3.4	<0.34	1.7 ±1.4	98±14
3	<3.4	<0.44	1.8 ±1.8	158±22
4	<3.4	<0.45	2.3 ±1.8	122±18
5	<3.4	--	--	--
6	<3.4	--	--	--
7	3.21	1.58±1.10	0.60±0.18	50±10
8	5.33	16.3 ±9.7	3.0 ±0.3	85±15

*Error terms represent 2 times the standard deviation of laboratory counting errors

A variety of analyses were performed on the 2 sets of urine samples collected from the 6 individuals chosen for study. Gross gamma activity in both sets of samples did not exceed the detection limit of 35,497 counts/2,000 sec (computed as the mean of 2 background and 2 control samples plus 3 times their standard deviation). Gross beta counts for each sample were not statistically different from controls or detection limits as previously defined. Gross alpha results for all individuals were likewise less than limits of detection, which were established by LASL as 5 counts above background. The concentration of total uranium did not exceed the detection limit that was calculated to be 74 neutron counts (95% confidence limit) and corresponds to a concentration of 0.6 pCi/l.

Colorimetric thorium determinations were all less than the detection limit of 20 µg/l. The comparison of results from alpha scintillation and alpha spectroscopy indicated adequate measurement of thorium radionuclides in the spiked samples. The detection limit (99% confidence) was determined to be 0.1 disintegrations per minute (dpm). When net alpha count data for each specific radionuclide were compared with the detection limit, all samples and controls were below the limit. However, when total dpm were compared, 2 of the 4 samples exceeded the detection limit with count rates of 0.15 and 0.18 dpm. Neither of the control samples exceeded the limit of detection. The net count data from alpha spectroscopy analysis suggest that Ra-224, Th-228, and Po-216 may be responsible for these elevations.

The limit of detection for Th-230 in urine can be used to estimate the associated minimum detectable dose rate. This calculation was performed for a 10-year-old child with a daily urinary excretion rate equal to the average for the Church Rock subjects (709 ml/24 hours) and an exposure to Th-230 that ceased 30 days before the urine was collected (the assumption that provides the most conservative estimate). It was also assumed that thorium was excreted in a manner similar to plutonium. Under these circumstances, radiometric analysis could detect internal deposition of Th-230 that would give a dose rate of 127 mrem/yr to bone surface, and 25.3 mrem/yr to total bone mass.

In the Ra-226 analysis, a single control sample and a distilled water blank were analyzed along with the Church Rock samples. LASL determined the limit of detection by computing the standard deviation of the control, blank, and urine samples with results less than or approximately equal to the levels of the control and blank. The 99% confidence detection limit was 0.34 pCi/l, based on

a mean of 0.144 pCi/l and a standard deviation of 0.064 pCi/l. Only 1 Church Rock sample was in excess of the detection limit. It is interesting to note that 4 of the 6 samples actually exceeded the mean calculated for the detection limit.

Detection limits for atomic absorption spectrophotometry for cadmium and lead were determined to be 0.05 µg/cc. All samples were found to have concentrations of these elements below the detection limits.

Animal Exposure

Appendix 3 presents gross radionuclide concentrations measured in samples from edible animal organs submitted to the EPA Las Vegas Laboratory. These data are corrected for reagent blank activity in Appendix 4. The corrected concentrations are more appropriate for purposes of comparison, as long as one keeps in mind that only 2 reagent blanks were analyzed, that these blanks were processed 3 months after the bulk of the animal samples, and that no goat control was obtained. The fact that radionuclide concentrations were measured in only 11 animals precludes definitive statements regarding representative concentrations for the animal herds in the Church Rock region.

When exposed (Church Rock) and control animals are compared, most radionuclide concentrations for most of the exposed organ samples are higher than those for controls. In particular, exposed animal concentrations of Th-230, Ra-226, Pb-210, and Po-210 are consistently higher than control concentrations. The cow from station A (above UNC tailings dam, but downstream from Kerr-McGee and UNC dewatering effluent) had higher liver and kidney concentrations of Th-230, Ra-226, Pb-210, and Po-210 than the other Church Rock cow. When radionuclide levels in all edible organs are compared, kidney is the organ with the highest concentrations, and muscle the lowest. Concentrations of Pb-210 and Po-210 are consistently higher than other radionuclides.

Radionuclide concentrations measured in bone samples are presented in Table 8. Although radionuclides in bone do not usually enter the human food chain (with the exception of ingested bone marrow and soup), their levels can provide useful data for evaluating pathways of animal exposure. In addition, most radionuclides of the uranium decay series are concentrated in bone to a greater extent than in other organs. This means that bone samples from exposed animals have radionuclide concentrations that exceed detection limits to a greater extent than samples from other organs, and are thus more appropriate for comparative purposes.

The bone concentration data show that for all radionuclides, exposed cows and sheep have higher concentrations than controls. Concentrations of Ra-226 exceed those of Pb-210 and Po-210 in both exposed and control cows. Sheep concentrations of Po-210 are generally higher than Ra-226, with the exception of the 1 exposed sheep that watered from the well located near the old Church Rock mine. There appears to be no consistent relationship between Ra-226 and Po-210 concentrations in the 2 goat samples. It is interesting to note that Pb-210 was detected only in half of the samples and that all of these concentrations were less than those reported for Po-210. When radionuclide concentrations are compared by ages of the animals, Po-210 and Ra-226 appear to be directly related to age for cows and sheep.

The organ concentration data in Appendix 4 can be used to estimate human radiation doses through animal ingestion. Human doses are calculated by assuming particular patterns of organ ingestion and applying factors that convert quantities of ingested radionuclides into various human organ doses (5-6). Table 9 lists human organ doses calculated for the annual ingestion of 78 kg/yr of the specified animal organs (estimated average annual consumption of meat by adults) (1). Calculations for adult meat consumption were chosen to represent the highest (most conservative) organ doses.

TABLE 8
Corrected[a] Radionuclide Concentrations in Animal Bone Samples,
Church Rock, New Mexico, 1979

Animal[b]	Sample Type	Location[c]	Age of Animal (Yrs)	Radionuclide Concentration (pCi/kg)[d]							
				U-238	U-235	U-234	Th-232	Th-230	Ra-226	Pb-210	Po-210
Exposed	Femur	J	2	72.30	3.55	82.21	0	29.83	639.79	71.84	296.97
Exposed	Femur	A	1	37.91	1.21	41.09	1.38	1.16[e]	2,295.09	0	771.27
Control	Femur & Tibia	0	3	0.74	0	1.22[e]	0	32.88	327.41	0	289.69
Sheep											
Exposed	Unspecified	G	5	26.85	2.48	27.07	4.15	9.30	407.26	229.48	1,036.30
Exposed	Femur	D	5	23.69	0	20.44	0	35.75	535.81	0	979.38
Exposed	Femur	K	2	36.85	6.58	3.48	4.27	21.94	579.15	0	667.12
Exposed[f]	Femur	I	8	43.82	1.83	42.36	1.52	20.44[e]	4,278.33	--	1,152.82
Control-1[g]	Femur	C	8	2.88	0	3.93[e]	0	34.92	384.00	866.92	1,745.38
Control-2	Femur	O	9	1.03	0.68	3.48	1.36	17.92	181.59	32.08	763.82
Goat											
Exposed	Unspecified	E	1	16.30	1.10	16.91	1.33	12.65	264.30	153.78	478.96
Exposed	Femur	L	3	5.22	0	6.26	1.62	3.84[e]	522.33	0	260.82

[a]Concentrations based on gross counts with reagent blanks subtracted; negative corrected concentrations are considered to be zero.

