Surface Water Pathway

Background

Surface water near the Fernald site became contaminated mainly by releases from the production facilities, runoff from the site, and overflow from the drainage ditches and waste pits. Natural drainage from the site to the Great Miami River occurs via Paddy's Run Creek which flows southward along the western edge of the site and (to a much lesser extent) to the northeast drainage (DOE 1972–1999). Uncontrolled runoff flows into Paddy's Run from several tributaries on site, such as the Storm Sewer Outfall Ditch (SSOD) at the southwest boundary. Before 1955, all runoff from the site went directly to Paddy's Run Creek and the northeast drainage. In 1955, the storm sewer lift station was installed to collect runoff water that had drained into an on-site storm sewer system. This controlled surface runoff and process wastewater were routed to one of the water treatment facilities, treated, and discharged through the Parshall Flume (previously, through Manhole 175) to an effluent line that transports the effluence to an outfall on the Great Miami River, located east of the site about 1.2 miles (2 km) downstream from the city of Ross. Since January 1995, the majority of the controlled runoff and process wastewater has been treated for uranium removal in the advanced wastewater treatment (AWWT) facility before being discharged. Some water was treated at other on-site locations to reduce nitrates, volatile organic compounds, and heavy metals. The major off-site discharge points are shown in Figure 7. The Parshall Flume, which is located near the previously used Manhole 175, is/was the final discharge point for major waste streams leaving the site (DOE 1972–1999).

During the years of production, each of the individual production plants had collection sumps and treatment equipment to remove uranium from the process waste water. Waste water from each plant was sampled and analyzed to ensure that the uranium concentration was within allowable regulatory limits. The filtrate was then pumped to the General Sump for discharge through Manhole 175 to the Great Miami River. The General Sump consisted of several receiving and settling tanks. The settling tanks were used to reduce the concentration of contaminants released to the Great Miami River (Killough et al. 1998a).

Paddy's Run Creek is a small stream with intermittent flow generally from January to May. The remainder of the year, the creek bed is dry except for intermittent flow following heavy rains. The creek joins the Great Miami River approximately 1.5 miles (2.4 km) south of the site and approximately 4.7 miles (7.6 km) upstream of Miamitown, Ohio (DOE 1972–1999).

The Great Miami River is a turbulent, year-round flowing river that is unsafe for swimming. It is not a source of public drinking water between the Fernald effluent outfall and its confluence with the Ohio River; however, some people fish in this part of the river (Killough et al. 1998a).

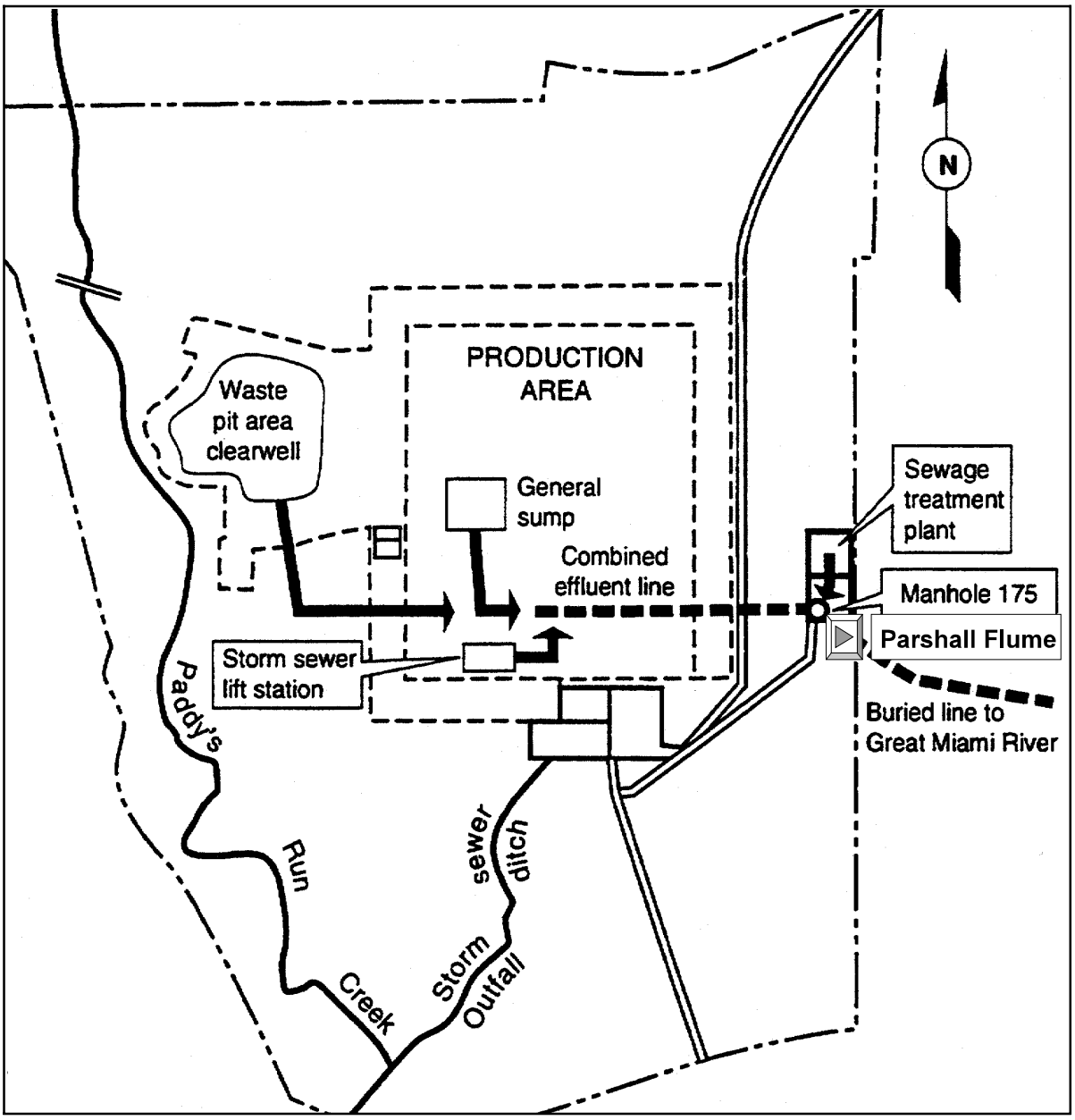

Figure 7. Surface water discharge points from the Fernald site to Paddy's Run Creek and the Great Miami River
(Reference: Killough et al. 1998b)

The northeast drainage (not shown in Figure 7) only receives uncontrolled runoff from the far northeast corner of the site. Only *current* surface water sampling results are available for this location. During many of the sampling events, this drainage system where it exits the site is dry, and no surface water samples are able to be collected. This point contributes very little to off-site surface water releases.

Because the storm sewer lift station was not connected to the production facilities, any chemical or radioactive contaminants in this surface runoff were assumed to be from leaks and spills (Killough et al. 1998a). Throughout the 1950s and 1960s, daily storm sewer samples continued to reflect spills and leaks of radioactive process effluents and chemicals from various portions of the site, including the pilot plant and waste pit area (Voilleque et al. 1995). When the capacity of the storm sewer lift station was reached, water overflowed through the SSOD to Paddy's Run. Flow to the storm sewer system, and ultimately to Paddy's Run Creek, was quite variable, depending on total rainfall and rainfall patterns (Killough et al.1998a). The facility monitored flow and uranium concentrations in the SSOD at a point downstream of its confluence with Paddy's Run Creek. However, there were additional, unmonitored releases to Paddy's Run Creek (because of surface flow from the western part of the site) at points north of its confluence with the SSOD (Killough et al.1998a).

A review of historical sampling data indicates that total uranium is the primary contaminant in effected surface water and sediments; however, other chemical releases were not reported. *Past* estimates of uranium releases to the Great Miami River and Paddy's Run Creek are relatively well known compared to other pathways of exposure (Killough et al. 1998a). The largest releases to the surface water occurred in the early 1960s. During this time, the average quantity of uranium released to the Great Miami River was estimated to be five times greater than the quantity released to Paddy's Run Creek (Killough et al. 1998a). Uranium releases to the river and creek steadily declined from the early 1970s to 1988 as a result of a decrease in production and improvements in the on-site effluent handling system (Voilleque et al. 1995). For example, the facility installed a surface water retention basin in 1986 to reduce the volume of effluent entering the SSOD from the site.

The major radioactive contaminants in surface water releases from the Fernald site were uranium and thorium. Smaller releases of radium 228 and radium 224 (decay products of thorium 232) and of radium 226 (decay product of uranium 238) also occurred. Other radionuclides, such as technetium 99, neptunium 237, and plutonium 239, were released in the processing of recycled uranium, which began in 1962 (Voilleque et al.1995).

The types of uranium used at the facility—and thus discharged to nearby surface waters—changed over the years of production. Natural uranium represented the greatest fraction of uranium in the releases to the Great Miami River from 1951 to 1967 and from 1970 to 1976. Releases of enriched uranium were minor until 1964; between 1964 and 1971, the percentage of enriched uranium fluctuated between 20% and 60% of all uranium released to the river. Only a small fraction of depleted uranium was released to the river until 1977. After 1977, the percentage of depleted uranium rose as high as 90% of the total uranium released to the river (Voilleque et al.1995).

Environmental Data

Beginning in the 1950s, the facility regularly measured uranium concentrations and volumes in liquid effluent released from the site to the Great Miami River. The facility collected composite samples from the liquid effluent released to the river and analyzed them for uranium on a daily basis. Monthly, quarterly, and semi-annual composites were analyzed for other radionuclides, such as radium 226, radium 228, cesium 137 (Killough et al.1998b; DOE 1972–1999).

Before the 1960s, there was no continuous monitoring of water flow through the storm sewer to Paddy's Run Creek (Voilleque et al. 1995). Facility personnel periodically took grab samples downstream of the storm sewer lift station and analyzed them for total uranium. The facility has been monitoring the effluent to Paddy's Run Creek from 1966 to the present. These samples are analyzed for uranium and a variety of radionuclides (Voilleque et al.1995; DOE 1972–1999).

Because the waste streams flowed from each production building on site, contractors for CDC concluded that several uranium compounds were present in the liquid waste effluent discharges to the Great Miami River and Paddy's Run Creek (Voilleque et al.1995). The ratios of various uranium compounds depended largely on the pH of the water. As a result of the high volume of water released from the site, many of the uranium compounds, even those that were relatively insoluble, would have been dissolved or suspended in the waste streams (Voilleque et al. 1995).

Radioactive contaminants (other than uranium) released to the surface water during the years of plant production included decay, fission, and activation products of uranium, thorium, and recycled uranium. For example, releases of thorium 232 and its decay product (radium 228) occurred during thorium processing beginning in 1954. No thorium was processed from 1958 to 1963 or since 1980 (Voillesque et al.1995). There were also "unplanned" past releases of contaminants to the Great Miami River and Paddy's Run Creek from accidental spills, leaks, or discharges from the facility. For the FDRP, CDC contractors estimated the concentrations of uranium and other radionuclides in these "unplanned" past releases by using information contained in incident reports and other written correspondence from 1954 to 1989 (Voilleque et al.1995). Unplanned releases were discharged through Manhole 175 to the Great Miami River or directly to Paddy's Run Creek. Also, certain radionuclides were measured in liquid effluents to the river and creek during various periods of operation. Beginning in 1976, concentrations of plutonium, neptunium, radium, and some fission products (cesium 137, ruthenium 106, technetium 99, and strontium 90) were measured in liquid effluents to the river (DOE 1972–1999). Contractors for the CDC estimated *past* concentrations of uranium and other radionuclides in Paddy's Run Creek and the Great Miami River at downstream locations where humans are likely to be exposed while wading, playing, or swimming based on measurements at the discharge points into the creek and the river, information about "unplanned" releases to these surface waters, and simple dilution models (Killough et al.1998a). The models accounted for the dilution and transport of contaminants in the receiving body of water. ATSDR scientists

reviewed the FDRP documents and agree with the methodology and results for estimating past concentrations in Paddy's Run Creek and the Great Miami River[4]. In 1964, maximum uranium concentrations were released to the Great Miami River with the next highest releases between 1959 and 1961 (Boback 1987; Killough et al.1998b), as shown in Table 13. The 1964 estimated annual concentrations of radioactive contaminants in off-site surface water is shown in Table 14.

Table 13. Maximum past chemical uranium concentrations in off-site surface water

Potential Exposure Point	Uranium Concentration (µg/L)	Year of Maximum Concentration
Paddy's Run Creek	529	1964
Great Miami River	17	1960 and 1964
Key: µg/L = micrograms of uranium per liter of water Source: Killough et al.1998b		

Table 14. 1964 estimated concentrations for radioactive contaminants in off-site surface water in picocuries per liter (and becquerels per liter)

Radioactive Contaminant	Great Miami River	Paddy's Run Creek
Uranium 234	5.99 (2.22×10^{-1})	183.33 (6.79)
Uranium 235	0.28 (1.04×10^{-2})	8.51 (0.315)
Uranium 238	5.70 (2.11×10^{-1})	175.77 (6.51)
Thorium 228	0.00 (1.57×10^{-5})	0.01 (0.0004)
Thorium 230	1.02 (3.78×10^{-2})	31.05 (1.15)
Thorium 231	0.28 (1.04×10^{-2})	8.51 (0.315)
Thorium 232	0.02 (8.24×10^{-4})	0.59 (0.0218)
Thorium 234	5.70 (2.11×10^{-1})	175.77 (6.51)
Radium 224	0.00 (1.48×10^{-5})	0.01 (0.0004)
Radium 226	0.62 (2.31×10^{-2})	27.54 (1.02)
Radium 228	0.87 (3.22×10^{-2})	0.06 (0.0024)
Protactinium 234	0.01 (3.37×10^{-4})	0.28 (0.0104)
Actinium 228	0.00 (9.06×10^{-5})	0.06 (0.0024)
Plutonium 239/240	0.00 (1.59×10^{-4})	0.11 (0.0039)
Source: Killough et al.1998b		

[4] The estimated past concentrations appear low for the Great Miami River compared to current sampling; however, past estimates are for exposure points and current samples are from manhole 175, discharge 001.

Until 1997, DOE routinely measured concentrations of uranium and other radionuclides downstream in the river and in the creek. Since 1997, DOE reports data only for sampling points where the surface water leaves the site and for a background location upstream in the river. DOE has continued to sample sediment in the river and has calculated water concentrations released through the effluent line to the river. The limits set in 1996 for release of uranium in treated water to the river are 600 lbs/yr total uranium with a uranium concentration of 20 µg/L (later changed to 30 µg/L) as a monthly average (DOE 1972–1999, DOE 2002b). Maximum *current* uranium concentrations at potential exposure points in the creek and river are in Table 15.

Table 15. Maximum current chemical uranium concentrations in off-site surface water

Exposure Point	Uranium Concentration (µg/L)	Year the Concentration Was Detected in Surface Water
Paddy's Run Creek downstream of site (sample location W7)	53	1990
Great Miami River Effluent (Manhole 175)	902*	1989
Great Miami River downstream of site (sample location W4, Miamitown)	4.8	1995

Key: µg/L = micrograms of uranium per liter water
* The concentration at Manhole 175 is the highest annual average concentration; the concentrations at Paddy's Run Creek (sample location W7) and the Great Miami River (sample location W4) represent annual maximum concentrations.
Source: DOE 1972–1999

Since 1983, EPA gave the State of Ohio the authority to permit federal sites under the National Pollutant Discharge Elimination System (NPDES) for non-radiological releases into Ohio surface waters (EPA 2004). The releases includes storm water run-off as well as treated effluent. Until March 1990, Ohio permitted six discharge points (two directly to Ohio waters and four on-site) with limited water analyses. In March 1990, the permit covered seven discharge points (one new on-site) and required more analyses on discharges (mainly for metals). ATSDR scientists reviewed the NPDES data from 1990 through 2002 and found no surface water concentrations leaving the site that exceeded ATSDR's CVs. Therefore, using available data, *current surface water releases of chemicals would not cause adverse human health effects and will not be evaluated further*. (Except for uranium, ATSDR does not have sufficient *past* chemical data to make a determination.)

Current maximum annual concentrations of radioactive contaminants at potential off-site exposure points in Paddy's Run Creek and the Great Miami River are presented in Table 16.

Table 16. Maximum current radioactive contaminant concentrations in off-site surface water

Exposure Point	Radionuclide	Concentration in pCi/L (Bq/L)	Year of Maximum Concentration
Great Miami River effluent (Manhole 175, Discharge 001)	Strontium 90	0.40 (0.01)	1990
	Technetium 99	< 1,690 (62.6)	
	Ruthenium 106	< 90 (3.33)	
	Cesium 137	< 11 (0.41)	
	Lead 210	< 8.5 (0.31)	
	Radium 226	< 4.9 (0.18)	
	Radium 228	< 10.6 (0.39)	
	Actinium 227	< 1.0 (0.04)	
	Thorium 228	< 0.3 (0.01)	
	Thorium 230	0.7 (0.03)	
	Thorium 232	< 0.5 (0.02)	
	Thorium 234	267 (9.89)	
	Uranium 234	185.4 (6.87)	
	Uranium 235	11.1 (0.41)	
	Uranium 236	7.0 (0.26)	
	Uranium 238	267 (9.89)	
	Neptunium 237	< 0.21 (0.01)	
	Plutonium 238	< 0.10 (0.00)	
	Plutonium 239/240	< 0.13 (0.00)	
Great Miami River downstream of effluent (sample location W4)	Strontium 90	1.2 (0.04)	1995
	Radium 226	0.29 (0.01)	
	Radium 228	5.2 (0.19)	
	Total uranium	3.2 (0.12)	
Paddy's Run Creek downstream of site (sample location W7)	Radium 226	0.07 (0.00)	1995
	Radium 228	6.00 (0.22)	
	Total uranium	13.5 (0.5)	

Key: pCi/L = picocuries per liter
Bq/L = becquerels per liter
Source: DOE 1972-1999; SED 1998 (updated 2004)

Estimated Exposure Doses

ATSDR scientists evaluated *past*, *current*, and *potential future* exposure to *chemicals* in off-site surface water at the Fernald site. While wading or playing in Paddy's Run Creek or the Great Miami River, exposure may occur via incidental ingestion and direct contact with the skin. Uranium is the only chemical evaluated for this pathway. Therefore, ATSDR scientists assumed that ingestion of surface water was the only route of exposure for *chemicals* in Paddy's Run Creek and the Great Miami River, because uptake into the body (absorption) through the skin is minimal for uranium (ATSDR 1999b). Also, exposure to sediment is not considered for chemical contaminants because it is unlikely that children will accidentally ingest sediments while wading and playing in the creek and river (Killough et al.1998b).