[b]Exposed animals taken from herds known to water at Rio Puerco or pipeline arroyo; control animals taken from areas distant from Rio Puerco

[c]Location on Figure 2; O = control animal from area not on map; note that location A is above the UNC dam but downstream from dewatering effluent of UNC and Kerr-McGee mines.

[d]Concentrations based on wet weight

[e]Estimated 3.29 sigma counting error is greater than or equal to calculated concentration.

[f]Animal exposed primarily to well water from old Church Rock uranium mine

[g]Sheep from area not exposed to Church Rock uranium mining and milling effluent, but noted for reportedly high levels of background radiation

TABLE 9
Human Organ Doses[a] (50-yr Dose Commitment in mrem)
Calculated for Animal Tissue Ingestion,
Church Rock, New Mexico, 1979

Animal[b]	Organ	Location[c]	Total Body	Bone	Liver	Kidney	Endosteum	Spleen
Cow								
Exposed	Muscle	J	0.5	0.6	0.9	4.9	0.4	8.3
Exposed	Muscle	A	0.7	0.7	2.0	11.3	0.3	19.4
Control	Muscle	O	0.3	0.5	0.5	1.9	1.3	3.2
Control	Cube Steak[d]	P	0.3	0.5	0.5	2.4	0.2	4.1
Exposed	Liver	J	7.4	15.5	8.5	41.7	9.1	70.2
Exposed	Liver	A	34.5	85.4	24.2	101.6	61.4	168.7
Control 1	Liver	O	2.0	3.3	4.0	20.9	3.2	35.8
Control 2[e]	Liver	O	5.9	14.4	5.1	18.7	12.7	31.1
Exposed	Kidney	J	58.2	138.9	41.8	198.5	163.8	332.5
Exposed	Kidney	A	117.6	276.2	100.3	485.4	286.2	819.9
Control	Kidney	O	32.2	78.1	24.8	118.3	37.2	198.9
Exposed	Bone (soup)	A	2.5	6.8	0.2	0.6	27.4	0.9
Exposed	Bone (soup)	J	0.3	0.9	0.1	0.2	0.8	0.4
Control	Bone (soup)	O	0.0	0.0	0.1	0.2	0.4	0.4
Sheep								
Exposed	Muscle	F	3.4	8.4	2.7	9.3	8.2	15.2
Exposed	Muscle	D	15.5	43.0	4.6	10.0	21.7	14.3
Exposed	Muscle	K	0.1	0.1	0.0	0.1	0.1	0.2
Exposed[f]	Muscle	I	1.9	4.8	0.6	1.2	22.1	2.0
Control 1[g]	Muscle	O	0.1	0.2	0.2	1.0	0.1	1.6
Control 2	Muscle	O	0.1	0.2	0.2	0.7	0.8	1.0
Exposed	Liver	D	185.1	518.8	40.5	59.1	379.7	70.3
Exposed	Liver	K	28.0	76.2	11.0	24.9	53.2	37.3
Exposed[f]	Liver	I	16.7	45.4	5.4	14.6	21.5	21.9
Control 1[g]	Liver	O	27.4	75.3	8.6	23.6	35.9	35.8
Control 2	Liver	O	21.2	57.4	7.8	24.9	27.4	39.3
Exposed	Kidney	D	437.6	1,201.4	141.2	385.4	643.0	587.6
Exposed	Kidney	K	26.2	60.7	24.9	114.4	62.8	193.0
Exposed[f]	Kidney	I	67.7	168.5	70.1	173.1	362.0	282.4
Control 1[g]	Kidney	O	55.4	140.6	34.0	148.1	68.3	246.2
Control 2	Kidney	O	21.0	44.6	28.4	124.4	65.7	210.8
Goat								
Exposed[h]	Muscle	E	3.3	7.9	2.5	6.7	34.7	10.9
Exposed	Muscle	L	116.4	312.1	5.7	10.3	1,447.2	14.4
Exposed	Liver	L	122.8	342.1	33.1	67.6	165.3	94.6
Exposed	Kidney	L	296.2	795.5	120.1	425.4	373.4	682.9

[a]Doses calculated for annual ingestion of 78 kg for specific organs (1, 5)
[b]Exposed animals taken from herds known to water at Rio Puerco or pipeline arroyo; control animals taken from areas distant from Rio Puerco
[c]Location on Figure 2; O=control animal from area not on map; P=cube steak from Las Vegas market; note that location A is above the UNC dam but downstream from dewatering effluent of UNC and Kerr-McGee mines
[d]Average between 2 samples
[e]Duplicate sample
[f]Animal exposed primarily to well water from old Church Rock uranium mine
[g]Sheep from area not exposed to Church Rock uranimum mining and milling effluent, but noted for reportedly high levels of background radiation
[h]Pb-210 sample lost; concentration of this radionuclide estimated from Pb-210:Po-210 ratio from appropriate organ

The doses in Table 9 are expressed as 50-yr dose commitments, meaning the doses to the indicated human organs that would accrue over 50 years from the annual ingestion of 78 kg of the specified animal organ or 130 l of soup prepared with bones. If these quantities of meat were ingested over many years, the 50-yr dose commitments for each year would be additive, and the annual doses would increase year by year until they reached a level equal to the 50-yr dose commitment in the fiftieth year of ingestion. As it is unlikely that an individual would consume 1 organ exclusively for 1 year, doses have been estimated for the more realistic consumption of animal organs in quantities proportional to each organ's percentage of the animal's total edible weight. Table 10 presents 50-yr dose commitments for annual consumption based on this approach for the organs of goats, cows, and sheep that yield the highest dose commitments. These calculations are based on the assumption that muscle, liver, and kidney comprise 94.8%, 3.4%, and 1.7%, respectively, of total edible body weight.

TABLE 10

Human Organ Doses* (50-yr Dose Commitment in mrem) from Consumption of Animals Giving Highest Doses, Church Rock, New Mexico, 1979

Animal	Total Body	Bone	Liver	Kidney	Endosteum	Spleen
Cow	3.9	8.4	13.2	22.6	7.3	38.3
Sheep	28.6	79.4	8.2	18.1	4428	26.1
Goat	119.7	321.5	8.6	19.4	1,3847	28.8

*Based on assumption that muscle, liver, and kidney are consumed in proportion to their contribution to total edible body weight (94.8%, 3.4%, and 1.7%, respectively)

When doses from exposed and control animals are compared, or doses from particular animal organs, the data from Table 9 are appropriate. These data indicate that calculated ingestion doses to all human organs are generally higher from exposed animals than from controls. There is a good deal of variability among doses from exposed animals, however. One should also note that doses calculated for ingestion of organs from 1 exposed cow and 1 exposed goat are less than those calculated for control organs. The variability between doses calculated for the duplicate cow liver sample also suggests that large differences must be evident before doses from exposed animals can be said to be higher than those from controls. The highest doses calculated for ingestion of a single animal organ are from the kidney. Doses from ingested muscle are generally the lowest, with the exception of 1 goat muscle sample.

There seems to be no consistent relationship among human organ doses that holds for all animal samples, although spleen and bone doses are usually higher than other human organ doses. The highest human organ doses calculated for ingestion of a single animal organ are to the endosteum from goat muscle ingestion, to the bone from sheep or goat kidney ingestion, and to the spleen from cow kidney ingestion. Doses of these magnitudes would not be expected unless humans were to ingest these animal organs as their exclusive source of animal protein. The data in Table 9 also suggest that boiling bones for soup would result in negligible organ doses to human consumers.

When raw data for ingestion doses are studied, inferences can be made with regard to the radionuclides that contribute the most to total organ dose. For

bone dose, Pb-210 and Po-210 are the major contributors, with the dose from Pb-210 usually exceeding that of Po-210 by a considerable margin. When present, Ra-226 is a substantial contributor to endosteal dose, but many of the samples have no detectable levels of this radionuclide. Pb-210 concentrations consistently give the highest total body doses, and Po-210 and Pb-210 provide the major contributions to liver dose. Concentrations of Po-210 provide the greatest doses to kidney and spleen. It is interesting to note that concentrations of Th-230 contribute mainly to endosteal dose and usually to a small extent.

Heavy metal concentrations measured in 1 exposed goat and 1 exposed sheep are presented in Appendix 5. Comparatively high concentrations of many metals in skin and wool indicate external contamination of the animals. The fact that elemental uranium concentrations were not high in these animals suggests that exposure to dust and sediment of the Rio Puerco has not resulted in elevated systemic levels of the radioactive isotopes of this element. All elemental concentrations are considered to be within normal limits for animals by consultants from the Southern Region Veterinary Toxicology and Entomology Research Laboratory of the U.S. Department of Agriculture.

DISCUSSION

Measured concentrations of radionuclides in river water samples suggest that New Mexico MPCs for Th-230, Ra-226, Pb-210, and Po-210 were exceeded for a period of days after the tailings spill. Water concentrations of all radionuclides dropped significantly over a period of 5-6 weeks postspill, though comparatively high concentrations of Pb-210 and Ra-226 were measured as late as January 1980. It has been difficult to determine whether water concentrations have returned to background levels because of sparse and conflicting data for prespill water concentrations. This point also has made it difficult to clearly prove that MPCs were exceeded, as they apply to water concentrations in excess of background concentrations. Mine dewatering effluent has been considered by the state to be "background" even though concentrations of certain radionuclides are much higher than found in the Rio Puerco upstream from the pipeline arroyo (when there is water in this usually dry river).

Because water concentrations above the MPCs lead to exposure excess primarily through chronic elevation and because radionuclide concentrations in Rio Puerco water have shown a substantial decline since the spill, we feel that radionuclides released to the water of the Rio Puerco by the UNC spill have posed no significant danger to human health. The fact that humans do not consume Rio Puerco water also supports this contention.

River water concentrations of heavy metals and other elements, though significantly elevated above prespill levels, were not high enough to suggest a danger to human health. For the stations that were monitored, many elemental concentrations were below EPA drinking water standards even immediately after the spill. Those concentrations that exceeded the standards, with the exception of selenium, returned to levels below the EPA stardards soon after the spill. Concentrations of arsenic, selenium, and zinc exceeded the New Mexico drinking water standards immediately postspill, but dropped to acceptable levels after 3 weeks. The data of Table 4 do not represent a detailed analysis of river water elemental concentrations; however, when these data are considered along with the fact that Rio Puerco water is not used by residents of Church Rock or Gallup for drinking or irrigation, they suggest little danger to human health. These elemental concentrations have been and will continue to be monitored by the NMEID.

Concentrations of radionuclides in water from wells adjacent to the Rio Puerco are not high enough to suggest shallow aquifer contamination; however, well water in the Church Rock area has been noted for its high background levels

of radionuclides. Eadie and Kaufmann (7) report a range of 0.10-1.40 pCi/l ($\bar{x}$=0.35) for Ra-226 in Church Rock wells. A study of Navajo Reservation wells initiated by the General Accounting Office in July 1979 reports Ra-226 concentrations ranging between 0.1 and 105 pCi/l.

Sediment concentrations are not normally regulated by NMEID, and background levels were not measured before the UNC spill. Roessler et al (8) report ranges of 18.1-96.6 pCi/g Ra-226 and 20.2-83.4 pCi/g U-238 for phosphate rock in Florida (areas with naturally high-background radiation), while Tso et al (9) report ranges of 0.60-1.53 pCi/g Ra-226 for natural radioactivity in southeastern tobacco soils. These concentrations, though measured for soil in other areas, suggest that Ra-226 levels did not exceed magnitudes recorded for high-background areas in the United States.

Following the UNC spill, the NMEID established maximum permissible sediment concentrations of Ra-226 and Th-230 for the purpose of cleanup operations. For Th-230, the NMEID employed the EPA methodology for determining site-specific screening levels. This approach involves modeling sediment dynamics and estimation of inhalation doses associated with environmental concentrations. The Th-230 cleanup concentration, set at 30 pCi/g inclusive of background levels, was estimated to give an annual dose to bone of 25 mrem. The Ra-226 cleanup level, set at 10 pCi/g, was calculated to give an annual dose to bone of 12 mrem. After the spill, sediment concentrations of Ra-226 dropped rapidly to below cleanup levels at all stations (according to UNC data), while Th-230 concentrations remained elevated in certain sections for many months even after efforts by UNC to remove the contaminated sediment. Although Pb-210 and Po-210 concentrations in tailings liquid were comparatively high, sediment concentrations of these radionuclides were not initially monitored, and cleanup criteria were not established until October 1980.