ATSDR scientists evaluated *past*, *current* and *potential future* exposure to *radioactive* contaminants in off-site surface water at the Fernald site. Although uranium and thorium are the primary radioactive contaminants, other radioactive contaminants are also evaluated for this pathway. *Past* exposure doses are estimated in the FDRP and FRAP for nine scenarios described briefly in Appendix D (Voilleque et al.1995; Shleien et al.1995; Killough et al.1998a, 1998b; CDC 1998, 1999). ATSDR scientists reviewed these documents and agree with their methodology and conclusions for past doses from exposures to radioactive contaminants near the site; however, ATSDR also estimated potential doses for exposures to radioactive contaminants in surface waters using the scenarios described below. ATSDR's estimated doses were used to evaluate possible public health implications although surface water releases were not found to be a major contributor to past off-site radiation doses.

ATSDR assumed exposure to radioactive contaminants in Paddy's Run Creek and the Great Miami River occurred via ingestion of surface water, external exposure to radioactive contaminants in surface water, and external exposure to sediments. Children residing near the Fernald site are most likely to play or wade in Paddy's Run Creek in the spring, when it is a flowing stream. Older children and adults may periodically play, wade, or fish in the Great Miami River but only sporadically, because the river can be turbulent and rapidly flowing. ATSDR evaluated two *hypothetical* exposure scenarios. The first scenario assumes exposure to a young child, 2 to 5 years old, weighing 13 kg, who ingests one-quarter of a liter of contaminated water while playing in Paddy's Run Creek five days per week, three months per year (EPA 1999). The second exposure scenario assumes exposure to an older child, 6 to 11 years old, weighing 20 kg, who ingests one-quarter of a liter of contaminated water while playing in the river five days per week, three months per year (EPA 1999).

ATSDR assumed exposure to a child because children, with their immature/developing systems, may have increased sensitivity to uranium's toxic effects. ATSDR does not have direct evidence that children play or have played in Paddy's Run Creek or the Great Miami River. However, 1990 Census data for Butler and Hamilton Counties indicate that 922 persons live within 1 mile

of the Fernald site. Of these, an estimated 110 persons are 6 years of age or younger (as discussed under Demographics). The closest residence was located directly east of the site and is currently located directly southeast of the site. Off-site contaminated areas in Paddy's Run Creek and the Great Miami River are not restricted.

Chemicals

ATSDR scientists calculated chemical doses from uranium exposure in surface water for the two hypothetical exposure scenarios (described above). The scientists compared these estimated exposure doses for ingestion of uranium in surface water to the health-based guidelines to determine if further evaluation was warranted. Additional information about the health-based guidelines for uranium is presented in the Public Health Implications section of this report.

Past Exposure

ATSDR used the estimated concentrations in Table 13 to calculate *past chemical* exposure doses for surface water pathways. The estimated doses for both scenarios are in Table 17 (below).

Table 17. Estimated maximum past uranium (chemical) exposure doses for hypothetical exposure scenarios for surface water pathways

Exposure Point	Estimated Maximum Uranium Concentration (µg/L)	Estimated Exposure Dose* (mg/kg/day)	Health-based Guideline for Uranium (mg/kg/day)
Scenario #1: Younger child playing in Paddy's Run Creek	529	0.002	0.002 (ATSDR Chronic Oral MRL)
Scenario #2: Older child playing in the Great Miami River	17	0.00004	

Key

µg/L = micrograms of uranium per liter water

mg/kg/day = milligrams of uranium per kilogram of body weight per day

* Equations used to estimate doses for this pathway are described in Appendix B—Exposure Doses and Health-Based Guidelines.

The estimated exposure doses do not exceed the health-based guideline for ingested chemical uranium even though very conservative assumptions were used. Further evaluation of exposure to uranium in surface water is not necessary; however, ATSDR scientists evaluated the public health hazard for this pathway, together with other exposure pathways (i.e., groundwater, soil, air, and biota) that contribute to total uranium exposure to nearby residents, in the Public Health Implications section of this report.

Current Exposure

ATSDR scientists used the concentrations in Table 15 to estimate *current* chemical exposure doses for the two hypothetical exposure scenarios for surface water pathways. For exposure scenario #2, incidental ingestion of water in the Great Miami River by an older child, ATSDR used the maximum uranium concentration detected in the Great Miami River downstream of the discharge point to the river. The estimated chemical exposure doses for both exposure scenarios are presented in Table 18 (below).

Table 18. Estimated current uranium (chemical) exposure doses for hypothetical exposure scenarios for surface water pathways

Exposure Point	Estimated Uranium Concentration (in µg/L)	Estimated Exposure Dose* (mg/kg/day)	Health-based Guideline for Uranium (mg/kg/day)
Scenario #1: Younger child playing in Paddy's Run Creek	53	0.0002	0.002 (ATSDR Chronic Oral MRL)
Scenario #2: Older child playing in the Great Miami River	4.8	0.00001	

Key

µg/L = micrograms of uranium per liter water

mg/kg/day = milligrams of uranium per kilogram of body weight per day

* Equations used to estimate doses for this pathway are described in Appendix B—Exposure Doses and Health-Based Guidelines.

The estimated exposure doses for scenarios #1 and #2, under current conditions, are lower than the health-based guidelines for ingestion of uranium; however, ATSDR scientists evaluated the public health hazard for this pathway, together with other exposure pathways (i.e., groundwater, soil, air, and biota) that contribute to total uranium exposure to nearby residents, in the Public Health Implications section of this report.

Radiation

ATSDR scientists calculated three types of exposure doses for radioactive contaminants in surface water: (1) a committed effective dose (whole body), (2) a committed equivalent dose (bone surface), and (3) an external dose to the whole body. The bone surface is the major target organ for radiation effects from ingested uranium and most other radioactive contaminants in the surface water.

Past Exposure

ATSDR estimated *past* exposure doses based on the estimated radioactive contaminant concentrations in the FDRP (Table 14) and on the exposure scenarios described above. (The FDRP exposure scenarios used less exposure time to surface water than ATSDR's scenarios; therefore, ATSDR estimated past doses using the same scenarios as used for current exposures.) Surface water releases were not found to be a major contributor to past off-site radiation doses; however, ATSDR scientists evaluated the contribution of this dose to total radiation exposure of nearby residents in the Public Health Implications section of this report.

Table 19. 1964 estimated exposure doses for radioactive contaminants in surface water for the hypothetical exposure scenarios

Exposure Point	Estimated Committed Effective Dose (whole body) for 1-Year Intake, in mrem (mSv)*	Estimated Committed Equivalent Dose (bone surface) for 1-Year Intake, in mrem (mSv)*
Scenario #1: Younger child playing in Paddy's Run Creek	3.66 (0.037)	82 (0.82)
Scenario #2: Older child playing in the Great Miami River	0.3 (0.003)	12 (0.12)
Key: mrem = millirem mSv = millisievert * Used ICRP 67 and ICRP 72 Methodology and Conversion Factors (ICRP 1993, 1995c). These estimated exposure doses are for exposure points downstream from releases.		

Current Exposure

ATSDR scientists used concentrations in Table 16 to estimate *current* radiation exposure doses for the two hypothetical scenarios for surface water pathways. The concentrations used for the Great Miami River were for Manhole 175, discharge 001 (more complete analyses) and not at an exposure point downstream. Therefore, these doses will appear high compared to past doses and are extremely conservative. Table 20 presents ATSDR's estimated committed effective doses and committed equivalent doses from ingestion of the water, and external doses from the water (submersion) in Paddy's Run Creek and the Great Miami River.

Table 20. Estimated current exposure doses for radioactive contaminants in surface water for the hypothetical exposure scenarios

Exposure Point	Year of Maximum Concentration	Estimated Committed Effective Dose (whole body) for 1-Year Intake, in mrem (mSv)*	Estimated Committed Equivalent Dose (bone surface) for 1-Year Intake, in mrem (mSv)*	Estimated External Dose for 1-Year Exposure, Maximum Concentration, in mrem (mSv)†
Scenario #1: Younger child playing in Paddy's Run Creek	1995	2.62 (0.026)	88.3 (0.883)	1.5E-06 (1.5E-08)
Scenario #2: Older child playing in the Great Miami River	1995	1.9 (0.019)	80.8 (0.808)	3.6E-06 (3.6E-08)

Key:
mrem = millirems
mSv = millisieverts
* Used ICRP 67 and ICRP 72 Methodology and Conversion Factors (ICRP 1993, 1995c).
† Used EPA's Federal Guidance Report No. 12 for Conversion Factors.

Using maximum concentrations of radionuclides in Paddy's Run Creek and Great Miami River sediments, ATSDR scientists determined that external exposure to the skin is insignificant and will not be evaluated further. Potential exposures to radioactive contaminants in surface water and other pathways that contribute to radiological doses at the site (i.e., groundwater, soil, air, biota) are evaluated further in the Public Health Implications section of this report.

Potential Future Exposure

Remedial activities at the Fernald site are expected to continue through 2006. During that time, contaminated soils and debris will be removed from the area and transported to off-site and on-site disposal areas. Since January 1995, most of the controlled runoff and other wastewater released as surface water has been processed through the Advanced Wastewater Treatment System and other treatment facilities. These activities should minimize releases to surface waters and the potential for human exposure to chemicals and radioactive contaminants in the Great Miami River and Paddy's Run Creek. *According to available environmental data, there is no indication that future activities will result in human exposure to contaminated surface water off site of the Fernald site at levels of health concern.* However, if additional information becomes available indicating that contaminants have been released or have migrated to off-site surface waters, this exposure pathways should be re-evaluated.

Groundwater Pathway

Background

Groundwater beneath the Fernald site occurs either as perched water in a glacial overburden, in a sand and gravel aquifer (the Great Miami Aquifer), or in underlying bedrock (to a much lesser extent). Perched water generally migrates from the surface and is trapped between the surface and a dense clay layer. It is shallow, occurs sporadically, and is not a good source of drinking water. There appears to be one or two dug wells in the area just south of New Haven Road that are shallow (approximately 24 to 27 feet deep) (USGS 1984). The Great Miami Aquifer begins about 25 meters (82 feet) below the site and is 38 to 53 meters (125 to 175 feet) thick (DOE 1972–1999); however, the aquifer is not quite as far below the surface in some off-site locations (USGS 1984). Most of the wells in the area are drilled at various depths into this aquifer, but there are several domestic and farm-use wells that are drilled quite shallow (approximately 50 feet or less). Under the site, the groundwater in this aquifer moves predominantly to the east under the waste pits and the former production area; however, it moves to the south in the southern portion of the site (DOE 1972–1999).

In late 1981, the State of Ohio's groundwater sampling results indicated elevated gross beta activity in three wells south of FMPC (Voillesque 1995). In December 1981, this contamination was identified as uranium by National Lead of Ohio (the operator of the plant) and confirmed by the U. S. Geological Survey in August 1982, eventually resulting in the closure of a private well down gradient of the site (USGS 1984). Because of the elevated uranium concentrations, DOE began routine monitoring of private wells near the site in 1982. In 1984, the Radiological Environmental Monitoring Program for monitoring groundwater was formally established, and property owners could request a one-time sampling of residential well water for uranium. The results were sent to the owners, and if the samples showed above-background concentrations of uranium, the owners had an option to participate in the routine monitoring program and were offered bottled water for household use (DOE 1972–1999). In 1986, 24 off-site wells were also analyzed for 16 metals. By 1987, three private wells (#12, #15, and #17) were used for monitoring purposes only. In 1991, a fourth private well (#13) was converted for solely monitoring purposes. By 1996, the Radiological Environmental Monitoring Program had expanded to include more than 30 private wells. (The private well locations are shown in Figure 8.) Wells were sampled monthly or quarterly, depending on their location. In 1996, DOE supplied some residences with public water, but some wells that are not in the uranium plume are still in use today. In 1997, DOE discontinued monitoring all but three former residential wells (#12, #13, and #14) but is currently sampling approximately 140 *monitoring* wells on-site and off-site at various frequencies to determine water quality. These samples are analyzed for uranium and 50 non-uranium groundwater constituents (DOE 1972–1999; DOE 2002).

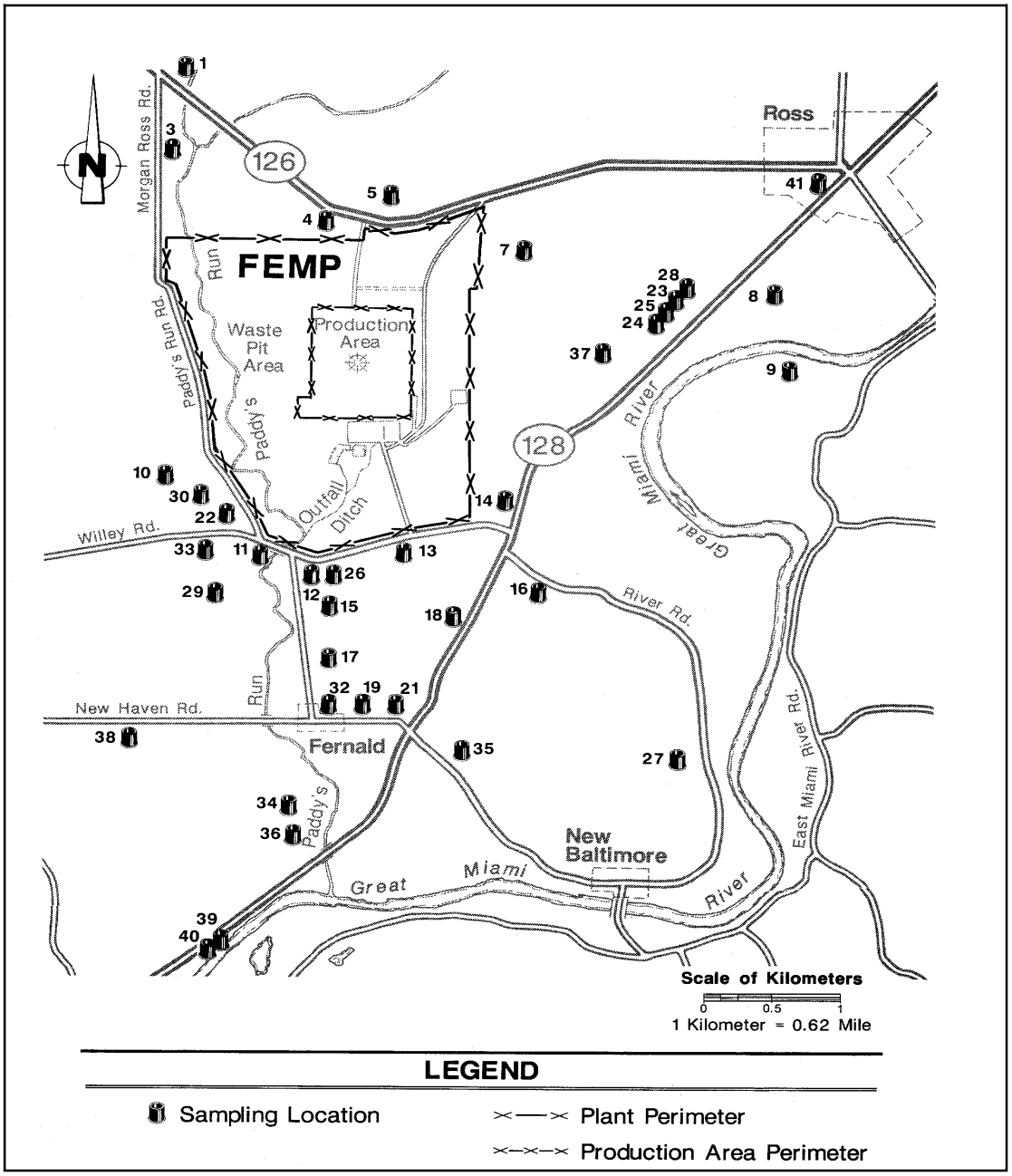

Figure 8. Location of privately owned sampled wells near the Fernald site
(Reference: DOE 1972–1999)

The aquifer apparently was contaminated by on-site leaks and spills, by contaminants migration from surface water releases and run-off, by seepage and migration from various on-site activities and from storage locations.

Although many residences in the immediate vicinity of the Fernald site have relied on privately owned wells supplied by the Great Miami Aquifer as their primary source of drinking water; some residences near the site have used and currently use cisterns as a drinking water source (ODH 1988). Cisterns potentially were contaminated by airborne releases; however, ATSDR reviewed potential doses from drinking cistern water as well as well water in this section.

Since 1986, DOE contractors have been remediating groundwater south of the site. For purposes of remediation, this plume (where uranium concentrations are greater than or equal to 30 micrograms (µg) total uranium per liter (L) of water[5]) is called the "South Plume" (DOE 2002). Additional uranium groundwater contamination exists on-site, but only the South Plume extends outside the site boundary. (Refer to Figure 9.) By 1999, the groundwater remediation program included 11 extraction wells in the *on-site* South Plume area (the "South Field") that began operating in July 1998, 6 extraction wells in the *off-site* South Plume that began operating in August 1993, and 5 re-injection wells that began operating in August 1998. The number and locations of these wells will vary over time as the remedy design for the aquifer evolves. DOE estimates that 68 hectares (169 acres) of the aquifer are contaminated at levels above 30 µg uranium/L of water (DOE 2002). The outermost uranium contour lines in Figure 9 represent the edge of the 30 µg/L uranium concentration plumes.

The primary sources of groundwater contamination *in the South Plume* are historical releases of uranium-contaminated water to the storm sewer outfall ditch (SSOD) and Paddy's Run Creek (Voilleque et al.1995). The SSOD became contaminated by overflow of the site's storm sewer system when heavy rains exceeded the storm sewer lift station capacity. Overflow from the SSOD discharged into Paddy's Run Creek. The Stormwater Retention Basin, which began operations in 1986, greatly reduced discharges of contaminants to the SSOD and Paddy's Run Creek (DOE 1972–1999). This creek also received contaminated runoff from the western portion of the site (Voilleque et al. 1995). Contaminated water from the SSOD and Paddy's Run Creek seeped into the underlying groundwater. Releases from the waste pits in the waste storage area and contaminated runoff and liquid releases to the Pilot Plant Drainage Ditch contributed to another area of groundwater contamination (Voilleque et al.1995). This second plume (the Pilot Plant Drainage Ditch Plume) starts just south of the waste pit area and extends to the east under part of the former production area (DOE 2003c). (Refer to Figure 9.)