Radionuclides deposited in sediment can give doses to humans by inhalation and ingestion routes. Inhalation doses are predicted to be highest in areas of tailings deposition that have dried and been mechanically disrupted by wind and the activities of humans or animals. The data for inhalation doses from all radionuclides combined suggest that the inhalation exposure route provides less of a dose than does the ingestion route for all tissues except respiratory lymph nodes (which are not considered to be particularly radiosensitive). If Th-230 is in its least soluble form (excretion class Y), comparatively high doses may be obtained from this radionuclide under conditions of high dust-loading. The comparatively large contribution of Th-230 to total inhalation dose indicates the importance of air monitoring in determining acute exposure from tailings during the days immediately following an environmental release. Since comparatively low concentrations of Th-230 were measured in animal tissues, the major source of animal exposure was probably chronic rather than from the dust derived from spill materials. However, the possibility of inhalation exposure and the undesirability of radionuclide buildup in the Rio Puerco river system make the efforts to remove the spill-contaminated sediments from the Rio Puerco well founded. Individuals in the Church Rock area might be able to reduce their yearly intake of radionuclides by avoiding the Rio Puerco banks during dry, dusty periods. Areas known to contain sediment with high radionuclide concentrations might also be marked to discourage public access.

The whole-body counts and urinalyses of Church Rock residents indicate no acute radiation hazard as a result of the UNC spill. The detected levels of total gross alpha counts and Ra-226 in urine from 3 of the residents suggests chronic exposure from a region of high background radiation. Urinalysis for Ra-226 in New York residents indicates these subjects excrete approximately one-tenth of the Ra-226 excreted by Church Rock subjects (10). However, daily urinary excretion of Ra-226 in Brazilians living in areas of high natural background radiation (11) is similar to that of Church Rock residents. Data from

areas of the United States with high background radiation indicate urine concentrations of Ra-226 that are much higher than those from Church Rock (12). In light of these comparisons and the forementioned detection limits, we do not recommend additional whole-body counting or urinalysis for radionuclides.

Concentrations of radionuclides in edible tissue of Church Rock animals suggest that these animals were exposed to higher levels than the controls from other areas. Four cows (2 exposed to mill tailings and 2 controls) from the Anaconda Mill area (approximately 10 miles northwest of Grants, New Mexico) had concentrations of Ra-226, Pb-210, and Po 210 in edible tissue that were higher than those measured in Church Rock cows but similar to the highest concentrations detected in the Church Rock goats and sheep (13). Rabbits collected in the Anaconda Mill area had edible tissue concentrations of Ra-226, Pb-210, and Po-210 that were generally lower than those in Church Rock animals (13). Concentrations of Pb-210 and Po-210 in rural German cows (with no industrial exposure) indicate levels in liver similar to those in Church Rock cows and levels in kidney lower than those in Church Rock and control cows (14).

The animal radionuclide concentrations may be due to chronic exposure to radionuclides in the Rio Puerco water, in soil, on vegetation, and in air. Chronic exposure is supported by the fact that the cow from location A and the sheep from location I (Fig. 2) had comparatively high radionuclide concentrations in edible tissues, but were exposed to mine effluent rather than spill materials. Since cows from Germany and the Anaconda Mill area had edible tissue concentrations of radionuclides comparable with, or in excess of some Church Rock exposed animals, such organ doses can result from exposure to radionuclides in soil and air alone. The tendency of bone radionuclide concentrations to increase with animal age also suggests that Ra-226 and Po-210 have been assimilated chronically rather than from a short exposure as a result of the tailings spill. Radionuclide uptake from chronic exposure is also supported by calculations that show annual dewatering effluent from Kerr-McGee and UNC mines (with 5 pCi/l of Ra-226) to contain an amount of Ra-226 similar to that released in the spill of tailings liquid. Years of chronic exposure to dewatering effluent therefore might lead to animal radionuclide levels that would exceed those expected from the pulse of tailings liquid released in the spill.

Contrasting Ra-226 concentrations with those of Po-210 and Pb-210 may provide insight into routes of exposure. Comparatively low ratios of Ra-226:Pb-210 or Ra-226:Po-210 suggest exposure from radon gas. Elevated ratios, on the other hand, suggest inhalation or ingestion of Ra-226 from tailings or mine dewatering effluent. For Church Rock cows and sheep, the Ra-226:Po-210 ratios in bone exceed those for the controls, suggesting comparatively higher levels of Ra-226 in the Church Rock environment. The Ra-226:Po-210 concentration ratios for both Church Rock and control cows exceed ratios reported for cows exposed only to uranium mill tailings at Anaconda but are considerably less than the ratios reported for Anaconda rabbits exposed to mill tailings (13). The Ra-226:Po-210 ratios calculated for Church Rock sheep are similar to those for cows exposed only to mill tailings.

The highest bone concentrations of Ra-226 were in the cow exposed solely to mine dewatering effluent and the sheep that drank from a well on the site of the old Church Rock uranium mine. These 2 animals also had comparatively high bone concentrations of Po-210--a finding which, along with the Ra-226 levels, suggests that chronic ingestion of mine dewatering effluents could lead to the radionuclide concentrations reported in Church Rock animals that drank from the Rio Puerco. When the bone concentrations reported in cows exposed to Anaconda Mill tailings (13) are expressed in terms of wet weight, these animals are seen to have lower Ra-226 concentrations than the Church Rock animals that drank mine dewatering effluent, but higher concentrations than the Church Rock animals exposed to Rio Puerco water after the spill. The bone concentrations of Pb-210

and Po-210 are higher in Anaconda cows than in all Church Rock animals. This comparison also suggests that the ingestion of mine dewatering effluent is an important exposure route for the Church Rock animals. This exposure would be expected to give higher levels of Ra-226 than of Pb-210 or Po-210. Exposure via inhalation, on the other hand, would be expected to result in comparatively elevated levels of radon daughters, as exhibited in the data for Anaconda animals (13).

In spite of the support for the mine dewatering exposure route, the data collected do not clearly identify the primary route by which Church Rock animals were exposed. Analysis based on a comparison of Ra-226 concentrations with those of other radionuclides may never be fruitful, in light of the fact that the UNC and Kerr-McGee mines have been removing Ra-226 from dewatering effluent over the past few years. Clarification of exposure routes is also hampered by the absence of data for concentrations of Pb-210 and Po-210 in the dewatering effluent of the 2 mines. More environmental and autopsy data are needed to establish the extent of the ingestion and inhalation exposures. Quantification of the inhalation pathway is particularly important because humans may receive similar exposure, the doses from which should be added to those calculated for animal ingestion to assess total radiation exposure.

Regardless of the route of exposure, the major contributions to human ingestion dose come from Pb-210 and Po-210 and not from Ra-226. Current treatment of mine water only removes Ra-226, and this radionuclide is usually diverted from the human food chain by deposition in inedible bone in any case. In considering efforts toward reduction of Pb-210 and Po-210 in animals, the relative contributions to animal radionuclide burden of inhalation, of soil and water ingestion, and of foraging should first be clarified. Human intake from food crops or from eating bone marrow should also be evaluated before decisions are made regarding exposure reduction.

The human ingestion doses that have been calculated for worst case conditions are small in comparison with those cited in regulations and guidelines for radiation doses to the general public. The state of New Mexico and the NRC currently use a modification of the recommendations of Committee II of the International Commission on Radiological Protection (4,15). Both NMEID and ICRP regulations specify MPCs above natural background for chronic ingestion of all fuel cycle radionuclides. The state of New Mexico MPCs are set to limit doses to gonads or blood-forming organs to 500 mrem/yr from all radionuclides provided there is no exposure to above-background external radiation. When exposure is from combinations of radionuclides, their separate concentrations must be combined by equations to determine whether dose limits have been exceeded (4, 15). In terms of the regulations established by NRC and NMEID, no patterns of animal or organ consumption would result in more than 70% of the composite MPC (assuming zero levels of background radiation), once the separate MPCs for water ingestion are modified to reflect meat ingestion.

However, the NMEID may limit quantities of radioactive materials released in air or water if it appears that daily intake of radioactive material from air, water, or food by a suitable sample of an exposed population group, averaged over a period not exceeding a year, would otherwise exceed the daily intake resulting from continous exposure to one-third the MPC for radionuclides, separately or in combination (4). This condition could result from the exclusive ingestion of sheep kidney, sheep liver, or goat kidney if background animal concentrations are considered to be zero, and animal radionuclide concentrations and ingestion rates are truly representative for the exposed Church Rock population.

The Federal Radiation Council (FRC) established radiation protection guides (RPG) that were chosen to represent a balance between the requirements of health protection and the beneficial uses of radiation and atomic energy (16). These

guides were developed both for individuals and for average population exposure. The FRC guides for doses above natural background to individuals are 1,500 mrem/yr to the thyroid; 500 mrem/yr to the bone marrow; and 1,500 mrem/yr to the bone for radionuclides other than Ra-226. Chronic ingestion of 78 kg/yr of either the organs or the animals with the highest concentrations of radionuclides would not give doses that exceed these guides.

In contrast to the exposure guides for individuals, the FRC's RPG for average whole-body or bone marrow dose to a suitable sample of an exposed population is 170 mrem/capita/yr. The population RPG for thyroid or bone dose is 500 mrem/yr. The FRC recommends these guidelines when evidence is insufficient for the establishment of doses to individuals, a situation that currently exists for the residents near the Rio Puerco. These guidelines could be exceeded by 50 years of continuous consumption of 78 kg/yr of sheep liver, goat kidney, or sheep kidney, but only under the conditions that the highest concentrations of radionuclides found in these animals are representative of the average ingestion concentrations and that background doses are zero. These conditions are unlikely.

The FRC also provided a "graded scale of action" for ingested concentrations of radionuclides that would result in average doses below the RPGs. Only particular radionuclides are specified in these criteria, and Ra-226 is the only radionuclide of relevance to the Church Rock data. Range II of the FRC guidelines suggests the need for quantitative surveillance and routine control. For Ra-226, Range II is 2-20 pg/d above background, which corresponds to annual ingestion doses to bone of 50-500 mrem/yr. Accumulation of Range II quantities of Ra-226 would be possible through ingestion of 78 kg/yr of cow kidney or goat muscle alone or by the consumption of goat organs in proportion to their percentage of total edible body weight. The daily consumption of 0.21 kg of goat muscle by adults would result in doses that would equal the upper limit of Range II, if the goat with the highest muscle concentration of Ra-226 were representative of the goat herds in the Church Rock area. This is probably not the case, as the highest goat muscle concentration of Ra-226 exceeds concentrations of this radionuclide in all other samples by at least a factor of 9.

The FRC recommendations for Range II quantities include efforts to limit environmental release, to provide accurate estimation of population exposure and its variation with time and location, and to monitor the detection of sharply rising ingestion doses. These recommendations also emphasize the need to determine methods of dose reduction for the segments of a given population that are most likely to receive the highest doses.

The EPA has adopted regulations (40 CFR 190) (17) for the uranium fuel cycle that will apply to uranium mills (but not mines) beginning December 1, 1980. These regulations pertain to the exposure of the general public as a result of planned discharges of all radionuclides with the exception of radon and its daughters. The dose limits for these conditions are annual 50-yr dose equivalents of 25 mrem to the whole body, 75 mrem to the thyroid, and 25 mrem to any other organ for any member of the public. The dose limits are not applicable to the exposure at Church Rock for a number of reasons: exposure via animal ingestion appears to be in part from mine releases; any exposure from the mill tailings spill would be exempted because of the accidental nature of the release; and the radionuclides responsible for the largest human doses (the radon daughters Pb-210 and Po-210) are excluded from the regulation.

In light of the above regulations, persons whose meat consumption comes from the exclusive ingestion of goat; of goat, sheep, or cow kidney; or of sheep liver could receive organ doses exceeding some guidelines. The fact that premining background levels of radionuclides were never measured obviously creates a dilemma. However, even if background levels for animal tissues are considered to be zero, the calculated ingestion doses would not result in violations of

existing state laws. Since the butchering of local animals would typically result in consumption of animal organs in proportion to their contribution to the edible weight of the animal, it is unlikely that many, if any, local residents have ingested quantities of radionuclides that have exceeded the guidelines. However, these judgments are based on data from a small number of exposed Church Rock animals that may not be representative of the entire livestock population. The doses calculated are consistent with levels that deserve additional monitoring and investigation. Until such monitoring provides further information, local residents should be given complete and clear explanations of these data and be informed of methods by which they can minimize their exposure to environmental radionuclides.

Because animal kidney and liver samples contain the highest levels of radionuclides in edible tissues, human exposure can be reduced by minimizing the consumption of these organs. One goat contained high muscle concentrations of Ra-226 without showing similar elevations in muscle Pb-210 and Po-210 or in bone Ra-226. We are reluctant to recommend that goat meat not be consumed on the basis of such scant information that might have resulted from a sampling or laboratory error. Obviously, the evaluation of radionuclide concentrations in goats deserves immediate attention. Reducing human consumption of goat muscle would be consistent with our recommendations regarding kidney and liver, but this recommendation differs from the former in terms of supporting data. If new data suggest high concentrations of Ra-226 in goat muscle, then we would certainly recommend reduced consumption.