Environmental Data

ATSDR scientists used sampling data from 1981 to the present to evaluate potential human exposure to contaminants in privately owned drinking water wells near the Fernald site. A

[5] The total uranium groundwater remediation level was changed from 20 µg/L to 30 µg/L in 2001 to reflect the EPA's adopted Safe Drinking Water Act Final Maximum Contamination Level (MCL) for uranium going into effect December 8, 2003 (DOE 2000a).

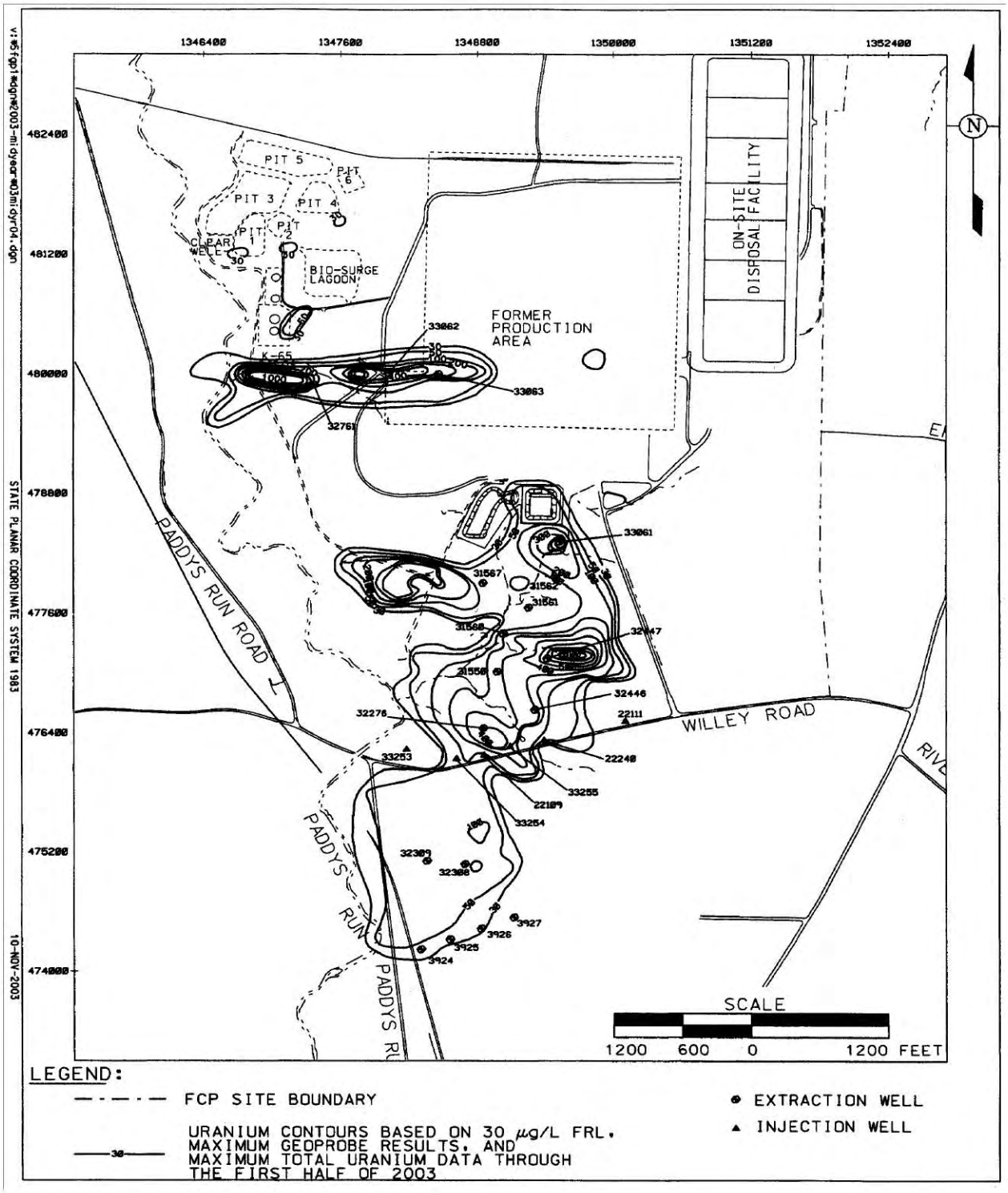

Figure 9. Location of on-site and off-site groundwater contamination (uranium) at the Fernald site (Reference: DOE 2003)

limited number of samples, collected from privately owned wells between 1981 and 1996, were measured for concentrations of metals, elements, and nitrates. *Most samples were analyzed for concentrations of total uranium only*. A summary of the sampling programs and data used to evaluate groundwater pathways for privately owned wells is provided in Table 21.

Groundwater modeling indicates that private wells south of the site may have been first impacted by the South Plume some time after 1962 (Voilleque et al. 1995). No private well sampling was conducted before 1981. Therefore, contractors for CDC *estimated* uranium concentrations by using measurements (sampling) of water from the SSOD and Paddy's Run Creek, known quantities of uranium releases to the SSOD, and empirical modeling techniques in the FDRP (Voilleque et al. 1995). Maximum uranium concentrations were found in surface water samples from the SSOD and Paddy's Run Creek during the 1960s (Voilleque et al. 1995). The highest *estimated* yearly median uranium concentration (lower bound) to the highest *estimated* yearly 95th percentile (upper bound) uranium concentration for residential wells 12 and 17 are 918 µg/L to 4,144 µg/L (Voillesque et al.1995). Although ATSDR scientists used these estimated concentrations to calculate potential doses, they appear to be extremely conservative. The actual concentrations were probably lower due to filtering during migration from the surface water to the groundwater and to the amount of dilution in the aquifer before the contaminants reached these wells.

Of the privately owned wells that were routinely *sampled*, only four wells (numbers 12, 13, 15, and 17) had uranium concentrations above the *proposed* EPA drinking water standard of 20 µg/L (Voilleque et al.1995; DOE 1972–1999) and the *final* EPA drinking water standard of 30 µg/L that went into effect December 8, 2003 (EPA 2000a). These wells are near each other and are directly within the South Plume. Well 15 historically had the highest concentrations, but it is uncertain if this well was ever used as a drinking water source. Wells 12, 15 and 17 were not used as drinking water sources after 1986. Well 13's water has contained uranium concentrations above 20 µg/L since 1992; however, it has not been used as drinking water since 1991. Maximum yearly (estimated and measured) uranium concentrations are presented in Table 22.

Table 21. Summary of sampling programs and activities that include private off-site wells

Date of Sampling	Program or Activity	Parameters Analyzed in Well Water	Wells Sampled and Comments
1981 to 1996	DOE's Radiological Environmental Monitoring & Comprehensive Groundwater Monitoring Programs (routine); sampling by owner's request; and State Route 128 (SR 128) study, in 1990 (DOE 1972–1999)	Monthly, quarterly, or annual analyses for total uranium (U) & 16 metals & elements (Primary & Secondary Drinking Water Standards); monthly analyses for nitrate-nitrogen (1983 to 1985).	Routine monitoring of up to 37 off-site private wells; special sampling conducted on a one-time basis at owner's request; SR 128 study sampled 18 wells and some cisterns along a 2-mile stretch of SR 128 south of the site for total U. Monthly samples analyzed for nitrate (1983 to 1985) in up to 26 wells & for 16 metals (1986 to 1995) in up to 37 wells.
1982	Sampling by US Department of the Interior, Geological Survey (USGS 1984)	Sampling off-site wells; analyzed for dissolved U; some samples analyzed for gross alpha & beta, metals, organics, & 15 other constituents.	USGS sampled ~ 30 wells in the area; 3 had elevated uranium concentrations but no other constituents at levels of health concern. Maximum concentration (430 µg/L) in well south of site; not attributed to waste storage area.
1986	Sampling by International Technology, Inc. (IT 1986)	Sampling off-site wells; analyzed for total U. Sampling of six cisterns in various directions from site; analyzed for radioactive materials.	IT collected 17 well samples. Five south of site had U concentrations above background (0.8 µg/L), maximum concentration was 269 µg/L. One well sample from east of site was 1.8 µg/L. No cistern data.
1985 to 1988	Sampling conducted by the Ohio Department of Health (ODH 1988)	Sampling of off-site private wells and cisterns, mostly from 1985 to 1986. Samples analyzed for gross alpha, gross beta, and total uranium. 14 wells were sampled for radon.	ODH sampled 246 private wells, 54 cisterns, 1 public water supply, & 2 industrial water supplies; 3 wells had U above background & drinking water standard; 2 wells used by local industries, 1 used for drinking water (maximum 370 µg/L). DOE reported 30% higher U concentration than ODH in split samples. One cistern had U concentrations (43 µg/L) greater than background & drinking water standards but had not been used for 2 years. Public water less than background. Radon in water not at level of health concern.
1986 to 1988	RCRA Program monitoring (DOE 1972–1999)	Quarter & semi-annual analyses for metals, organics, radionuclides, total phenol, chloride, fluoride, nitrates, sulfates, and more	Fernald sampled water from off-site private wells 8, 12, 15, 17, and 26 only (onsite RCRA monitoring was continued after 1988)
1996 to present	Integrated Environmental Monitoring Program - Combined all FEMP groundwater monitoring programs into a single program (DOE 1972–1999, 2000, 2001b, 2002, 2003c)	Quarterly and monthly analyses for total uranium and 49 constituents (South Plume wells) Quarterly and monthly analyses for total uranium and 27 constituents (boundary and monitoring wells)	DOE sampled water from ~50 monitoring wells in South Plume DOE sampled water from 33 boundary wells east and south of FEMP plus additional on-site and off-site monitoring wells

Table 22. Maximum yearly concentrations of uranium (in µg/L or ppb) in private wells 12, 13, 15, and 17 off site of the Fernald facility from 1982 to 1997

Year	Well 12	Well 13	Well 15	Well 17	Comments
1960s to 1981	918 to 4,144*	NA	918 to 4,144*	918 to 4,144*	Concentration range represents the highest *estimated* yearly median concentration (lower bound) to the highest *estimated* yearly 95th percentile (upper bound) concentration for residential wells based on known releases to and measurements in the storm sewer outfall ditch and Paddys Run, and an empirical transport model.
1982	310	NA	554	99	Fernald facility began sampling private wells.
1983	306	NA	578	68	
1984	270	NA	365	68	
1985	243	NA	360	55	
1986	332	1	378	61	
1987	410	1	330	170	Wells 12, 15, and 17 used for monitoring purposes only.
1988	300	2	310	73	Wells 12, 15, and 17 used for monitoring purposes only.
1989	350	1	320	54	Wells 12, 15, and 17 used for monitoring purposes only.
1990	210	2	330	56	Wells 12, 15, and 17 used for monitoring purposes only.
1991	190	14	310	54	Wells 12, 13, 15, and 17 used for monitoring purposes.
1992	307	30	260	50	Wells 12, 13, 15, and 17 used for monitoring purposes.
1993	176	78	264	NS	Wells 12, 13, and 15 used for monitoring purposes only; well 17 not sampled after September 1992
1994	162	99	219	NS	Wells 12, 13, and 15 used for monitoring purposes only.
1995	177	93	177	NS	Wells 12, 13, and 15 used for monitoring purposes only.
1996	41	120*	NS	NS	Well 12 concentrations were reported in SED; well 13 maximum uranium concentration was reported in 1996 but was estimated from trend analysis (DOE 1998).
1997	145*	60*	NS	NS	Concentrations in wells 12 and 13 were estimated from trend analysis (DOE 1998).

Key: * - estimated concentrations
NA = data not available
NS = not sampled
SED = Site-Wide Environmental Database
µg/L = micrograms of uranium per liter of water
ppb = parts per bllion
Sources: DOE 1972–1999; Voillesque et al. 1995; DOE 1998

As part of routine environmental monitoring at the site, *private* well water samples were analyzed for metals from 1986 through 1995 (DOE 1972–1999). Iron and manganese were frequently detected above EPA's Secondary Drinking Water Standards (which are based on aesthetics and not on adverse health effects); however, most of the iron and manganese concentrations are typical for groundwater in this area (DOE 1972–1999). In 1988, 1989, and 1990, the maximum concentration of manganese in private wells was above ATSDR's media-specific comparison value (CV) for chronic ingestion by children but not for adults. However, the concentration that exceed the CV was from a different well in each of these years (two wells barely exceeded the CV, and their mean concentrations were below the CV). The highest concentration (1800 µg/L) was reported in 1990 for a well that had a median concentration for all sampling years of less than 15 µg/L. This well's mean concentration for all sampling years (including this outlier) is less than ATSDR's CV. Therefore, *manganese was not selected as a contaminant of concern for groundwater based on private well sampling.* ATSDR does not have a CV for iron in water. Iron is an essential element, but gastric distress (i.e., stomachaches and diarrhea) has been observed at ingestion rates of 70 mg/day which is roughly equivalent to an adult drinking 2 L of water a day with 35 mg iron/L of water, assuming 100% bio-availability (Goldhaber 2003). Since oral bio-availability of iron is closer to 18% and the maximum iron concentration detected in a private well was 31 mg/L, *iron was not selected as a contaminant of concern for groundwater based on private well sampling*. (Most of the iron concentrations were one-half to one-tenth the value that would cause these temporary adverse health effects.) Arsenic was consistently detected in one well above ATSDR's CV, EPA's Maximum Contaminant Level (MCL) for drinking water, and background concentrations; however, this well was not near the site. Arsenic was also detected above the MCL in a few other wells at lower concentrations. (Refer to Table 23.) The background concentration range for arsenic reported in FEMP's background water quality study is 1 to 29 µg/L. Since arsenic concentrations exceeded ATSDR's CV and EPA's MCL, *arsenic was selected as a contaminant of concern for groundwater and will be discussed further*. Other metals (e.g., *cadmium, lead, and zinc*) were detected at levels above ATSDR's CVs or EPA's Drinking Water Standards in very few private well water samples collected from 1986 to 1995. Because these contaminants were detected so infrequently, ATSDR could not identify any patterns of contamination in the wells, nor could ATSDR identify any individual wells that were consistently contaminated. Therefore, *ATSDR did not select these metals as groundwater contaminants of concern based on private well sampling*.

Since 1996, DOE has been analyzing groundwater samples collected from *monitoring* wells on site and in the South Plume area for approximately 50 potential contaminants and from boundary monitoring wells near the east and south site boundaries for 27 potential contaminants. DOE compares the concentrations to Final Remediation Level (FRL)[6] and reports the exceedances in the annual environmental reports. However, the FRLs are not always the same as ATSDR's health-based CVs. Because there is limited non-uranium off-site groundwater monitoring data,

[6] The Final Remediation Levels (FRLs) are specified in the Record of Decision for Remedial Actions at Operable Unit 5 (DOE 1996).

Table 23. Private Well Sampling (non-radioactive) from 1986 through 1995
(maximum concentrations in µg/L/ number of wells exceeding ATSDR's comparison values(CVs))

Year/ number of wells sampled	Arsenic	Iron	Manganese
	CV: CREG=0.02, MCL=10	(EPA's Secondary Drinking Water Standard = 300)	CV: Chronic RMEGc=500
1986/ 24 wells	43/ 3>MCL	4430/ 13>SDWS	None
1987 /25 wells	48/ 4>MCL	2950/ 10>SDWS	None
1988/ 26 wells	94/ 2>MCL	9090/ 15>SDWS	555/ 1>CV
1989/ 25 wells	71/ 2>MCL	3600/ 14>SDWS	560/ 1>CV
1990/ 32 wells	35/ 2>MCL	17000/ 16>SDWS	1800/ 1>CV
1991/ 34 wells	20/ 2>MCL	16000/ 15>SDWS	None
1992/ 37 wells	35/ 4>MCL	12000/ 18>SDWS	None
1993/ 36 wells	49/ 4>MCL	4000/ 16>SDWS	None
1994/ 31 wells	63/ 1>MCL but detection limit (20)>MCL	3577/ 12>SDWS	None
1995/ 32 wells	13100/ 3>MCL	31,370/ 13>SDWS	None

Key:
CREG = cancer risk evaluation guide
EPA = US Environmental Protection Agency
MCL = EPA's Maximum Contaminant Level for drinking water
RMEGc = reference dose (or concentration) media evaluation guide for a child
SDWS = EPA's Secondary Drinking Water Standards
NOTE: analytes not included for following reasons:
(1) no CV for essential elements calcium, potassium, magnesium, and sodium
(2) barium, chromium, copper, nickel, selenium, and silver did not exceed CVs
(3) cadmium, lead, and zinc infrequently exceeded CVs ; not consistent in any one well or area
Sources: DOE 1972–1999

ATSDR reviewed information concerning potential contaminants identified in on-site, South Plume, and boundary monitoring well sampling data from 1997 through 2002. The constituent was not considered a potential off-site contaminant of concern if its concentration was elevated infrequently. ATSDR identified *arsenic* in the South Plume near Paddy's Run Road; *boron* in the South Field (on site); *fluoride* northwest of the site, in the waste pit area, and in the South Field (on site); *manganese* in the South Field, in the waste pit area, in the South Plume, and near the eastern boundary; *molybdenum*, *nitrate/nitrite*, *technetium 99* and *trichloroethene* in the waste

pit area. Because the groundwater flows to the east under the waste pit and production areas, ATSDR scientists ruled out any of the contaminants from the waste pits and production area if it was not detected above the CV on the eastern boundary. *This eliminated molybdenum, nitrate/nitrite, technetium 99, and trichloroethene*. Elevated arsenic concentrations (maximum of 117 µg/L) consistently showed up predominantly in the South Plume area near Paddy's Run Road south of Willey Road. Because it is not predominant in samples from on-site groundwater, *ATSDR scientists do not believe that it is related to the site but will discuss possible exposure to arsenic further in this report.* The elevated boron concentrations appear in samples from two wells in the South Field, but the boron does not appear to be migrating off site; therefore *boron will not be discussed further*. Fluoride concentrations were elevated northwest of the site in 1988 and 1989 sampling. Although these wells were not sampled after March 1989, the concentrations were elevated but were decreasing. The source of this fluoride is unknown but does not appear to be from the site. The most elevated concentration was reported in August 1988 for a well approximately 1 mile south of the site boundary. This well was sampled from August 1988 through June 1991 with no other samples exceeding ATSDR's CV. Fluoride was slightly elevated in the waste pit area; however, sampling on the eastern site boundary did not show consistent elevated concentrations. Therefore, *fluoride* is not considered a current contaminant of concern and will not be discussed further. Manganese concentrations appear to be elevated in many of the wells on-site and off-site. Although this could be a natural phenomena, manganese is a minor impurity (<1%) of uranium ores and ore concentrates, and potential on-site sources could have included waste in the pits, South Field area, solid waste landfills, Plants 1, 2/3, and 8 areas, laboratory area, and flyash piles. Three wells close to each other in the South Plume had elevated concentrations of manganese. The highest concentration in one of the wells appears to be an outlier, but the next highest concentration (5,410 µg/L) is similar to the maximums from the other two wells (4,400 µg/L and 1,729 µg/L) and will be used for screening purposes. Three wells near the eastern boundary had very similar elevated concentrations of manganese. The maximum concentration from these wells (2,250 µg/L) was used for screening purposes. *Exposure to manganese in groundwater will be discussed further.*

Off-site monitoring well data were also reviewed for non-uranium radioactive contaminants from 1991 through 1995. No other radioactive contaminant was detected at a concentration that would have significantly contributed to off-site exposure from ingestion of groundwater.