RECOMMENDATIONS

1. Since the clarification of the human health effects of the UNC spill took many months; we support efforts by all governmental agencies to improve emergency response capabilities. Because it is difficult for the public to be fully and continuously informed, we recommend that representatives of residents from the area affected by a uranium fuel cycle accident, as well as nongovernmental scientists, be invited to participate in all deliberations over impact analysis.
2. To enhance New Mexico's efforts to clarify background radiation levels and to improve its capacity for responding to radiation emergencies, we recommend the expansion of the analytical capabilities of the state laboratory responsible for radionuclide analysis of environmental samples.
3. Much insight into proper approaches to assessing health and environmental impacts of uranium mill accidents has been accumulated through the experience of the UNC spill. This insight should be incorporated into plans for response to future mine and mill accidents by all governmental agencies concerned.
4. Rational efforts to limit radionuclide doses to Church Rock residents depend on clarification of average animal radionuclide concentrations and of exposure routes of livestock eaten by the general population. To this end, we recommend that a surveillance program be established for radionuclide concentrations in Church Rock animals.
5. Although current regulations concerning radionuclides in animals are not being violated, the health risks of low-dose radiation are not completely known. We recommend that Church Rock residents be informed of ways to decrease their exposure to radionuclides, not eating liver and kidney of local animals, not drinking well water known to contain high background levels of radionuclides; and avoiding the banks of the Rio Puerco when the banks are dry and dusty.
6. Since most state regulations are based on elevations of radionuclide concentrations above background levels, we support further study of such

background concentrations in air, water, and food near all uranium fuel cycle facilities.

7. The size of the human population exposed to the UNC spill and to routine mine and mill discharges is very small, and the human doses from these exposures are comparatively low. These 2 conditions make epidemiologic studies of mortality and morbidity insensitive in detecting possible health consequences of radiation exposure (18,19,20). We therefore think that a cancer registry for Church Rock residents alone would be of no scientific value in attempting to alleviate their concerns.

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APPENDIX I

Methods for Human in vivo Radionuclide Analysis

The gamma spectroscopy counting system used for whole-body counting was housed in a low-background chamber and included the following arrangement of equipment. Twin phoswich detectors (each composed of a sandwich of sodium and potassium iodide phosphors) were placed on the chest region for monitoring low-energy photon emitters such as uranium, thorium, and their daughters. A CsI detector was placed over the chest region to monitor the more energetic photons from such emitters as K-40, Cs-137, uranium, thorium, and their daughters. A hyper-pure germanium (HpGe) detector was placed over the liver region for monitoring low-energy photon emitters of the uranium decay series. A large-volume lithium drifted germanium [Ge(Li)] detector was placed under the subject to monitor the whole-body radiation from energetic photon emitters of the uranium decay series, K-40 and Cs-137.

Urine samples were assayed by the Los Alamos Scientific Laboratory (LASL) for gross alpha, beta, and gamma activity. Gross gamma activity was measured with a 4" x 4" NaI detector over an energy range of 0-2.0 MeV. For each individual, the entire spot urine sample was counted and a 400-cc aliquot of the 24-hour collection was assayed. Each sample was counted for 2,000 s. Gross beta activity was measured with a 2π windowless gas flow proportional counter, using planchets plated with 0.5 cc aliquots of sample. An energy range of 0.167-1.7 MeV was examined for a counting time of 60 min, with a counting efficiency of 61%-71%. Gross alpha activity was measured by packaging the same planchet samples with a ZnS-coated mylar film and counting with a scintillation counter at an efficiency of 49.4 $\pm$ 1.1% for 1,000 min.

Uranium was analyzed by neutron activation and delayed neutron counting of U-235 in 25-cc aliquots of each urine sample at the LASL Omega West Reactor Facility. Thorium was analyzed colorimetrically and radiometrically using 200-ml aliquots of the 24-hour urine samples. The colorimetric method employed arsenazo-III dye as a chromagen for thorium. The radiometric analysis involved precipitation of metals from aliquots (125-500 cc) of urine from 4 individuals by the oxalate method, ion exchange, and subsequent electroplating onto stainless steel planchets. Misplacement of urine samples resulted in inadequate volume for 2 individuals. The samples from Church Rock subjects were compared with control samples composed of pooled urine from LASL employees not exposed to radionuclides and with a solution spiked with Th-230 and decay products of the Th-232 series. The planchets were counted in the alpha scintillation system described previously. After alpha scintillation analysis, planchets were counted for 1,212 min in an alpha spectrometer gated for 2.9-7.3 MeV in order to pick up Th-230 and the Th-232 series (excluding Po-212).

The 24-hour urine samples were also analyzed for radium-226 using a de-emanation technique. Aliquots of 200 cc were allowed to sit for 32 days to permit equilibrium between radium-226 and radon-222. The radon-222 was then isolated and counted for 900 min. Aliquots of the 24-hour urine samples were also analyzed for elemental lead and cadmium by atomic absorption spectrophotometry at the LASL facility.

APPENDIX 2

Methods for Detecting Radionuclides in Animal Samples

The procedure employed for analysis of tissue and bone samples was an adaptation of a method developed at the EPA Las Vegas Laboratory (Field Studies Branch, Office of Radiation Programs) for collecting the naturally occurring uranium-238 series of radionuclides on polystyrene air filters. In this procedure samples were first wet-ashed with a combination of nitric acid, sulfuric acid, and hydrogen peroxide to prevent volatilization of polonium and lead. The bone samples were decomposed with only nitric acid and hydrogen peroxide to avoid the precipitation of large quantities of calcium sulfate with sulfuric acid. Samples of bone were also boiled in 1.5 l of water for 8 hours to simulate the leaching that would take place if bones were used in the preparation of soup. After the destruction of organic material, samples were treated with hydrofluoric acid to dissolve and volatilize silicates and then treated with hydrochloric acid. Samples were split into 2 equal fractions for sequential uranium, polonium, and thorium separation by anion exchange and for sequential radium and iron separation by precipitation techniques. Before decomposition, all samples were spiked with both U-232, Po-208, and Th-234 tracers and a stable lead carrier to determine the chemical recoveries of these elements in the separation procedures.

In the anion exchange separation each sample was diluted with 9M HCl and passed through an anion exchange column which removed uranium and polonium. Absorbed iron was eluted with HI, and uranium and polonium were sequentially eluted with dilute HCl and concentrated HNO_3, respectively. The original eluate (containing thorium) was diluted with 7M HNO_3 and passed through the column again to remove the thorium. The thorium was then eluted with 9M HCl. The uranium and thorium fractions were converted to sulfates and electroplated on stainless steel discs. The polonium fraction was converted to its chloride salt and spontaneously plated on a nickel disc. The 3 discs were counted on an alpha spectrometer to measure U-234, U-235, U-238, Th-230, Th-232, Po-210, and the U-232 and Po-208 tracers. Thorium recovery was determined by beta counting for Th-234.

Radium-226 and Pb-210 were separated by precipitating $Ba(Ra)SO_4$ in the presence of HCl and then precipitating $PbSO_4$ after removing the chloride anion by evaporation. The concentration of Ra-226 was determined by the classical emanation technique. Lead-210 was determined by allowing its Po-210 daughter to ingrow for a period of approximately 3 months, followed by spontaneous plating of the polonium on a nickel disc and analysis by alpha spectrometry. For Pb-210 analysis, the lead recovery was determined by atomic absorption analysis of the added lead carrier, and the recovery of the ingrown polonium was determined by measuring a second Po-208 spike added at the time the Pb-210 was isolated.

Practical detection limits for a typical analysis can be estimated from the reagent blank analyses performed during this study. These limits, which appear below, are expressed on per sample, per 100 g tissue, and per 5 g bone bases. Results below these limits should not be considered significant.

Detection Limits for Radionuclide Analysis of Animal Samples, Church Rock, New Mexico, 1979

	Per sample (pCi)	100 g tissue (pCi/kg)	5 g bone (pCi/kg)
U, Th	0.03	0.3	6
Po	0.3	3	60
Pb	1	10	200
Ra	0.3	3	60

APPENDIX 3

Uncorrected[a] Radionuclide Concentrations in Samples of Edible Animal Organs, Church Rock, New Mexico, 1979

Animal[b]	Organ	Location[c]	Radionuclide Concentration (pCi/kg)[d] U-238	U-235	U-234	Th-232	Th-230	Ra-226	Pb-210	Po-210
Cow										
Exposed	Muscle	J	0.29[e]	0[e]	0.42	0[e]	0.11[e]	0.61	1.7	8.0
Exposed	Muscle	A	0.0017[e]	0.040[e]	0.17[e]	0.047[e]	0.094[e]	0.91[e]	2.6	17.0
Control	Muscle	O	0.13[e]	0.025[e]	0.41[e]	0.067[e]	0.23	0.17[e]	2.1	3.9
Control	Cube Steak[f]	P	0.19[e]	0.044[e]	0.46	0.039[e]	0.06[e]	0.65	2.6	4.5
Exposed	Liver	J	3.6	0.027[e]	3.6	0.015[e]	0.42	0.39	12.0	37.0
Exposed	Liver	A	4.5	0.14[e]	4.3	0.14	1.7	1.9[e]	50.0	135.0
Control-1	Liver	O	0.38[e]	0.029[e]	0.058[e]	0.19	0.18	0.37[e]	7.9	30.0
Control-2[g]	Liver	O	0.29	0.13[e]	0.54	0.067[e]	1.0	0.68	14.0	26.0
Exposed	Kidney	J	11.0	0.21[e]	11.0	0.33[e]	0.43[e]	8.0	63.0	265.0
Exposed	Kidney	A	3.7	0.29	3.1	0.042[e]	3.2	11.0	135.0	655.0
Control	Kidney	O	0.19[e]	0.029[e]	0.073[e]	0.15	0.22	0.45[e]	51.0	160.0
Exposed	Bone (Soup)[h]	A	0.027[e]	0.0[e]	0.0049[e]	0.0072[e]	0.013[e]	1.8	2.0	0.86
Exposed	Bone (Soup)	J	0.036	0.0064[e]	0.021[e]	0.0040[e]	0.020[e]	0.27	1.9	0.54
Control	Bone (Soup)	O	0.016[e]	0.0022[e]	0.0045[e]	0.015[e]	0.068	0.16	1.1	0.55
Sheep										
Exposed	Muscle	F	0.51	0.14[e]	0.89	0.42	0.40	1.0	14.0	14.0
Exposed	Muscle	D	0.36[e]	0.030[e]	0.76	0.21	0.41	1.9	41.0[e]	14.0
Exposed	Muscle	K	0.21[e]	0.062[e]	0.099[e]	0.0[e]	0.088[e]	0.23[e]	1.4	1.4
Exposed[i]	Muscle	I	0.15[e]	0.013[e]	0.43	0.17	0.23[e]	2.4[e]	3.3	2.8
Control-1[j]	Muscle	O	0.19[e]	0.0[e]	0.21	0.015[e]	0.031[e]	0.17[e]	2.0	2.4
Control-2	Muscle	O	0.13	0.011[e]	0.15[e]	0.0[e]	0.20	0.38[e]	1.9	2.1
Exposed	Liver	D	2.2	0.23[e]	2.1	0.23[e]	1.2	12.0	308.0	53.0
Exposed	Liver	K	2.0	0.053[e]	2.1	0.0[e]	2.1	1.4	49.0	30.0
Exposed[i]	Liver	I	1.7	0.14[e]	1.5	0.010[e]	0.20[e]	0.94	32.0	18.0
Control-1[j]	Liver	C	0.48[e]	0.041[e]	0.46	0.13	0.11[e]	1.1	31.0	29.0
Control-2	Liver	O	0.19[e]	0.029[e]	0.30	0.078[e]	0.22	0.36[e]	40.0	32.0
Exposed	Kidney	D	4.4	0.21[e]	5.3	1.0	4.8	9.6	740.0	463.0
Exposed	Kidney	K	0.0[e]	0.0[e]	3.4	0.14[e]	2.3	6.0	54.0	159.0
Exposed[i]	Kidney	I	3.2	0.48[e]	2.8	2.2	27.0	9.2	87.0	227.0
Control-1[j]	Kidney	C	0.93	0.058[e]	1.2	0.38	0.77	3.6	107.0	201.0
Control-2	Kidney	O	0.73	0.15[e]	0.88	0.63[e]	4.2	4.2	37.0	172.0
Goat										
Exposed	Muscle	E	0.96	0.011[e]	1.3	0.20	0.99	3.0	5.3[k]	10.0
Exposed	Muscle	L	0.76	0.042[e]	1.2	0.16	0.52	96.0	6.3	12.0
Exposed	Liver	L	0.80[e]	0.067[e]	0.60[e]	0.36	1.4	0.03[e]	227.0	76.0
Exposed	Kidney	L	1.4	0.48[e]	0.77[e]	1.6[e]	0.81[e]	7.8	525.0	549.0
Reagent Blank $\bar{x}$(n=2)			0.017	0.005	0.020	0.247	1.450	0.290	0.006	0.010

[a] Concentrations based on gross counts without subtraction of reagent blanks

[b] Exposed animals taken from herds known to water at Rio Puerco or pipeline arroyo; control animals taken from areas distant from Rio Puerco

[c] Location on Figure 2; O=control animal from area not on map; P=cube steak from Las Vegas market; note that location A is above the UNC dam but downstream from dewatering effluent of UNC and Kerr-McGee mines.

[d] Concentrations based on wet weight

[e] Estimated 3.29 sigma counting error is greater than or equal to calculated concentration.