Residences near the Fernald site also used cisterns as a source of drinking water (ODH 1988; Pinney 2000). Investigations conducted by the Ohio Department of Health from 1985 to 1988 identified and sampled water from 54 cisterns near the site used as drinking water sources (ODH 1988). Only one cistern located immediately north of the site contained water contaminated with uranium at concentrations above background (0.4 to 3 µg/L). The concentration in this cistern (43.3 µg/L) was lower than uranium concentrations detected in the off-site private wells in the South Plume but higher than EPA's drinking water standard. Rainwater collected via roof gutters provided the water for this system. The cistern had been disconnected 2 years before this sampling, had not been used for drinking water since, and had not been disturbed for 2 years

prior to sampling (ODH 1988). ATSDR does not know if the uranium concentration was higher or lower when the water was being used.

DOE contractors are currently remediating the Great Miami aquifer as part of site remedial activities (DOE 1972–1999). The remedy consists of a series of groundwater extraction and re-injection wells and a centralized water treatment facility. The extraction system is designed to prevent further southward movement of the South Plume. Six groundwater extraction wells make up the South Plume module. Four of these wells have been operating since 1993. Groundwater from the plume is being monitored for total uranium, radionuclides, metals, elements, and volatile organic compounds. No other groundwater contaminants of concern have been identified. Groundwater *south* of the South Plume is being monitored for chemical contaminants, including arsenic, phosphorus, potassium, sodium, and volatile organic compounds, originating from the Paddy's Run Road site south of the Fernald site.

Estimated Exposure Doses

ATSDR scientists evaluated *past*, *current*, and *potential future* exposure to *chemical* contaminants in off-site groundwater near the site. Uranium was the main chemical evaluated for this pathway. Other chemicals (metals) were detected infrequently in private wells sampled from 1986 through 1995. No non-uranium chemical data are available prior to 1986. The data used to evaluate chemical contaminants (other than uranium) are in Table 23. ATSDR also reviewed DOE's current chemical contaminant data for wells south and east of the site and determined that exposure to *arsenic* and *manganese* needed further evaluation.

ATSDR evaluated *past*, *current*, and *potential future* exposure to *radioactive* contaminants in the groundwater pathway. *Past* exposures to radioactive contaminants were addressed in the FDRP and the FRAP (Voilleque et al. 1995; Shleien et al. 1995; Killough 1998a, 1998b; CDC 1998, 1999). Although no private well sampling was conducted before 1981, CDC contractors estimated past uranium concentrations in three wells south of the site for 1960 to 1981. Private wells were potentially first impacted by the South Plume sometime after 1962. Although CDC's estimates appear to be very conservative, ATSDR scientists used a range of concentrations, including CDC's estimates, as shown in Table 22 to calculate doses prior to 1981.

ATSDR assumed ingestion as the primary route of exposure to uranium in well water. Uranium dissolved in water is not likely to volatilize; therefore, inhalation was not considered a significant route of exposure. Skin contact with uranium in water may have occurred but is not likely to contribute significantly to uranium exposure dose (ATSDR 1999b; CDC 1998).

ATSDR evaluated two *hypothetical* exposure scenarios for this pathway. The first scenario assumes chronic exposure to a young child, 1 to 6 years of age, who weighs 13 kilograms (kg)

and ingests 1 liter (L) of contaminated water a day. The second assumes chronic exposure to an adult who weighs 70 kg and ingests 2 L of contaminated water a day (ATSDR 1992; EPA 1999).

Chemicals

ATSDR scientists calculated *past* and *current* chemical exposure doses from uranium and *current* chemical doses from arsenic and manganese in groundwater for the two hypothetical exposure scenarios described above. ATSDR compared the estimated exposure doses to health-based guidelines to determine if further evaluation of a public health hazard was warranted.

Past Exposure

Although ATSDR scientists do not know if wells 12, 13, 15, and 17 were used as drinking water sources or how frequently they were used, the scientists evaluated potential *past* exposure based on uranium concentrations in these wells, because they had the highest concentrations and represent possible points of human exposure. The scientists evaluated past exposure by using a range of maximum uranium concentrations that were *measured* and *estimated* in the well water. This range is from 578 micrograms of uranium per liter of water (578 µg/L) to 4,144 µg/L. The lower bound of the range is the maximum concentration *measured* in an off-site private well, which was in well 15 in 1983 (Table 22). The upper bound is the maximum *estimated* upper bound (95th percentile) concentration in wells 12, 15, and 17 in the 1960s (Voilleque et al. 1995). Maximum uranium concentrations and estimated *past chemical* exposure doses for a child (scenario #1) and an adult (scenario #2) are presented in Table 24 (below).

Table 24. Maximum concentrations and estimated chemical exposure doses for current and past exposure to uranium in water from private well 15 near the Fernald site

	Maximum Exposure Concentration or Range (µg/L or ppb)	Exposure Doses* (mg/kg/day)
Current Exposure		
Scenario #1: child	<100 to 283	<0.008 to 0.022
Scenario #2: adult	<100 to 283	<0.003 to 0.008
Past Exposure		
Scenario #1: child	578 to 4,144	0.04 to 0.32
Scenario #2: adult	578 to 4,144	0.02 to 0.12
Key: µg/L = micrograms uranium per liter water ppb = parts per billion mg/kg/day = milligrams of uranium per kilogram of body weight per day * Equations used to estimate doses for this pathway are described in Appendix B. In estimating current doses, ATSDR assumed that the lower bound of the exposure concentration range (<100 ppb) was equal to 100 ppb.		

As Table 24 shows, the estimated exposure doses for *past* exposure are 0.04 to 0.3 milligrams of uranium per kilogram of body weight per day (mg/kg/day) for a child and 0.02 to 0.1 mg/kg/day for an adult. The exposure estimates exceed the health-based guideline (0.002 mg/kg/day) for ingestion of uranium.

Further evaluation of past exposure to chemical uranium in groundwater and the additional contribution of other pathways (e.g., air, surface water, biota) to total uranium exposure of nearby residents are discussed in the Public Health Implications section of this report.

Current Exposure

For current ingestion of *chemical* uranium in private well water, ATSDR used a range of potential exposure concentrations in off-site private wells that were routinely sampled by the Fernald facility. Well 15 had the highest overall uranium concentrations of the wells that were routinely sampled. ATSDR used a *range* of concentrations in well 15 because uranium levels in the South Plume, and in this well, have been decreasing since the facility stopped operating. The upper end of this range is 283 µg/L which is the highest annual *average* uranium concentration detected in well 15 under current conditions. This concentration was reported for 1989, 1990, and 1993 (DOE 1972–1999). Uranium concentrations in well 15 have declined to less than 100 µg/L in 1999 (DOE 1972–1999). Therefore, ATSDR used a concentration of 100 µg/L as a lower-bound concentration to estimate exposure dose under current conditions.

The estimated exposure doses for *current* ingestion exposure to *chemical* uranium in off-site private well water are shown in Table 24 (above). The estimated exposure doses for *current* exposure are 0.008 to 0.022 mg/kg/day for a child and 0.003 to 0.008 mg/kg/day for an adult, as shown in Table 24. Therefore, the estimated exposure doses for a child and an adult exceed the health-based guideline for ingested chemical uranium (0.002 mg/kg/day). Further evaluation of current exposure to chemical uranium in groundwater and the additional contribution of other exposure pathways (i.e., soil, air, surface water, and biota) to total uranium exposure to nearby residents are discussed in the Public Health Implications section of this report.

Residents of the Fernald site area may also be obtaining drinking water from cisterns (ODH 1988). Many of these cisterns were sampled in 1985 to 1988 (ODH 1988). Only one cistern was found to be contaminated with uranium but at concentrations lower than currently present in off-site wells in the South Plume. Contamination of this cistern was presumed to have resulted from air particulate releases of uranium from the Fernald facility (ODH 1988). Because air particulate releases are currently much lower than they were while the facility was operating, it is not likely that cisterns in the Fernald area are contaminated at higher concentrations than reported for 1985 to 1988. This is especially true if cistern water is agitated frequently, as during regular use, or cleaned out periodically.

For *current* ingestion of arsenic and manganese in off-site groundwater, ATSDR used a range of concentrations in off-site wells that were sampled by the facility. In 1995, well 3 had the highest arsenic concentration; however, this value is suspicious since it is a couple orders of magnitude higher than concentrations reported for this well in other years, and the well is upstream of the site. Well 19 (near New Haven Road south of the site) typically had the highest concentrations with the maximum reported as 94 µg/L in 1988. The maximum *current* concentration of arsenic from DOE's South Plume monitoring wells was 117µg/L. This well is near Paddy's Run Road, south of Willey Road. Therefore, for arsenic, ATSDR used a concentration range of 94 to 117 µg/L. Because manganese was not considered a contaminant of concern for private wells that were sampled, ATSDR used the maximum concentrations found in three monitoring wells in the South Plume which ranged from 1,729 µg/L to 5,410 µg/L. Table 25 gives the maximum estimated exposure doses and the health-based guidelines for arsenic and manganese in groundwater. Although the maximum concentrations for both arsenic and manganese were found in wells in the South Plume (where no one is drinking well water now), someone may have been drinking this water before an alternate water supply was provided. Also, arsenic and manganese have been identified in other wells near the site and may not be site-related. However, because the guidelines are exceeded for the hypothetical scenarios, *arsenic and manganese will be discussed in the Health Implications section of this report.*

Table 25. Maximum estimated doses for current exposure to arsenic and manganese in off-site groundwater at the Fernald site

	Scenario	Exposure Doses* (mg/kg/day)	Health-based Guideline* (mg/kg/day)
Arsenic	#1: child	0.007 - 0.009	0.0003 (chronic oral MRL) 0.005 (acute oral MRL)
	#2: adult	0.003	
Manganese	#1: child	0.133 - 0.416	0.07 (ATSDR's interim health guideline)**
	#2: adult	0.049 - 0.155	

Key: mg/kg/day = milligrams contaminant per kilogram body weight per day
MRL = minimal risk level (health-based guideline)
* Equations used to estimate doses are described in Appendix B as well as a description of the Health Guidelines.
** (ATSDR 1997)

Radiation

ATSDR scientists calculated two types of doses for exposure to radionuclides in groundwater: (1) a committed effective dose (whole body) and (2) a committed equivalent dose (bone surface). The bone surface is the major target organ for radiation effects from ingested uranium and most other radioactive contaminants in the groundwater at this site. *Past and current* radiation exposure doses were calculated for the two hypothetical exposure scenarios previously described.

Past Exposure

ATSDR estimated *past* uranium exposure doses based on the estimated concentrations from CDC's FDRP and FRAP (Voilleque et al. 1995; Shleien et al. 1995; Killough 1998a, 1998b; CDC 1998, 1999). Because the maximum exposure scenario for ingestion of contaminated groundwater in the FDRP and FRAP differs from ATSDR's scenarios, *past* exposure doses were calculated using ATSDR's hypothetical scenarios and a range of concentrations, as described for chemical uranium. This range is from 391 pCi/L (14.5 Bq/L) to 2,800 pCi/L (103.7 Bq/L). *Past* committed effective (whole body) and committed equivalent (bone surface) doses from ingestion of radioactive uranium in off-site private well water are in Table 26.

Current Exposure

For *current* exposure to radioactive uranium, ATSDR scientists used a range of concentrations in well 15, as described for chemical uranium. The range is from 67 pCi/L (2.48 Bq/L) in 1999 to 190 pCi/L (7.04 Bq/L) in 1989, 1990, and 1993 (DOE 1972–1999). The *current* estimated committed effective (whole body) and committed equivalent (bone surface) doses from ingestion of radioactive uranium in off-site private well water are in Table 26 (below). As stated previously, no other radioactive contaminant would significantly contribute to off-site exposures from ingestion of groundwater.

Table 26. Uranium concentrations and past and current estimated committed effective and committed equivalent doses from ingestion of off-site groundwater near the Fernald site (lifetime doses based on 1 year ingestion)

Scenario	Range of Maximum Concentrations in pCi/L (Bq/L)	Committed Effective Dose from one year of ingestion in mrem (mSv)*	Committed Equivalent Dose to Bone Surface from one year ingestion in mrem (mSv)*
Past:			
#1: child	391 (14.5) to 2,800 (103.7)	45 (0.45) to 320 (3.2)	662 (6.62) to 4,731 (47.31)
#2: adult	91 (14.5) to 2,800 (103.7)	50 (0.50) to 360 (3.6)	783 (7.83) to 5,602 (56.02)
Current:			
#1: child	<67 (<2.48) to 190 (7.04)	<8.0 (<0.08) to 22 (0.22)	<113 (<1.13) to 321 (3.21)
#2: adult	<67 (<2.48) to 190 (7.04)	<9.0 (<0.09) to 24 (0.24)	<134 (<1.34) to 380 (3.80)

Key: pCi/L = picocurie per liter
Bq/L = becquerel per liter
mrem = millirem
mSv = millisievert
* ATSDR used ICRP Reports 69 and 72 dose conversion factors in these calculations (ICRP 1995a, 1996)

ATSDR estimated these doses by using the International Commission on Radiological Protection's (ICRP's) models and methodology (ICRP 1995a). Further evaluation of the health effects from exposure to radioactive uranium in groundwater is discussed in the Public Health Implications section of this report.

In July 1996, ATSDR conducted a health consultation to evaluate exposure to uranium and other radioactive materials in water used by nearby residents for non-potable purposes (ATSDR 1996a). The consultation used 1994 groundwater sampling data that reflected conditions of exposure at the site close to the time the health consultation was issued. The results of this evaluation indicate that exposure doses from non-potable uses do not exceed health-based guidelines (for chemical uranium) and do not contribute significantly to estimated committed effective doses and equivalent doses (for radioactive uranium and other radioactive materials) for this pathway.

Potential Future Exposure

ATSDR scientists evaluated the likelihood that off-site residential wells would be impacted by contaminants from the Fernald site in the future. According to available environmental data, residents using private wells near the Fernald site in the future are not likely to be exposed to chemicals and radioactive materials from the site. Wells that were impacted by the South Plume have been provided with an alternate water source (public water supply).Wells to the east of the site are not currently being impacted, but ATSDR recommends that DOE continue to monitor groundwater in the future at the eastern site boundary for any site-related contaminants.

If additional information becomes available indicating that off-site concentrations of groundwater contaminants have increased to levels exceeding drinking water standards or other appropriate comparison values, the groundwater exposure pathway should be re-evaluated.

Biota Pathway

Background

There are several farms and gardens near the Fernald site that operated before, during, and after the facility was active. Chemicals and radioactive contaminants released from the site to air, soil, and water may have been deposited onto, or been incorporated into, plants, animals, or animal products (collectively known as *biota*) off site. For example, chemicals and radionuclides released to air may be deposited onto soil and pasture grass, which may be ingested by farm animals or taken up into the root systems of edible plants. These contaminants may then accumulate in the edible portions of the animals (i.e., flesh, milk, eggs) and plants and be consumed by humans. Ingestion of contaminated biota by nearby residents is a potential pathway for human exposure to contaminants from the Fernald site.

Environmental Data

The Fernald site did not routinely collect biota samples when the facility was in operation. Single milk samples were collected from dairy cows on a farm near the site in 1959 and 1960, because the cows grazed on land owned by the site. Milk samples from these cows were also collected and analyzed five times in 1965 and 1966 (Shleien et al.1995), but routine sampling did not begin until 1980 (DOE 1972–1999). Milk samples were also collected from a farm in Indiana and later in Kentucky to determine background conditions. The samples were analyzed for total uranium or uranium isotopes. In 1958, 1959, 1963 through 1968, and 1984 until 1991, grass and other forage near the site were analyzed for fluoride and uranium (Shleien et al.1995). The fluoride concentrations in the grass samples did not exceed the Kentucky standard of 80 mg/g, which is frequently used for screening purposes; however, grass samples were not collected when most fluoride compounds were being used at the site (Plant 4 in 1954, Plant 5 beginning in 1953, Plant 7 from 1954 through 1956, Plant 8 in 1966, and Pilot Plant beginning in 1951). *Therefore, grass sampling could not be used to qualitatively shed light on fluoride air releases in the 1950s*. A comparison was done to show the relationship between uranium soil concentration and uranium in grass. In 1985, the site determined that there was a strong relationship between the distance from the site and uranium concentrations in the soil but only a weak relationship between soil concentrations and uranium concentrations in the grass samples. Basically, the grass sampling results reflected the general trend of atmospheric uranium releases but do not correlate well with the potential uranium uptake in vegetables, beef, poultry or eggs. Produce from three neighboring farms was sampled in 1974. In 1984, the site began routine analyses of uranium in potatoes (with and without their skins) and uranium concentrations in fish from the Great Miami River. In 1988, the facility began collecting a larger variety of produce from farms located within two miles of the site. Tomatoes, corn, soybeans, potatoes, onions, turnips, cabbage, lettuce, green beans, apples, and pumpkins were routinely sampled. Samples were also collected from several farms 10 to 30 miles from the site to represent background conditions (DOE 1972–1999).

Because biota sampling was not routinely conducted while the facility was in operation, contractors for CDC used various mathematical models to predict *past* concentrations of uranium and other radionuclides in biota near the Fernald site in the FDRP (Killough et al.1998a, 1998b). The models incorporate information, such as (1) deposition of uranium and radionuclides onto pasture grass, (2) uptake of radionuclides in soil by plants, (3) consumption of contaminated water and soil by pasture animals, and (4) accumulation of radionuclides in surface water into freshwater fish (Killough et al.1998a, 1998b). The models were used to estimate concentrations of uranium and other radionuclides (thorium 228, 230, 231, 232, and 234; radium 224, 226, and 228; actinium 228; protactinium 234m and 234; and plutonium 239) in vegetables, meat (beef and poultry), fish in the Great Miami River, milk, and eggs. ATSDR scientists reviewed the FDRP documents and agree with the methodology and results for estimating these *past* concentrations which ATSDR used to estimate both chemical and radiological doses from uranium. These concentrations and the estimated doses will be discussed further in this section.

In 1988, the facility sampled fish from the Great Miami River, upstream and downstream of the Fernald outfall. A variety of fish species, including largemouth bass, longear sunfish, river carpsucker, long nose gar, gizzard shad, and channel catfish were sampled (DOE 1972–1999). The results (including those from upstream samples) were approximately two orders of magnitude higher than for years before or after 1988; therefore, ATSDR considered these results suspicious and did not consider them representative of maximum concentrations under past or current conditions at the site.

From 1990 to 1994, milk samples were occasionally analyzed for other radioactive contaminants, such as strontium 90; radium 226 and 228; and thorium 228, 230, and 232. In June and August 1994, ATSDR was assisted by the EPA's National Air and Radiation Environmental Laboratory (NAREL) in sampling milk from the dairy farm bordering the southeast portion of the Fernald property. Samples were also collected from a second dairy farm 6 miles northwest of the site. All samples were analyzed for radioactive contaminants including uranium 234, 235, and 238; potassium 40; and lead 212. In 1995, ATSDR released a health consultation discussing the sampling results (ATSDR 1995a). These data are used in this public health assessment.

In August 1994, ATSDR and EPA's NAREL staff collected produce samples from local produce stands near and downwind of the Fernald site, and from stands upwind and far from the site (ATSDR 1996b). The samples of bell peppers, tomatoes, cabbage, corn, squash, onions, green beans, cucumbers, beets, and cantaloupe were analyzed for radioactive contaminants including uranium 234, 235, and 238; and potassium 40. In 1996, ATSDR released a health consultation discussing the sampling results (ATSDR 1996b). These data are used in this public health assessment.

Table 27 presents the maximum *past* and *current* uranium concentrations in off-site biota near the Fernald facility. Several sources of environmental data mentioned above are represented in

this table (DOE 1972–1999; Killough et al. 1998a, 1998b; ATSDR 1995a, 1996b). The maximum *measured* uranium concentration in milk (20 µg/L) was in a sample collected in August 1965. Generally, the *past measured* uranium concentrations were less than 1 µg/L. The maximum *measured* uranium concentration in grass near the site (680 µg/g) was in a sample collected in April 1963 near the junction of the SSOD and Paddy's Run Creek, and the next highest concentration (607 µg/g) was in a sample collected in April 1967 east of the production area near the solid waste incinerator. These levels decreased significantly by the 1980s, which may indicate that concentrations in vegetation could have been higher in the 1950s. Because biota samples were not collected in the 1950s, ATSDR scientists used CDC's estimated concentrations from the FDRP for the maximum *past* concentrations in Table 27. These values, converted to radiological units, were also used to estimate *past* doses for radioactive uranium.

Table 27. Estimated and measured maximum past and current uranium concentrations in off-site biota near the Fernald site

Maximum Past Uranium Concentrations (in units of µg/kg, or ppb, for vegetables, meat, eggs, and fish; in µg/L, or ppb, for milk)				
Vegetables*	Meat	Milk	Eggs	Fish
189	12	48	3	865
Maximum Current Uranium Concentrations (in units of µg/kg, or ppb, for vegetables, meat, eggs, and fish; in µg/L, or ppb, for milk)				
Vegetables*	**Meat**	**Milk**	**Eggs**	**Fish**
83.5	ND (< 36)	16	NS	122

Key
µg/kg = micrograms per kilogram
µg/L = micrograms per liter
NS = not sampled
ND = not detected (lower limit of analytical detection)
ppb = parts per billion

* Uranium concentrations in vegetables are reported as an average of the maximum concentrations in tomatoes, peppers, potatoes, carrots, beets, onions, radish, soybean, cabbage, eggplant, green beans, pumpkins/squash, lima beans, turnips, and corn.

Source: Killough et al. 1998a, 1998b (for past uranium concentrations); and DOE 1972–1999; ATSDR 1995a, 1996b; DOE 2000b (for current concentrations).

Table 28 presents the average of the maximum *current* radioactive contaminant concentrations in off-site biota. Not all types of biota were sampled each year, and not all types of biota were analyzed for every radionuclide; however, table 28 represents *measured* data from several different sources (DOE 1972–1999; ATSDR 1995a, 1996b).

Table 28. Measured maximum *current* radioactive contaminant concentrations in off-site biota in pCi/g (Bq/kg) for vegetables, meat, and fish and in pCi/L (Bq/L) for milk

Radionuclide	Vegetables*	Meat	Milk†	Fish
Total Uranium	0.05 (0.0000)	- - - - -	11 ± 1.5 (0.407 ± 0.056)	0.082 (0.0000)
Uranium 234	NA	0.01 (0.000)	1.9 ± 0.6 (0.07 ± 0.022)	NA
Uranium 235/236	NA	0.01 (0.000)	<0.44 (<0.016)	NA
Uranium 238	NA	<0.004 (0.000)	1.7 ± 0.55 (0.063 ± 0.02)	NA
Thorium 228	NA	<0.01 (0.000)	2.6 ± 1.2 (0.10 ± 0.04)	NA
Thorium 230	NA	<0.03 (0.000)	0.6 ± 0.5 (0.02 ± 0.02)	NA
Thorium 232	NA	<0.01 (0.000)	<1.2 (<0.044)	NA
Radium 226	NA	<0.03 (<0.000)	0.9 ± 0.3 (0.03 ± 0.01)	NA
Radium, 228	NA	NA	4.9 ± 3.6 (0.18 ± 0.13)	NA
Protactinium 234	NA	NA	900 ± 675 (33.33 ± 25)	NA
Strontium 90	NA	0.00 (0.000)	1.1 ±0.7 (0.04 ± 0.03)	NA
Cesium 137	NA	<0.02 (<0.000)	<10 (<0.417)	NA
Plutonium 238	NA	<0.004 (0.000)	NA	NA
Plutonium 239	NA	<0.003 (0.000)	NA	NA

Key:

pCi/g = picocurie per gram

Bq/kg = becquerel per kilogram

pCi/L = picocurie per liter

Bq/L = becquerel per liter

NA = not analyzed for

* Concentrations in vegetables are reported as the averages of the maximum concentrations in tomatoes, peppers, potatoes, carrots, beets, onions, radishes, soybeans, cabbage, eggplant, green beans, pumpkin/squash, lima beans, turnips, and corn.

† Due to limited data on radionuclides (other than uranium) concentrations in milk samples, ATSDR used average maximums of each for current years in the calculations.

Source: DOE 1972–1999; ATSDR 1995a, 1996b; DOE 2000b

Estimated Exposure Doses

ATSDR scientists evaluated *past*, *current*, and *potential future* exposure to *chemicals* in biota off site of the Fernald facility. Uranium was the only chemical evaluated for this pathway, because ATSDR has no information indicating that other chemicals are present in biota near the facility.

ATSDR scientists also evaluated *past, current*, and *potential future* exposure to *radioactive* contaminants in biota off site of the Fernald facility. From 1989 through 1995, total uranium concentrations were analyzed in produce, milk (discontinued in March 1995), and fish. Other radionuclides evaluated in milk sampled from 1990 through 1994 were strontium 90, radium 226 and 228, thorium 228, 230, and 232, and protactinium 234. Collectively, all radionuclides evaluated for this pathway are referred to as "radioactive contaminants."

ATSDR scientists assumed that ingestion is the only route of exposure to chemicals and radioactive materials in biota. In estimating doses for biota pathways, ATSDR evaluated two *hypothetical* exposure scenarios. The first scenario assumes chronic exposure to a young child, 1 to 6 years of age, weighing 13 kilograms, who consumes from 50% to 100% of a daily diet of vegetables, meat, eggs, and milk from farms and gardens near the site, and consumes 50% of a daily diet of fish from the Great Miami River (EPA 1999; Killough et al. 1998b**).** The second assumes chronic exposure to an adult, weighing 70 kg, who consumes from 50% to 100% of a daily diet of vegetables, meat, eggs, and milk from farms and gardens near the site, and consumes 50% of a daily diet of fish from the Great Miami River (EPA 1999; Killough et al. 1998b). ATSDR's assumptions about ingestion rates and percent of biota consumed by residents from areas near the site for the hypothetical exposure scenarios (described above) are presented in Table 29. These assumptions are very similar to the scenarios used in CDC's FDRP for individuals maximally exposed to contaminants from consumption of vegetables from a garden irrigated with river water and fish from the Great Miami River.

Table 29. Age-specific ingestion rates and percent of biota consumed by Fernald residents for two hypothetical exposure scenarios for the biota pathway

Scenario	Total Ingestion Rates* (in units of kg/day for vegetables, meat, eggs, and fish; and L/day for milk)					Percent of Diet from Area Near Fernald Site*	
	Vegetables	Meat	Milk	Eggs	Fish	Vegetables, Meat, Fish	Milk, Eggs
#1: child	0.2	0.05	0.36	0.01	0.01	50	100
#2: adult	0.3	0.08	0.24	0.03	0.02	50	100

Key:
kg/day = kilograms per day
L/day = liters per day
Notes:
For Scenario #1, the age-specific ingestion rates represent an average of rates for males and females in three age categories (0–1, 1–4, and 5–9 years old).
For Scenario #2, the age-specific ingestion rates represent an average of rates for males and females in six age categories (15–19, 20–24, 25–29, 30–39, 40–59, and $\geq$ 60 years old).
* Total ingestion rates do not account for the fact that only 50% of the total amount of vegetables, meat, and fish are consumed under the hypothetical exposure scenarios.
Source: Killough et al. 1998b; EPA 1999

Chemicals

ATSDR scientists calculated doses for exposure to chemical uranium in biota. The estimated exposure doses for chemical uranium are compared to health-based guidelines for ingested uranium to determine whether further evaluation of a public health hazard is warranted.

Past Exposure

Because routine sampling was not conducted for all types of biota during the early period of facility operations, ATSDR scientists used CDC's estimated concentrations in biota (vegetables, meat, milk, eggs, and fish) near the site to predict levels of past human exposure that may have resulted from consumption of these foods (Killough et al. 1998b). CDC's estimated past uranium concentrations in biota are presented in Table 27. ATSDR's estimated doses for the two hypothetical exposure scenarios are presented in Table 30.

Table 30. Estimated maximum past uranium (chemical) exposure doses for hypothetical exposure scenarios for biota pathway

Scenario	Estimated Exposure Doses* (in mg/kg/day)					
	Vegetables	Meat	Milk	Eggs	Fish	Total Biota
#1: child	0.0015	0	0.0013	0	0.0003	0.0031
#2: adult	0.0004	0	0.0002	0	0.0001	0.0007

Key: mg/kg/day = milligrams of uranium per kilogram of body weight per day
* Equations used to estimate doses for this pathway are described in Appendix B—Exposure Doses and Health-Based Guidelines.

The *past* estimated exposure dose for a child slightly exceeds the health-based guidelines for ingestion of highly soluble uranium. The estimated exposure dose for an adult does not exceed the health-based guideline. However, ATSDR scientists evaluated the public health hazard from this pathway for both children and adults together with other pathways that contribute to total uranium exposure to nearby residents in the Public Health Implications section of this report.

Current Exposure

ATSDR scientists used maximum current uranium concentrations in biota sampled near the Fernald facility to estimate current chemical exposure doses. These uranium concentrations are presented in Table 27. ATSDR's estimated doses are presented in Table 31.

These doses do not exceed the health-based guideline for ingested chemical uranium. However, the potential health hazard for this pathway was evaluated with other pathways in the Public Health Implications section of this report.

Table 31. Estimated maximum current uranium (chemical) exposure doses for hypothetical exposure scenarios for biota pathway

Scenario	Estimated Exposure Doses* (in mg/kg/day)					
	Vegetables	Meat	Milk	Eggs	Fish	Total Biota Dose
#1: child	0.0006	NA	0.0004	NA	0.0000	0.001
#2: adult	0.0002	NA	0.0001	NA	0.0000	0.0003

Key: mg/kg/day = milligrams of uranium per kilogram of body weight per day
NA = not analyzed
* Equations used to estimate doses are discussed in Appendix B—Exposure Doses and Health-Based Guidelines.

Radiation

The whole body and bone surface are the major target organs for ingested uranium and other radionuclides in biota. ATSDR calculated committed effective doses (whole body) and committed equivalent doses (bone surface) for both scenarios described previously.

Past Exposure

Past exposures to radioactive contaminants in biota were calculated based on CDC's maximum estimated concentrations for *uranium* in off-site biota from the FDRP (Voilleque et al.1995; Shleien et al.1995; Killough et al. 1998a, 1998b) and ingestion rates in Table 29. These estimated doses are presented in Table 32 and are based on 1-year chronic ingestion. When compared to multiple years of exposure as used in CDC's scenarios, the results are very similar.

Table 32. Estimated maximum past (radioactive) uranium committed effective (whole body) and committed equivalent (bone surface) doses based on 1-year exposure for hypothetical exposure scenarios for the biota pathway

Estimated Committed Effective Doses* in millirem (in millisieverts)						
Scenario	Vegetables	Meat	Milk	Eggs	Fish	Total
#1: child	1.4 (0.014)	0.0 (0.000)	1.3 (0.013)	0.0 (0.000)	0.3 (0.003)	3.0 (0.030)
#2: adult	1.2 (0.012)	0.0 (0.000)	0.5 (0.005)	0.0 (0.000)	0.4 (0.004)	2.1 (0.021)
Estimated Committed Equivalent Doses* in millirem (in millisieverts)						
#1: child	20.7 (0.207)	0.3 (0.003)	18.9 (0.189)	0.0 (0.000)	4.7 (0.047)	44.6 (0.446)
#2: adult	19.4 (0.194)	0.3 (0.003)	7.9 (0.079)	0.1 (0.001)	5.9 (0.059)	33.6 (0.336)

* ATSDR used ICRP Reports 69 and 72 dose conversion factors in these calculations (ICRP 1995a, 1996)and ingestion rates in Table 29

Current Exposure

Current exposures to radioactive contaminants in biota are based on maximum *measured* concentrations presented in Table 28. The estimated *current* committed effective doses (whole body) and committed equivalent doses (bone surface) from ingestion of radioactive contaminants in biota are presented in Table 33. Current doses are higher than past doses because past doses are based on estimated concentrations of only uranium, whereas current doses are based on measured concentrations of all radioactive contaminants in biota. The elevated bone surface dose for a child from milk is attributed mainly to the concentration of radium 228. Further evaluation of these doses is discussed in the Public Health Implications section of the report.

Table 33. Estimated maximum *current* doses based on 1-year exposure to radioactive contaminants for hypothetical exposure scenarios for the biota pathway

Scenario	Biota Type	Estimated Committed Effective Dose (whole body) for Intake in Year With Maximum Concentrations	Estimated Committed Equivalent Dose (Bone Surface) for Intake in Year With Maximum Concentrations
#1: child	Vegetables	0.6 mrem (0.006 mSv)	8.1mrem (0.08 mSv)
	Meat	< 1.3 mrem (< 0.013 mSv)	< 47 mrem (< 0.47 mSv)
	Milk	19.3 ± 14.3 mrem (0.193 ± 0.143 mSv)	587 ± 439 mrem (5.87 ± 4.39 mSv)
	Fish	< 0.1 mrem (< 0.001 mSv)	< 1 mrem (< 0.01 mSv)
	Total	21.1 ± 14.3 mrem (0.211 ± 0.143 mSv)	643 ± 439 mrem (6.43 ± 4.39 mSv)
#2: adult	Vegetables	0.5 mrem (0.005 mSv)	7.5 mrem (0.075 mSv)
	Meat	< 2 mrem (< 0.02 mSv)	< 75 mrem (< 0.75 mSv)
	Milk	3.1 ± 2.1 mrem (0.031 ± 0.021 mSv)	93 ± 63 mrem (0.93 ± 0.63 mSv)
	Fish	< 0.1 mrem (< 0.001 mSv)	< 1 mrem (< 0.01 mSv)
	Total	5.7 ± 2.1 mrem (0.057 ± 0.021 mSv)	176.5 ± 63 mrem (1.77 ± 0.63 mSv)

Key
mrem = millirems
mSv = millisieverts (1 mSv = 100 mrem)

Potential Future Exposure

Remedial activities at the Fernald site are expected to continue through 2006. During that time, contaminated soils and debris will be removed from the area and transported to on-site and off-site disposal areas. As the remedial projects are completed, releases to air, soil, and surface water should be reduced. In turn, this should limit the uptake and accumulation of contaminants in plants, animals, and animal products and the potential for human exposure to chemicals and radioactive contaminants in off-site biota. *According to available sampling, there is no indication that future activities will result in off-site human exposure to contaminated biota.* However, if additional information becomes available indicating that contaminants have been released or migrated to biota off site, biota pathways should be re-evaluated.

Public Health Implications

Introduction

The exposure pathway analyses for air, soil, surface water, groundwater, and biota indicate that chemicals and radioactive materials have been released from the Fernald site to the environment. Some contaminants have migrated to off-site areas, where human exposures have or may have occurred in the past or may be occurring currently. The exposure pathway is considered *complete* if all five elements discussed in the Environmental Contamination, Exposure Pathways, and Potentially Exposed Populations section link the contaminant source to a receptor population. A *potential* pathway exists when one or more of the elements are missing but exposure could have occurred in the past, could be occurring now, or could occur in the future.