[f] Average between 2 samples

[g] Duplicate sample

[h] Units of pCi/soup sample of approximately 1.5 l

[i] Animal exposed primarily to well water from old Church Rock uranium mine

[j] Sheep from area not exposed to Church Rock uranium mining and milling effluent, but noted for reportedly high levels of background radiation

[k] Sample lost; Pb-210 concentration estimated from Pb-210:Po-210 ratio from appropriate organ

APPENDIX 4

Corrected[a] Radionuclide Concentrations in Samples of Edible Animal Organs, Church Rock, New Mexico, 1979

Animal[b]	Organ	Location[c]	Radionuclide Concentration (pCi/kg)[d] U-238	U-235	U-234	Th-232	Th-230	Ra-226	Pb-210	Po-210
Cow										
Exposed	Muscle	J	0.21	0.00	0.40	0.00	0.02	0.00	0.00	6.64
Exposed	Muscle	A	0.00	0.01	0.15	0.00	0.00	0.00	0.00	15.56
Control	Muscle	O	0.05	0.00	0.39	0.02	0.14	0.00	0.00	2.57
Control	Cube Steak[e]	P	0.12	0.02	0.44	0.00	0.00	0.00	0.00	3.21
Exposed	Liver	J	3.52	0.00	3.58	0.00	0.33	0.00	5.44	55.69
Exposed	Liver	A	4.42	0.11	4.28	0.10	1.61	0.80	43.53	133.71
Control-1[f]	Liver	O	0.30	0.00	0.04	0.14	0.09	0.00	1.12	28.65
Control-2[f]	Liver	O	0.21	0.10	0.52	0.02	0.91	0.00	7.56	24.71
Exposed	Kidney	J	0.92	10.18	10.98	0.28	0.34	6.85	56.22	263.65
Exposed	Kidney	A	3.63	0.25	3.08	0.00	3.11	9.92	128.67	653.73
Control	Kidney	O	0.10	0.00	0.05	0.10	0.12	0.00	43.64	158.53
Exposed	Bone (Soup)[g]	A	0.01	0.00	0.00	0.00	0.00	1.55	0.55	0.57
Exposed	Bone (Soup)	J	0.02	0.00	0.02	0.00	0.00	0.02	0.45	0.25
Control	Bone (Soup)	O	0.00	0.00	0.00	0.01	0.05	0.00	0.00	0.26
Sheep										
Exposed	Muscle	F	0.39	0.10	0.86	0.35	0.26	0.00	4.14	12.03
Exposed	Muscle	D	0.18	0.00	0.71	0.10	0.20	0.00	25.57	10.92
Exposed	Muscle	K	0.13	0.04	0.08	0.00	0.00	0.00	0.00[h]	0.11
Exposed[i]	Muscle	I	0.07	0.00	0.41	0.13	0.14	1.28	0.00	1.49
Control-1[j]	Muscle	O	0.12	0.00	0.19	0.00	0.00	0.00	0.00	1.25
Control-2	Muscle	O	0.06	0.00	0.13	0.00	0.11	0.00	0.00	0.83
Exposed	Liver	D	2.04	0.17	2.05	0.13	1.01	9.60	293.92	50.18
Exposed	Liver	K	1.93	0.03	2.08	0.00	2.02	0.37	62.96	28.79
Exposed[i]	Liver	I	1.63	0.12	1.48	0.00	0.12	0.00	26.15	16.83
Control-1[j]	Liver	O	0.41	0.02	0.44	0.09	0.03	0.05	44.83	27.77
Control-2	Liver	O	0.21	0.00	0.28	0.04	0.14	0.00	34.06	30.81
Exposed	Kidney	D	4.03	0.08	5.19	0.78	4.37	4.23	708.48	456.70
Exposed	Kidney	K	0.00	0.00	3.31	0.00	1.94	1.59	28.11	153.82
Exposed[i]	Kidney	I	3.03	0.42	2.75	2.10	26.80	6.68	72.20	224.04
Control-1[j]	Kidney	O	0.62	0.00	1.11	0.20	0.40	0.00	80.15	195.63
Control-2	Kidney	O	0.52	0.08	0.82	0.51	3.95	1.16	18.88	168.38
Goat										
Exposed	Muscle	E	0.87	0.00	1.27	0.15	0.89	1.72	0.00	8.50
Exposed	Muscle	L	0.53	0.00	1.13	0.02	0.25	92.62	0.00	8.03
Exposed	Liver	L	0.56	0.00	0.53	0.22	1.11	0.00	206.29	71.86
Exposed	Kidney	L	0.77	0.26	0.59	1.23	0.07	0.00	471.30	538.26

[a] Concentrations based on gross counts with reagent blanks subtracted

[b] Exposed animals taken from herds known to water at Rio Puerco or pipeline arroyo; control animals taken from areas distant from Rio Puerco

[c] Location on Figure 2; O=control animal from area not on map; P=cube steak from Las Vegas market; note that location A is above the UNC dam but downstream from dewatering effluent of UNC and Kerr-McGee mines.

[d] Concentrations based on wet weight

[e] Average between 2 samples

[f] Duplicate sample

[g] Units of pCi/soup sample of approximately 1.5 l

[h] Sample lost; Pb-210 concentration estimated from Pb-210:Po-210 ratio from appropriate organ

[i] Animal exposed primarily to well water from old Church Rock uranium mine

[j] Sheep from area not exposed to Church Rock uranium mining and milling effluent, but noted for reportedly high levels of background radiation

APPENDIX 5

Heavy Metal Concentrations[a] in Animals Exposed to Tailings Spill, Church Rock, New Mexico, 1979

Animal	Tissue	Location[b]	Al	As	Ba	Cd	Pb	Mo	Se	U	V	Zn
Goat	Skin & Wool	L	1,398.00	0.65	13.67	0.02	0.14	<0.20	2.84	<0.50	0.95	45.05
	Muscle	L	10.02	<0.10	<2.00	<0.01	<0.10	<0.20	0.62	<0.50	<0.10	35.35
	Liver	L	2.07	<0.10	<2.00	<0.01	0.10	1.64	2.82	<0.50	<0.10	35.42
	Kidney	L	16.60	0.15	<2.00	0.13	<0.10	0.28	1.26	<0.50	<0.10	14.80
	Blood	L	<5.00	0.18	<2.00	<0.01	<0.10	<0.20	0.75	<0.50	<0.10	5.19
Sheep	Skin & Wool	D	2,070.00	1.64	42.88	0.05	2.80	<0.20	<0.10	<0.50	3.91	34.19
	Skin & Wool	D	1,398.00	0.49	<2.00	2.00	0.10	<0.20	0.14	<0.50	0.34	48.84
	Muscle	D	7.43	<0.10	<2.00	<0.01	<0.10	<0.20	0.14	<0.50	<0.10	26.71
	Liver	D	<5.00	<0.10	<2.00	0.11	0.19	1.12	0.53	<0.50	<0.10	38.91
	Kidney	D	7.43	0.11	<2.00	0.53	0.30	0.27	0.56	<0.50	<0.10	18.75
	Blood	D	<5.00	0.14	<2.00	<0.01	<0.10	<0.20	0.10	<0.50	<0.10	5.19

[a]All concentrations in µg/g wet weight
[b]See Figure 2