As the Environmental Contamination, Exposure Pathways, and Potentially Exposed Populations section describes, ATSDR scientists used available information to estimate maximum human exposure doses at off-site locations where human exposure is likely to occur in order to screen potential contaminants. These doses were estimated for hypothetical scenarios for each completed and potential exposure pathway (i.e., groundwater, soil, air, surface water, and biota); however, very conservative assumptions were use. For instance, in the air pathway section for exposure to radioactive particles, ATSDR scientists used ICRP dose conversion factors that are based on the activity median aerodynamic diameter (AMAD) of one micron, but DOE's sampling during remedial activities in 1998 indicated that more than 70% of the particles were greater than 15 microns and in late 2000 and early 2001 a study showed the majority of the particles were greater than 9 microns (non-respirable). Also, during historical plant operations, approximately 60% of the particles released from the *scrubbers* were 25 microns or greater, but the remaining 40% were approximately 0.5 microns. The majority of the particles released through the dust collectors during past plant operations were between 2.1 and 14 microns (Killough et al.1998). The ICRP model assumes that one micron particles are deposited in three regions of the respiratory system: (1) the most exterior region, the nasal passage (N-P); (2) the middle region, the trachea and bronchial tree (T-B); and (3) the deepest region, the pulmonary parenchyma (P). As particle size increases, more particles are deposited in the N-P region and fewer in the deep lung (P region). The estimated percent depositions in the regions of the respiratory system for five-micron particles are 73% in the N-P region, 8% in the T-B region and 8% in the P region. The model also considers the absorption of the particles into the bloodstream which depends on the particle size and the solubility of the compound. In general, particles less than two microns are rapidly absorbed into the blood; however, five-micron particles would not be easily absorbed. Larger particles can be transported up the throat by mucociliary action and swallowed. For estimating doses for uranium, ATSDR scientists assumed that 5% of the swallowed particles are absorbed into the bloodstream through the gastrointestinal tract (ICRP 1979, 1994, 1995; ATSDR 1999). In this section, ATSDR typically re-examines the exposure pathways and estimated doses under less conservative, more realistic conditions than those used for screening in the Environmental Contamination, Exposure Pathways, and Potentially Exposed Populations section.

Tables 34 and 35 summarize the estimated maximum exposure doses from the Environmental Contamination, Exposure Pathways, and Potentially Exposed Populations section. One person could not receive maximum doses from all pathways, given that various locations during different time periods are involved. For instance, the maximum exposure from uranium in air would have occurred east of the site in 1955, but the maximum radon exposure would have occurred west of the silos in 1959/1960. Maximum groundwater doses would have been received by someone chronically drinking water from a South Plume well in the 1960s and 1970s. Doses from contaminated soil, surface water, or biota could have been received by anyone in the community with maximum doses received when the plant was operating.

Table 34. Estimated *past* uranium and radiation doses for all exposure pathways

Contaminant and Exposure Scenario	Estimated Maximum *Chemical* Exposure Dose (mg/kg/day)	Estimated Maximum *Radiation* Exposure Dose in mrem (mSv)/yr*	Number of Potentially Exposed Persons†
URANIUM—*completed* pathway for air, groundwater, and biota; *potential* pathway for soil and surface water			
Air Pathway*—maximum exposure immediately east of site prior to 1960s (maximum in 1955)*			
Scenario #1: child	0.0015		Unknown—110 maximum
Scenario #2: adult farmer	0.0025	~852 (8.52) for 1955	Unknown—812 maximum
Groundwater Pathway*—maximum exposure south of site in 1960s and 1970s in South Plume*			
Scenario #1: child	0.04–0.32	45 (0.45)–320 (3.2)	Unknown—25 maximum
Scenario #2: adult	0.02–0.12	50 (0.5)–360 (3.6)	Unknown—200 maximum
Biota Pathway*—maximum exposure to anyone eating locally grown produce, meat, fish, and dairy products*			
Scenario #1: child	0.0031	3 (0.03)	Unknown—110 maximum
Scenario #2: adult	0.0007	2.1 (0.02)	Unknown—812 maximum
Soil Pathway*—potential maximum exposure to child playing in the soil immediately east of the site*			
Scenario: young child	0.0002	0.2 (0.002)	Unknown—30 maximum
Surface water*—potential maximum exposure to young child playing in PRC and older child playing in GMR*			
Scenario #1: young child	0.002	3.66 (0.037)	Unknown—110 maximum
Scenario #2: older child	0.00004	0.3 (0.003)	Unknown—110 maximum

Key:
mg/kg/day = milligrams of uranium per kilogram of body weight per day
mrem/yr = millirem per year; mSv = millisieverts (1mSv = 100 mrem)
PRC = Paddy's Run Creek; GMR = Great Miami River
* The estimated maximum radiation exposure dose is the committed effective dose from one year of exposure.
† The number of potentially exposed persons is estimated using 1990 Census data for persons residing within 1 mile of the site. The number of older children (>6 years of age) and adults in this population is unknown; an upper-bound estimate was made by subtracting the number of children under 6 from the total population.

Table 35. Estimated *current* chemical and radiation doses for all exposure pathways

Contaminant and Exposure Scenario	Estimated Maximum *Chemical* Exposure Dose (mg/kg/day)	Estimated Maximum *Radiation* Exposure Dose in mrem (mSv)/yr*	Number of Potentially Exposed Persons†
URANIUM and all RADIOACTIVE CONTAMINANTS—*completed* pathway for air, groundwater, and biota; *potential* pathway for soil and surface water			
Air Pathway*—maximum exposure to person east/northeast of site (not including radon exposure)*			
Scenario #1: child	0.0000002	0.58 (0.0058)–all rad 0.025 (0.0003)–uranium	Unknown—110 maximum
Scenario #2: adult farmer	0.0000004	8 (0.08)–all rad 2.1 (0.021)–uranium	Unknown—812 maximum
Groundwater Pathway—*exposure to persons drinking water from South Plume wells*			
Scenario #1: child	<0.008–0.022	<8 (<0.08)–22 (0.22)	Unknown—0 maximum
Scenario #2: adult	<0.003–0.008	<9 (<0.09)–24 (0.24)	Unknown—0 maximum
Biota Pathway*—exposure to anyone eating locally-grown produce, meat, fish, and dairy products*			
Scenario #1: child	0.001	21 (0.21)–all rad	Unknown—110 maximum
Scenario #2: adult	0.0003	5.7 (0.057)–all rad	Unknown—812 maximum
Soil Pathway*—potential exposure to child playing in soil east of the site*			
Scenario: child	0.0002	Pre-1992: 0.74 (0.0074) Post-1992: 0.38 (0.0038)	Unknown—30 maximum
Surface Water Pathway*—potential exposure to young child playing in PRC and an older child playing in GMR*			
Scenario #1: young child	0.0002	2.62 (0.026)	Unknown—110 maximum
Scenario #2: older child	0.00001	1.9 (0.019)	Unknown—110 maximum
ARSENIC—potential pathway northwest and south of site (**groundwater**); northwest and northeast of site (**soil**)			
Scenario #1: child	0.007–0.009	NA	Unknown—30 maximum
Scenario #2: adult	0.003	NA	Unknown—30 maximum
MANGANESE—potential pathway south and east of site (**groundwater**)			
Scenario #1: child	0.133–0.416	NA	Unknown—0 maximum
Scenario #2: adult	0.049–0.155	NA	Unknown—0 maximum

Contaminant and Exposure Scenario	Estimated Maximum *Chemical* Exposure Dose (mg/kg/day)	Estimated Maximum *Radiation* Exposure Dose in mrem (mSv)/yr*	Number of Potentially Exposed Persons†
Key: mg/kg/day = milligrams of chemical contaminant per kilogram of body weight per day mrem/yr = millirem per year mSv = millisievert (1 mSv = 100 mrem) * The estimated maximum radiation exposure dose is actually the committed effective dose (whole body for a lifetime) from one year of exposure. † The number of potentially exposed persons is estimated using 1990 Census data for persons residing within one mile of the site boundary. The number of older children (>6 years of age) and adults in this population is not known; an upper-bound estimate was made by subtracting the number of children under 6 from the total population.			

Not all environmental exposures result in adverse health effects. As part of the public health implications discussion, ATSDR scientists consider several factors that influence whether an exposure to a chemical or radionuclide can cause harm. These factors are exposure levels, duration and frequency of exposure, route of exposure, toxicity or radioactivity of the substance, and how the body handles the substance following exposure. ATSDR scientists also consider a person's overall health and nutritional status, lifestyle (e.g., smoking and alcohol consumption, diet, level of physical activity), and genetic makeup, because these factors can affect whether exposure to a chemical or radionuclide results in adverse health effects.

Uranium is the main *chemical* contaminant evaluated in this section, because the exposure pathways analyses indicate that estimated exposure doses for most chemicals present in off-site environmental media are either indeterminate or do not exceed their respective health-based guidelines. There are several *past* estimated exposure doses for non-uranium chemicals that were not able to be determined. They include doses for past air releases from non-production activities such as coal burning, waste burning, fire training, etc. described in the air pathway section and from non-uranium chemical concentrations in soil, surface water, and groundwater before 1986.

There is one *current* estimated exposure dose that ATSDR could not determine by the information reviewed. ATSDR could not determine whether off-site concentrations of specific elements, such as beryllium, were of public health concern. Waste pit remedial activities began in late 1999, and it appears that beryllium is/was located in three waste pits. Although the site used relatively small quantities of beryllium, air monitoring was not sufficient to say whether airborne particles of beryllium or other specific chemicals were at a level of health concern.

In some cases the *current* radiation doses in Table 35 are higher than the *past* radation doses in Table 34, because the past doses are usually based on only uranium concentrations; whereas, current doses are based on several radionuclides that were detected during the sample analyses. ATSDR believes that, especially in these cases, the total radiation dose for past exposures were higher than those based solely on uranium concentrations.

Uranium

Pathways of Exposure

Uranium is both a *chemical* and a *radioactive* contaminant evaluated in all exposure pathways for this site. As stated previously, one person could not receive maximum doses from all pathways given that the maximum concentrations occurred at various locations during different time periods. Table 36 summarizes total *chemical* uranium exposure doses for various directions from the site. (Effects of uranium as a radioactive material will be discussed in the Radioactive Materials and Radiation Exposure section.)

Table 36. Estimated maximum total chemical uranium doses for persons living near the site

PAST EXPOSURES		CURRENT EXPOSURES
East/northeast of site prior to 1960s		*East/northeast of site since 1989*
Child	Air pathway—0.0015 mg/kg/day Soil pathway—0.0002 mg/kg/day Biota pathway—0.0031 mg/kg/day* Total estimated dose—0.0048 mg/kg/day*	Air pathway—0.0000 mg/kg/day Soil pathway—0.0002 mg/kg/day Biota pathway—0.0010 mg/kg/day Total estimated dose—0.0012 mg/kg/day
Adult	Air pathway—0.0025 mg/kg/day* Biota pathway—0.0007 mg/kg/day Total estimated dose—0.0032 mg/kg/day*	Air pathway—0.0000 mg/kg/day Biota pathway—0.0003 mg/kg/day Total estimated dose—0.0003 mg/kg/day
South of site in 1960s and 1970s		*South of site since 1989*
Child	Groundwater pathway—0.04–0.32 mg/kg/day* Biota pathway—0.003 mg/kg/day* Total estimated dose—0.043–0.323 mg/kg/day*	Groundwater pathway—0.008–0.022 mg/kg/day* Biota pathway—0.001 mg/kg/day Total estimated dose—0.009–0.023 mg/kg/day*
Adult	Groundwater pathway—0.02–0.12 mg/kg/day* Biota pathway—0.0007 mg/kg/day Total estimated dose—0.021–0.121 mg/kg/day*	Groundwater pathway—0.003–0.008 mg/kg/day* Biota pathway—0.0003 mg/kg/day Total estimated dose—0.003–0.008 mg/kg/day*
West of site before 1989		*West of site since 1989*
Child	Surface water pathway—0.002 mg/kg/day Biota pathway—0.003 mg/kg/day* Total estimated dose—0.005 mg/kg/day*	Surface pathway—0.0002 mg/kg/day Biota pathway—0.001 mg/kg/day Total estimated dose—0.0012 mg/kg/day
Adult	Biota pathway only—0.0007 mg/kg/day	Biota pathway only—0.0003 mg/kg/day
North/northwest of site before 1989		*North/northwest of site since 1989*
Child	Biota pathway only—0.0031 mg/kg/day*	Biota pathway only—0.001 mg/kg/day
Adult	Biota pathway only—0.0007 mg/kg/day	Biota pathway only—0.0003 mg/kg/day
Key: mg/kg/day = milligrams of uranium per kilogram of body weight per day * values that exceed ATSDR's Health Guideline for highly soluble uranium (0.002 mg/kg/day)		

For chemical uranium, the pathway that would have contributed the most significant dose would be drinking water from a South Plume well; however, no one is drinking this water now. ATSDR has determined that residents who drank water from privately owned wells in the South Plume at the maximum uranium concentrations estimated in the past were likely to have experienced adverse health effects. These effects were likely to have been mild and transient (short-lived). They may have involved biochemical or histological (structural) changes to the kidney. ATSDR provides a more detailed discussion below.

Residents have expressed concerns about possible adverse health effects from past exposure to uranium in water from privately owned wells near the facility. Preliminary analyses of data from residents participating in the Fernald Medical Monitoring Program (FMMP) suggest a higher than expected occurrence of kidney cancer. Because the FMMP participants may not be representative of the Fernald community as a whole, these findings should be interpreted with caution. It is not known whether any of the observed increases in health effects are related to chemicals and radioactive releases from the Fernald site, because there has been no in-depth assessment of past exposures to site-related contaminants in individual wells near the site and because there are contaminant sources other than the Fernald facility near the site.

According to the estimated *total* uranium doses for *all* exposure pathways under *past* conditions at the site (shown in Table 36), the health hazard posed by the groundwater pathway may have been compounded by exposure to uranium via other potential pathways (i.e., biota). However, groundwater contributes the most to total uranium exposure to nearby residents. ATSDR scientists used predicted uranium concentrations for the most highly contaminated private wells in the South Plume from the FDRP, as well as actual uranium measurements for these wells, to estimate exposure doses under past conditions at the site. Very conservative assumptions were used for the estimates from the FDRP.

Next to the contribution by ingestion of groundwater, ingestion of biota by a child and inhalation of airborne particles by an adult contributed more than other potential pathways to the total chemical uranium doses under *past* conditions. ATSDR used very conservative assumptions to estimate these exposure doses. First, ATSDR assumed that a child ingests all their vegetables, meat, milk, eggs, and fish from nearby sources, and that these biota are contaminated at the maximum concentrations of uranium detected. This is unrealistic, because local foodstuffs are not always available and would not contain maximum concentrations all year, every year. Second, ATSDR assumed that an adult farmer breathed the maximum airborne uranium concentration while performing heavy work near the site for 10 hours a day for 351 days a year. However, this farmer would not always be breathing the maximum concentration detected and would probably not work outdoors for 10 hours per day for 351 days a year due to inclement weather and days off for sickness or holidays. If ATSDR had used more realistic assumptions, the estimated doses would have been considerably lower. Therefore, ATSDR scientists believe exposures to uranium via biota consumption by a child or inhalation of airborne uranium particles by an adult farmer contributed minimally to the total chemical uranium dose to nearby residents under *past* conditions.

The groundwater pathway contributes most to the total uranium dose to residents living in the South Plume area under *current* conditions at the site before an alternate water supply was provided. ATSDR estimated exposure doses by using the range of uranium concentrations detected in monitoring wells in the South Plume. Uranium concentrations in the South Plume have been declining since the facility stopped operating. In 1989, 1990, and 1993, the highest annual average uranium concentration detected in a former privately owned well (well 15) was 283 micrograms per liter of water (µg/L). ATSDR used 283 µg/L as an upper-bound exposure concentration to estimate exposure doses under current conditions. In 1999, maximum concentrations in this well were not detected above 100 µg/L. ATSDR used 100 µg/L as a lower-bound exposure concentration to estimate exposure doses under current conditions. The estimated upper-bound exposure doses for ingestion of water from privately owned wells under current conditions exceed the health-based guideline for ingested uranium by a factor of 10, while the estimated lower-bound doses exceed the guideline by a factor of 3. No one is currently drinking water from these wells.

Properties Affecting the Chemical Toxicity of Uranium

Uranium is a radioactive metal that occurs naturally in the environment (natural uranium). Small amounts of natural uranium are present in rocks, soil, surface water and groundwater, air, plants, and animals; these small amounts contribute to natural background radiation. The amount of background radiation that has been measured in drinking water in different parts of the United States is generally less than 1 pCi/L, or approximately 1.5 µg/L of water (ATSDR 1999b).

There are essentially three kinds of uranium: natural, enriched, and depleted uranium. Uranium is primarily a mixture of three isotopes, uranium 234 (U-234), uranium 235 (U-235), and uranium 238 (U-238), in different proportions. All uranium isotopes react the same chemically but have a different number of neutrons. They are radioactive but have different amounts of radioactivity per gram of material (*specific activity*). U-238 has the lowest specific activity, and U-234 has the highest. By weight, more than 99% of natural uranium is U-238, 0.72% is U-235, and 0.005% is U-234. Depleted uranium is primarily U-238 with U-235 and U-234 depleted. Enriched uranium can be enriched up to 97.3% U-235 but is generally enriched up to 3% U-235 (ATSDR 1999b).

Uranium can harm people in two ways, as a chemical toxin affecting the kidneys and as a radioactive substance. Because natural uranium produces very little radioactivity, the chemical effects of uranium are more harmful than the radioactive effects. Exposure to the more radioactive enriched uranium can produce greater injury to the kidney than natural uranium due to the combined effects of chemical and radioactive properties. There is evidence from dogs and rodents studies that exposure to 90% enriched uranium produces greater kidney toxicity than chemical and radioactive effects separately, because the chemical and radiological effects appear to be additive (Filippova et al. 1978). However, the uranium used in these studies was 90% enriched U-235, which has a much higher specific activity than the enriched uranium used at the Fernald site. The Fernald site used up to 1.25% enriched uranium (Voilleque et al. 1995). No studies were found in which 1.25% enriched uranium produced additive effects on the kidney.

The kidney is the target organ for the toxic effects from uranium. This means that renal toxicity is the first adverse effect that occurs as exposure doses increase from low to high levels. The extent of toxicity is determined primarily by the exposure route, type of uranium compound, and solubility of that compound. Ingested uranium compounds are less toxic than inhaled uranium compounds, partly because uranium is poorly absorbed (e.g., less than 5%) from the gastrointestinal tract following ingestion. Highly soluble uranium compounds are more readily absorbed from the gastrointestinal tract into the blood and are more toxic than less-soluble compounds when exposure occurs by ingestion (Tannenbaum and Silverstone 1951). Table 37 categorizes the relative water solubility and kidney toxicity of several uranium compounds.

Table 37. Relative water solubility and kidney toxicity of various uranium compounds

Relative Water Solubility	Relative Toxicity to Kidney	Uranium Compound
Most water soluble	Most toxic	Uranyl nitrate hexahydrate Uranyl fluoride (uranium hexafluoride) Uranium tetrachloride Uranium pentachloride
Low water solubility	Low to moderate toxicity	Uranium trioxide Sodium diuranate Ammonium diuranate
Relatively insoluble	Least toxic	Uranium tetrafluoride Uranium dioxide Uranium peroxide Triuranium octaoxide

The uranium compounds used and produced by the Fernald facility (e.g., uranium dioxide, uranium trioxide, uranium tetrafluoride, and uranium hexafluoride) ranged from insoluble to most water soluble. Acids were used in many of the production processes at the facility; if they were released to the environment along with the uranium, these acids may have enhanced the uranium's water solubility.

Evidence for Uranium Toxicity and Health-Based Guidelines

There are very few epidemiological or occupational studies indicating that ingestion of uranium results in adverse effects on the kidney. There is one case study of an individual who developed kidney damage within 16 hours of ingesting 15 grams of uranyl acetate in a suicide attempt (Pavlakis et al. 1996). The kidney damage persisted for 6 months before the kidneys began to recover. However, normal kidney function did not fully return. Assuming that this individual weighed 70 kilograms, the estimated exposure dose (body dose) to this individual was 210 milligrams per kilograms (mg/kg). In contrast, ATSDR's maximum estimated exposure dose for nearby residents exposed to uranium in water from privately owned wells in the South Plume is 0.3 mg/kg (Table 36)—many times lower than the dose ingested during this suicide attempt.

Studies using laboratory animals provide the majority of evidence for kidney toxicity from ingestion of uranium. Several studies have been conducted in which animals were exposed to uranium in drinking water and in their diet for acute (less than 14 days), intermediate (15 days to less than 365 days), and chronic (more than 365 days) durations. The animals developed kidney toxicity when uranium was present at sufficient doses (Maynard and Hodge 1949; Tannenbaum and Silverstone 1951; Domingo et al. 1987; Ortega 1989; Gilman et al. 1998; ATSDR 1999b).

ATSDR's health-based guidelines for intermediate and chronic exposure to uranium via *ingestion* is based on kidney toxicity in rabbits exposed to uranyl nitrate hexahydrate, a soluble uranium compound, in drinking water for 90 days. A dose-dependent change in kidney structure, considered indicative of kidney toxicity, was noted at a lowest-observed-adverse-effect-level (LOAEL) of 0.06 mg/kg/day (Gilman et al.1998). Dogs and rabbits appear to be the most sensitive species (ATSDR 1999b). The LOAEL was divided by 30 to account for uncertainty in extrapolating data from animal studies to humans and to be protective of individuals who may be more sensitive than the general population to uranium toxicity. The health-based guideline is 0.002 mg/kg/day. *Only ATSDR's estimated doses for past exposure to uranium for persons drinking water from wells south of the site exceed the LOAEL (0.06 mg/kg/day) for uranium.*

ATSDR has established health guidelines for *inhalation* of both soluble and insoluble uranium compounds. The guideline for *insoluble* uranium is 8×10^{-3} mg/m^3, based on structural changes (lesions) in kidneys of dogs exposed to uranium dioxide dust 6 hours a day, 6 days a week, for 5 weeks (Rothstein 1949). The Rothstein study provided no information about the size of uranium particles used. Therefore, the guideline was based on the conservative assumption that uranium particles were 2 microns or less in diameter. Available environmental sampling data and historical process information indicate that most uranium particles released from the Fernald facility were larger than 2 microns in diameter and were composed primarily of insoluble uranium compounds. Because the respiratory tract can absorb smaller particles more readily than larger particles, the guideline serves as a conservative basis for screening the possibility of causing health hazards for individuals near this site. *None of ATSDR's past and current estimated or measured airborne concentrations of uranium (presented in the Environmental Contamination, Exposure Pathways, and Potentially Exposed Populations section) exceed the health-based guideline for inhalation.*

The health guideline for *inhalation* of *soluble* uranium is 3×10^{-4} mg/m^3, based on kidney lesions in dogs exposed to uranium chloride in air 6 hours a day, 6 days a week, for 1 year (Stokinger et al. 1953). Because the Stokinger study provided no information about uranium particle size, ATSDR scientists assumed that particles were 2 microns or less in diameter. ATSDR's estimated maximum airborne uranium concentration, which occurred in 1955, exceeds this health-based guideline. This past concentration (3.7×10^{-3} mg/m^3) was only for 1955. The concentrations for all other past years before 1989 were approximately 6×10^{-4} mg/m^3, which slightly exceed the health guideline. In estimating these concentrations, ATSDR used conservative assumptions, i.e., the diameter of the particles. Also, ATSDR does not consider this health-based guideline for

inhalation of soluble uranium to be as relevant to the Fernald site, because most of the uranium compounds released in the past were either relatively insoluble or had low water solubility.

Other federal agencies have set limits for uranium in the environment and workplace. In 2000, the EPA established a Maximum Contaminant Level (MCL) for uranium in drinking water of 30 μg/L (or 30 parts per billion), which went into effect December 8, 2003 (DOE 2000a). Groundwater concentrations in the South Plume exceeded this MCL in the past and exceed it today, but currently no one is drinking this water.

Mechanisms of Uranium Toxicity

Several mechanisms for uranium-induced kidney toxicity have been proposed. In one of these, uranium accumulates in specialized (epithelial) cells that enclose the renal tubule, where it reacts chemically with ion groups on the inner surface of the tubule. This interferes with ion and chemical transport across the tubular cells, causing cell damage or cell death. Cell division and regeneration occur in response to cell damage and death, resulting in enlargement and decreased kidney function. Heavy metal ions, such as uranyl ions, may also delay or block the cell division process, thereby magnifying the effects of cell damage (Leggett 1989, 1994; ATSDR 1999b).

Animal and human studies conducted in 1940s and 1950s provide evidence that humans can tolerate certain levels of uranium, suffering only minor effects on the kidney (Leggett 1989). Most of these studies involved inhalation exposures to uranium; however, the kidney is also the target organ for inhaled uranium. On the basis of this tolerance, the International Council on Radiologic Protection (ICRP) adopted a maximal permissible concentration of 3 μg of uranium per gram of kidney tissue for occupational exposure in 1959 (Spoor and Hursh 1973). This level has often been interpreted as a threshold for chemical toxicity.

More recent papers have been published on effects of uranium at levels below 3 μg/g, and those papers have discussed possible mechanisms of uranium toxicity (Diamond 1989; Leggett 1989, 1994; Zhao and Zhao 1990; Morris and Meinhold 1995). It is thought that the kidney may develop an acquired tolerance to uranium after repeated doses; however, this tolerance involves detectable histological (structural) and biochemical changes in the kidney that may result in chronic damage. Cells of the inner surface of the tubule that are regenerated in response to uranium damage are flattened, with fewer energy-producing organelles (mitochondria). Transport of ions and chemicals across the tubule is also altered in the tubule cells (Leggett 1989, 1994; McDonald-Taylor et al.1997). These effects may account for the decreased rate of filtration through the kidney and loss of concentrating capacity by the kidney following uranium exposure. Biochemical changes include diminished activity of important enzymes (such as alkaline phosphatase), which can persist for several months after exposure has ended. Therefore, acquired tolerance to uranium may not prevent chronic damage, because the kidney that has developed tolerance is not normal (Leggett 1989). Acting on the basis of this recent information for

uranium, researchers have suggested that exposure limits be reduced to protect against these chronic effects on the kidney.

Renal damage appears to be definite at concentrations above 3 μg/g for a number of different animal species, but mild kidney injury can occur at uranium concentrations as low as 0.1 to 0.4 μg/g in dogs, rabbits, guinea pigs, and rats after they inhale uranium hexafluoride or uranium tetrachloride over several months (Maynard and Hodge 1949; Hodge 1953; Stokinger et al. 1953; Diamond 1989). Zhao and Zhao proposed a limit of uranium to the kidney of 0.26 μg/g based on renal effects in a man who was exposed to high concentrations of uranyl tetrafluoride dust for 5 minutes in a closed room (Zhao and Zhao 1990). The man showed signs of kidney toxicity, including increased protein content in the urine (proteinuria) and non-protein nitrogen. These signs persisted for 4.6 years, gradually returning to normal values. The kidney content 1 day after the accident was estimated to be 2.6 μg/g.

A review of studies of uranium effects on the kidney (Morris and Meinhold 1995) suggests a probability distribution of threshold values for kidney toxicity ranging from 0.1 to 1 μg/g, with a peak at about 0.7 μg/g. The researchers proposed that the severity of effects increases with increasing dose to the kidney with probably no effects below 0.1 to 0.2 μg/g, possible effects on the kidney at 0.5 μg/g, more probable effects at 1 μg/g, and more severe effects at 3 μg/g and above (Morris and Meinhold 1995; Killough et al. 1998b).

ATSDR's health-based guidelines for ingested (and inhaled) uranium are lower than the lower limit threshold for kidney toxicity proposed by Morris and Meinhold (1995). This is probably because ATSDR's guidelines are derived by use of levels of toxicity observed in animal studies, and those guidelines incorporate safety factors to account for uncertainty in extrapolating from animals to humans and to protect the most sensitive human individuals (ATSDR 1992).

Mild effects on the kidney can be detected by sensitive tests of kidney function. Some tests provide insight into the nature of the damage, while others are fairly non-specific for uranium toxicity. Increased urinary excretion of proteins (proteinuria), amino acids (amino aciduria), or glucose (glucosuria) may indicate kidney damage or cell death. Increased urinary excretion of enzymes that are important to kidney function, such as catalase, alkaline phosphatase, N-acetyl-β-glucosaminidase (NAG), and hydrolase, may also indicate kidney damage. Catalasuria, or increased urinary excretion of catalase, may be one of the most important indicators of uranium toxicity to the kidney (Leggett 1989; ATSDR 1999b). Other urinary biochemicals, such as β-2-microglobulin and non-protein nitrogen, are commonly used indicators of kidney damage.

Urinalysis has limitations as a test for kidney toxicity. First, the presence of substances in urine may indicate that kidney damage has occurred, but it cannot be used to determine whether the damage was caused by uranium. Second, most uranium leaves the body within a few days of exposure, so that urine tests can be used only to determine whether exposure has occurred in the

past week or two. Finally, the tests may be used to detect mild effects on the kidney, but such effects are generally transient in nature and may not result in permanent damage.

More severe effects involve greater damage to the kidney that is likely to be clinically manifest and longer lasting. The kidney has incredible reserve capacity and can recover even after showing pronounced clinical symptoms of damage; however, biochemical and functional changes can persist in a kidney that appears to have recovered structurally (Leggett 1989, 1994; CDC 1998).

*ATSDR has determined that residents who chronically drank water from privately owned wells in the South Plume at maximum concentrations estimated for **past** exposure were likely to have experienced adverse health effects on their kidneys. These effects may have involved biochemical or structural changes to the kidney but were likely to have been mild and not permanent.*

*ATSDR determined that there are no exposure pathways for uranium that pose a public health hazard (from chemical effects) under **current** conditions at the site if persons were not drinking well water from the South Plume area during the period after 1989 and before an alternate water supply was provided.*

Radioactive Materials and Radiation Exposures

Pathways of Exposure

For most of the *past* pathways evaluated, uranium was the main contaminant considered. Most of the off-site concentrations were estimated by modeling techniques during the FDRP. For each *current* pathway evaluated, the period of time covered by the sampling programs varied, as did the types of analyses. For groundwater and soil, the radiation exposure evaluation was based on uranium detected in private wells and in off-site soil. For the air pathway, radon with its decay products and external exposures were evaluated separately. Other radionuclides evaluated include total uranium; strontium 90; technetium 99; cesium 137; radium 226 and 228; thorium 228, 230, and 232; neptunium 237; and plutonium 238 and 239. For surface water, the radiation exposure evaluation was based on strontium 90, radium 226 and 228, and total uranium. For the biota pathway, there were four major groups analyzed for different radioactive contaminants:

- Vegetables analyzed for total uranium;
- Meat analyzed for uranium 234, 235/236, and 238, thorium 228, 230, and 232, radium 226, strontium 90, cesium 137, and plutonium 238 and 239;
- Milk analyzed for total uranium; and for uranium 234, 235/236, and 238, thorium 228, 230, and 232, radium 226 and 228, protactinium 234, strontium 90, and cesium 137; and
- Fish analyzed for total uranium.

Although it would be expected that the radioactive material with the highest concentration in media would be uranium, that is not always the case. In surface water, for example, the radioactive contaminants with the maximum concentrations downstream of the effluent line in the Great Miami River are technetium 99 in 1992, radium 226 in 1993, and radium 228 in 1994 and 1995 (DOE 1972–1999). Furthermore, the radioactive contaminant with the highest concentration is not always the main contributor to the exposure dose—for example, the largest contributor to current bone surface dose, especially in children, is radium 228.

Table 38 summarizes ATSDR's estimated past and current exposure doses (to the whole body, bone surface, and lungs) from radioactive contaminants in completed and potential exposure pathways. Of course, the highest doses were estimated for past exposure, primarily in 1959 and 1960 for radon and in 1955 for uranium. Inhalation of radon and radon decay products in air posed a human health hazard under past conditions at the site, mainly for persons living immediately west of the silos. However, airborne uranium particles also contributed significantly to estimated past lung (committed equivalent) doses for persons living immediately east/northeast of the site. Those persons living immediately east/northeast of the site would also have received the highest past whole body (committed effective) doses. Since the plant discontinued operations in 1988, most of the airborne contaminants are primarily radon and radon decay products from Silos 1 and 2 at much lower concentrations than in the past.

Radiological Effects of Radioactive Materials and Radiation Exposure

When radioactive materials undergo spontaneous transformation (decay), they transform into other elements, emitting radiation energy in the form of particles (such as alpha or beta particles) or waves (such as gamma rays or x-rays). Each radioactive material has a unique decay pattern, and each transformation (decay) gives off a unique energy level. Alpha particles are relatively large and do not travel far, depositing all of their energy near where they were emitted. Gamma rays and x-rays are energy waves, not particles, and can travel long distances, releasing their energy gradually. Each emission interacts with tissue differently and has a different effect on the tissues and organs of the body. Also, a person's age affects how much of a contaminant he or she ingests/inhales and how sensitive his or her tissues/organs are to radiation interactions. For each radioactive contaminant and age group, there are unique conversion factors used to convert from concentrations in media that a person may ingest, inhale, or be externally exposed to the potential dose received by that person.

In the health implication analysis, potential effects from each radioactive material in each route of exposure for different age groups were reviewed for each year of potential exposure. Potential health effects from total radiation exposure from all routes were considered for both whole body doses and organ doses for children and adults, using the year of the highest potential dose and lifetime doses (for this site, ATSDR used 53 years, from 1951 through 2003; additional years until the site remedial activities are complete in 2006 should add very little to the total).

When evaluating the *carcinogenic* effect of radiation, ATSDR scientists use the committed effective dose (CED) for the whole body of 5,000 millirem (50 millisieverts) over 70 years as a comparison value. ATSDR believes the CED of 5,000 millirem (50 millisieverts) over 70 years is protective of human health. This value was derived after reviewing the peer-reviewed literature and other documents developed to review the health effects of ionizing radiation (ATSDR 1999c). In 1994, the General Accounting Office (GAO) released a report reviewing the US radiation standards and radiation protection issues (GAO 1994) and in 2000 further refined their results (GAO 2000). According to the later report, "conclusive evidence of radiation effects is lacking below a total of about 5,000 to 10,000 millirem, according to scientific literature," which was also the consensus of experts they interviewed (GAO 2000). Between 10 rem and 5 rem, the data are not clear as to the health effects. Below 5 rem the effects are not observed—they are only assumed based on four possible dose-response curves.

ATSDR used a lifetime scenario for someone living immediately east/northeast of the site and someone living immediately west of the silos since the plant began operations in 1951. This scenario would have the person exposed as a child for 10 years and 28 years as an adult when the plant was operating and 15 years since it ceased operating. The CED for the hypothetical individual east/northeast of the site would be between 7 and 8.4 rem (0.07 and 0.084 sieverts), and the CED for the individual west of the silos would be approximately 2.2 rem (0.022 sievert), mainly from airborne exposure. The dose for the individual east/northeast of the site exceeds ATSDR health-based guideline for lifetime dose but is in the inconclusive range according to GAO. Although the CED for the individual west of the silos does not exceed ATSDR's health-based guideline for whole body exposure, this individual would have received the highest lung dose. Because this individual's lung dose would be almost solely from radon progeny and the individual east/northeast of the site would have received a lower dose but from radon and uranium, the likelihood of dying from lung cancer would be approximately the same for both individuals. ATSDR's estimate of the percentage increase in the number of lung cancer cases was slightly higher that CDC's FDRP which is due to ATSDR evaluating the maximum range doses to the maximally exposed individuals. According to CDC's reported doses from *past* exposure to radon, radon decay products, and other radioactive contaminants in air, there was a moderate to high increased likelihood that nearby residents would develop lung cancer (CDC 1998). CDC's Fernald Dosimetry Reconstruction Project and Fernald Risk Assessment Project indicate that while the facility was operating, nearby residents were exposed to radioactive materials from the site at levels that were greater than expected from background sources, and that these exposures resulted in an increased number of lung cancer deaths: 1% to 12% greater than expected in a community without exposures from the Fernald facility (CDC 1998). ATSDR also looked at the likelihood of developing or dying from bone cancer for these two scenarios. The likelihood of developing bone cancer would increase about 7% for the maximally exposed individual to the east/northeast but minimal for the resident living to the west of the silos.

ATSDR typically uses the current MRL for external, chronic exposure to ionizing radiation (100 mrem or 1 mSv per year CED) as the comparison value when evaluating non-carcinogenic

effects of radiation. This level is based on whole body exposure to average background radiation in the United States (360 mrem/yr or 3.6 mSv/yr) with added uncertainty factors for human variability. This annual dose has not been associated with adverse health effects or increases in incidences of any type of cancers in humans or other animals (ATSDR 1999c). The only maximally exposed individuals who would have exceed this MRL were persons living close to the east/northeast side of the site when the plants were operating. Most of their exposure would have been from airborne radioactive particles as described previously. Hopefully these persons are part of the FMMP.

*Of the exposure pathways evaluated for radioactive contaminants from the Fernald site, none pose a human health hazard under **current** conditions. Likewise, total current radiation exposure to nearby residents for all pathways combined is not likely to result in adverse human health effects.*

Table 38. Estimated maximum total radiation doses from 1 year of exposure for persons living near the site

	Pathway	PAST EXPOSURES		CURRENT EXPOSURES	
		Committed Effective Dose (whole body) in mrem (mSv)	**Committed Equivalent Dose (bone surface or lung) in mrem (mSv)**	**Committed Effective Dose (whole body) in mrem (mSv)**	**Committed Equivalent Dose (bone surface or lung) in mrem (mSv)**
East/Northeast of site		*Prior to 1989*		*Since 1989*	
Child	Air (uranium)	~280(2.8)–1700(17)*	~2300(23)–13700(137)lung* ~270(2.7)–1500(15)bone surface*	0.58 (0.006)	4.3(0.043) lung 15.4(0.154) bone surface
	Radon†	- - - - -	~9500(95)–14300(143)*lung	- - - - -	~1060(11)–1600(16) lung
	Soil	0.2 (0.002)	2.9(0.03) bone surface	0.74 (0.007), <1992 0.38(0.004), >1992	25.27 (0.253) bone surface, <1992 11.37 (0.114) bone surface, >1992
	Biota‡	3.0 (0.03)	45(0.45)bone surface	21(0.21)	643 (6.43) bone surface
	Total dose	**~283(2.8)–1703(17)**	**lung:~11800(118)–28000(280) bone surface:~318(3.2)–1548(15)**	**22 (0.22)**	**lung:~1064(11)–1604(16) bone surface: 670(6.7)–684(6.8)**
Adult	Air (uraium)	~140(1.4)–850(8.5)*	~1100(11)–6600(66)lung* ~170(1.7)–1000(10)bone surface*	8.0 (0.08)	36.7(0.367) lung 247(2.47) bone surface
	Radon†	- - - - -	~9500 (95) lung*	- - - - -	~1060 (11) lung
	Biota‡	2.1 (0.02)	34(0.34) bone surface	5.7 (0.057)	177 (1.77) bone surface
	Total dose	**~142(1.4)–852(8.5)**	**lung:~10600(106)–16100(161) bone surface:~204(2.0)–1034(10)**	**14 (0.14)**	**lung:~1097 (11) bone surface: 424(4.24)**
South of site		*In 1960s and 1970s*		*Since 1989*	
Child	Groundwater	45 (0.45)–320 (3.2)	662(6.62)–4731(47.31) bone surface	<8 (<0.08)–22 (0.22)	<113(<1.13)–321 (3.21)bone surface
	Biota‡	3.0 (0.03)	45 (0.45) bone surface	21 (0.21)	643 (6.43) bone surface
	Total dose	**48 (0.48)–323 (3.23)**	**707 (7.07)–4776 (47.76)**	**<29(<0.29)–43(0.43)**	**<756(<7.56)–964(9.64)bone surface**

	Pathway	PAST EXPOSURES		CURRENT EXPOSURES	
Adult	Groundwater	50 (0.5)–360 (3.6)	783(7.83)–5602(56.02)bone surface	<9(<0.09)–24 (0.24)	<134 (<1.34)–380 (3.80)bone surface
	Biota‡	2.1 (0.02)	34 (0.34) bone surface	5.7 (0.057)	177 (1.77) bone surface
	Total dose	**52 (0.52)–362(3.62)**	**817 (8.17)–5636(56.36)**	**<15(<0.15)–30(0.30)**	**<311(<3.11)–557(5.57)bone surface**
West of site		*Before 1989*		*Since 1989*	
Child	Radon	- - - - -	~37895(379)–60632(606) lung	- - - - -	~2140 (21.4)–8544(85.4) lung
	Surface water	3.66 (0.037)	82 (0.82) bone surface	2.6 (0.03)	88 (0.88) bone surface
	Biota‡	3.0 (0.03)	45 (0.45) bone surface	21 (0.21)	643 (6.43) bone surface
	External	8.0 (0.08)–84 (0.84)	- - - - -	7 (0.07)–59(0.59)	- - - - -
	Total dose	**15 (0.15)–91(0.91)**	**lung:~37895(379)–60632(606) bone surface: 127(1.27)**	**31(0.31)–83(0.83)**	**lung:~2140(21.4)–8544(85.4) bone surface: 643(6.43)**
Adult	Radon	- - - - -	~37,895 (379) lung	- - - - -	2140 (21.4)–5340(53.4) lung
	Biota‡	2.1 (0.02)	34 (0.34) bone surface	5.7 (0.06)	177 (1.77) bone surface
	External	8.0 (0.08)–84 (0.84)	- - - - -	7–59 (0.07–0.59)	- - - - -
	Total dose	**10(0.10)–86(0.86)**	**lung:~37895(379) bone surface: 34(0.34)**	**13(0.13)–65(0.65)**	**lung: 2140(21.4)–5340(53.4) bone surface: 177 (1.77)**
North/Northwest of site		*Before 1989*		*Since 1989*	
Child	Biota only‡	3.0 (0.03)	45 (0.45)bone surface	21(0.21)	643 (6.43) bone surface
Adult	Biota only‡	2.1 (0.02)	34 (0.34) bone surface	5.7 (0.06)	177 (1.77) bone surface

Key:

* Information from CDC's FDRP and ATSDR scenarios used for calculations. FDRP reports total 38-year dose; ATSDR table shows annual dose.

† According to NCRP Report 78, risk of developing lung cancer from equivalent one-year dose is less for a child then for an adult, but risk from equivalent thirty-year dose is approximately equal for a child or an adult (at time of first exposure) (NCRP 1984).

‡ Biota doses in the past were estimated during the FDRP and were primarily based on uranium releases. Current biota doses are based on concentrations of all radioactive analytes with the maximum value for radium 228 in milk being the driving factor. ATSDR believes that the maximum current dose estimates for biota are in the higher range of probable doses but the past estimates are probably too low.

Arsenic

Arsenic was detected in soil at this site and determined to be a contaminant of concern for inhalation of re-suspended soil particles. The maximum off-site soil concentrations are 9.2 mg/kg northwest of the site and 5.3 mg/kg northeast of the site. From the data reviewed, the mean concentration of off-site soil samples is 5 mg/kg. ATSDR's CV is 0.5 mg/kg; however, the CV is based on 70 years of exposure, 365 days per year, and a risk of developing cancer of one in a million. It is also based on the conservative assumption that no threshold exists for cancer.

Arsenic is present in the environment in both inorganic and organic forms. (Note: the term "organic" refers to compounds containing carbon and hydrogen.) Inorganic forms of arsenic predominate in soils and are more toxic than organic forms (NEPI 1998). When humans and other mammals are exposed to inorganic arsenic, they metabolize it to the much less toxic methylated organic form, which is readily excreted from the body. This methylation is effective as long as the inorganic arsenic intake remains below a level of 0.2 to 1 mg of arsenic per day (ATSDR 1998), indicating that people can tolerate a certain level of arsenic exposure. At higher exposure levels (a "threshold"), the body's capacity to detoxify arsenic appears to be exceeded or saturated, leading to increased arsenic in the blood and possible adverse health effects. At the concentrations found near the site, a person would have to ingest between 22 and 189 grams of soil per day to reach this "threshold" level. Children normally ingest more soil than adults and can ingest relatively little soil one day and large amounts the next day. Also, some children, who ingest non-food items (i.e., soil) early in their childhood, can consume 5 to 10 grams/day. Children who do not exhibit this behavior ingest about 50 to 100 mg/day. No one in this site's neighboring community would be consuming enough soil to reach this "threshold."

Arsenic was also detected in off-site groundwater. The maximum concentration of arsenic detected in groundwater from a *private* well was 13,100 µg/L; however, this value is suspicious because it is greater than an order of magnitude higher than any other concentration for this well and several orders of magnitude higher than arsenic concentrations in most other private wells. However, this well was located up gradient of the site but was consistently above EPA's maximum contaminant level for drinking water. The next highest concentration of arsenic in a *private* well was 94 µg/L in 1988 in a well near New Haven Road south of the site. Arsenic was also detected in some *monitoring* wells in the South Plume area (maximum current concentration of 117 µg/L).[7] The estimated maximum arsenic exposure dose is 0.009 mg/kg/day for a child and 0.003 mg/kg/day for an adult. Although these doses exceed the health-based guideline for chronic ingestion of arsenic (0.0003 mg/kg/day) and the child dose exceeds the health-based guideline for acute ingestion of arsenic (0.005 mg/kg/day), the health-based guidelines are conservative to account for variation in sensitivity among the population and other factors discussed below.

[7] ATSDR acknowledges that these levels of arsenic may not be related to site activities.

The lowest levels at which toxicity—including skin and gastrointestinal effects—have been reported in humans are 0.014 to 0.05 mg/kg/day. These findings are based on a study of Taiwanese who drank arsenic-contaminated water for 45 years (Tseng et al.1968; Tseng 1977). EPA derived a health guideline of 0.0003 mg/kg/day for adverse effects on the skin (e.g., hyperpigmentation, keratosis) and a skin cancer slope factor of 1.5 $(mg/kg/day)^{-1}$, based on the Taiwanese study (IRIS 1998). Although ATSDR's estimated ingestion doses are lower than levels shown to cause adverse effects in the Taiwanese, the study has important limitations that should be mentioned. First, the study reported an association between arsenic in drinking water and skin cancer but failed to account for a several potential confounding factors, including exposure to other sources, genetic susceptibility to arsenic, and poor nutritional status of the exposed population. Therefore, arsenic exposure may have been underestimated, leading to an overestimation of cancer effects associated with exposure levels. Second, the cancer slope factor is based on the conservative assumption that no threshold exists for cancer. As discussed previously, arsenic carcinogenicity appears to have a threshold. Lastly, the adverse effects observed in the Taiwanese were due to absorbed arsenic. Considering the limitations of the study and the conservative assumptions in the dose calculations, ATSDR scientists conclude that adverse effects are not likely from ingestion of arsenic in the groundwater and soil.

Manganese

Although manganese concentrations did not exceed health-based guidelines during the limited private well sampling, three DOE monitoring wells in the South Plume showed maximum concentrations in the range from 1,729 µg/L to 5,410 µg/L. If people had drunk this water before an alternate water source was supplied, the exposure doses would have ranged from 0.133 to 0.416 mg/kg/day for a child and 0.049 to 0.155 mg/kg/day for an adult. ATSDR's interim health guideline for manganese is 0.07 mg/kg/day (ATSDR 1997). Manganese was also detected in off-site surface soil up-gradient of the site and probably is not related to the site.

Manganese is a naturally occurring element that is essential for normal functioning of the human body. Toxicity in humans has been associated with both deficiencies and excess intake of manganese. Sub-optimal manganese intake may be more of a concern than excess intake: several diseases including multiple sclerosis, cataracts, osteoporosis, and epilepsy may be associated with low levels of manganese in the body (ATSDR 1997). Two cases of manganese deficiency have been reported for persons consuming 0.11 to 0.34 mg of manganese per day (Doisy 1973; Friedman et al.1987). Because manganese is essential in the human diet, the National Research Council has established a range of "estimated safe and adequate daily dietary intakes" (ESADDIs) for manganese. The ESADDIs are 0.3 to 2.0 mg/day for children under 6 years of age and 2 to 5 mg/day for persons over 11 years of age (NRC 1989). The World Health Organization estimates that the average consumption of manganese in the adult diet ranges from 2 to 9 mg/day, and that an intake of 8 to 9 mg/day is "perfectly safe." ATSDR's interim health guideline for manganese ingestion (0.07 mg/kg/day) is equivalent to a daily intake of 4.9 mg for a 70-kg adult and 0.91 mg for a 13-kg child. The daily intake for a 70-kg adult is within the World

Health Organization's "perfectly safe" range, and the daily intake for a 13-kg child is within the ESADDI range. Manganese deficiency is not likely for Fernald area residents, however, additional sources of manganese are contributed by the diet and other environmental sources. Even considering these other sources, ATSDR estimates that the daily intakes of manganese for the Fernald area residents are likely to be within "safe and adequate" ranges and are not likely to result in adverse health effects.

Health Outcome Data

In general terms, health outcome data are morbidity data (incidence rate of a disease in a population) and mortality data (incidence rate of death in a population). ATSDR evaluated these data to determine the overall health status of the Fernald community and to identify specific adverse health effects that may be occurring as a result of exposure to chemicals and radioactive materials from the Fernald site.

ATSDR evaluated the following sources of health outcome data for this public health assessment: (1) results of cancer analyses among participants in the Fernald Medical Monitoring Program (Pinney 1999b, 2001); and (2) a Master of Science thesis on β-2-microglobulin levels in urine of potentially exposed persons living near the Fernald site (Kammer 1998).

NIOSH has conducted and initiated several investigations involving Fernald workers. A brief description of these projects is provided below. However, these studies were not used for this public health assessment because they pertain to Fernald workers.

The community group FRESH conducted a survey of adverse health effects reported by residents of the Fernald community which indicates that Fernald residents are concerned about various types of cancer and non-cancer effects. A compilation of the survey findings is provided at the end of the Community Concerns section of this report.

The Fernald Medical Monitoring Program (FMMP) and the Fernald Workers Medical Monitoring Program (FWMMP)

In January 1985, Fernald area residents filed a class action lawsuit against National Lead of Ohio (the site manager from 1954 to 1985) and DOE. These legal actions resulted in the establishment of a Settlement Fund and two programs: the Fernald Medical Monitoring Program (FMMP) and the Fernald Workers Medical Monitoring Program (FWMMP).

The objectives of the FMMP are to (1) provide a complete medical evaluation of the current health status of eligible persons, (2) provide a comprehensive evaluation of risk factors for illnesses or diseases of participants, (3) provide education to participants on how to modify risk factors for illness or disease, and (4) establish a good baseline database that may be useful for subsequent epidemiological research (Pinney 1999a, 2001).

Persons who lived or worked within 5 miles (8.1 kilometers) of the former Feed Materials Production Center for any 2-year continuous period between January 1, 1952 and December 18, 1984 are eligible to participate in the residents' program (FMMP). Persons who worked at the

production center as employees of National Lead of Ohio or National Lead Industries are excluded from participating. Participation is on a voluntary basis. Participants initially complete questionnaires on health risk, health status, lifestyle, and possible exposure. A physician administers a history, a physical examination, and medical tests. The first set of formal examinations was initiated in December 1990. From 1990 to November 1998, medical examinations were offered every 3 years. As of December 1998, medical examinations are offered every 2 years. The FMMP is slated to operate for 25 years in total. Confidentiality of medical records is maintained, and data on participants are stored in a computerized database. Participants receive medical advice based on the results of their examinations and tests. In addition, each woman over 40 receives an annual mammogram.

As of June 1999, a total of 8,520 adult participants (persons 18 years or older at the time of their first examinations) have enrolled in the FMMP and had their first medical examinations. As of December 1, 1998, the age range of participants is 19 to 95, and slightly more than half (55%) of the participants are women, almost all (99%) are Caucasian, most (73%) are married, and most (84%) have education beyond high school (Pinney 1999b).

Two research questions are being addressed by the FMMP. The first question addresses whether the number of newly diagnosed cancer cases among FMMP participants, for the first 4 years in the program (first medical examination plus 48 months of followup), is greater than what would be expected among a similar population. The analysis includes FMMP participants (7,937 persons in FMMP) who had their first medical examinations before December 1, 1993 and who were diagnosed at the first medical exam plus 48 months (i.e., each participant contributes about one to four person-years).

Analyses were performed for 16 cancer systems and for all cancer sites combined. Four different comparison populations were used: (1) the National Cancer Institute's Surveillance, Epidemiology, End Result (SEER) data for all of the United States; (2) the SEER data for Ohio; (3) the Ohio Cancer Surveillance (for Butler, Warren, and Clermont counties); and (4) the Ohio Cancer Surveillance (for Ohio as a whole). The *a priori* best comparisons are the SEER data for Ohio and the tri-county Ohio Cancer Surveillance data.

The findings for the FMMP population indicate that the number of new cancer cases for three types of cancer (urinary system and kidney/renal pelvis, melanoma of the skin, and prostate) was greater than expected. The incidence of urinary system cancer in the FMMP population was statistically significant with all four comparisons. Within the urinary system, the incidence for kidney/renal pelvis cancer was significant compared only to the tri-county Ohio area (2.50, 1.14–4.75). The incidence of melanomas of the skin and prostate cancer was significant compared only to the SEER Ohio data; e.g., melanomas (2.22, 1.11–3.97) and prostate cancer (1.53, 1.12–2.70). The researchers acknowledge that the greater than expected incidence of prostate cancer was possibly attributable to the introduction of a new diagnostic test (i.e., PSA)

that improved the identification of existing cases, rather than an actual increase in the number of new cases. Although not statistically significant, the expected number of new lung cancer cases among FMMP participants increased from 1% to 12% over the expected number of cases. This is consistent with the predictions made in the CDC's Community-Based Lung Cancer Risk Assessment (Pinney 1999b).

The researchers state that the FMMP volunteer population is representative of the general population, although they acknowledge possible sources of bias because of a "healthy volunteer screening effect" and because a volunteer study population, rather than a representative sample of the entire Fernald community, was used. These results are considered to be screening-level, because the analysis addressed only whether there is an excess of a specific type of cancer in neighboring residents. No data analyses were performed to determine if this excess is related to historical radiation or chemical exposures from the Fernald site (Pinney 1999b).

The second research question being addressed is whether the rate of certain chronic medical conditions, reported by participants in the FMMP at the time of their first medical examination, is greater than the rate reported from national health databases—the National Health and Nutrition Examination Survey (NHANES) and National Health Interview Survey (NHIS). The chronic conditions being analyzed were selected by a review of the scientific literature, interviews with the medical community, and input from Fernald residents (Pinney 1999b). This phase of analysis of FMMP data is being funded by ATSDR's Division of Health Studies.

The FWMMP (for former workers) is similar to the FMMP (for residents), although there are some important differences. The FWMMP involves workers and information collected from participants focused on occupational histories. Participants are re-examined annually. The FWMMP has about 3,000 participants. Because the focus of this public health assessment is on community residents, rather than workers, this report does not discuss the FWMMP in depth.

Effects of Uranium-Contaminated Drinking Water on Urinary β-2-Microglobulin Concentration

A retrospective study, conducted by a Master of Science student from the University of Cincinnati College of Medicine, evaluated renal (kidney) effects for residents living near the Fernald site who drank water contaminated with uranium (Kammer 1998). Contaminated wells were identified by use of water concentration measurements with results equal to or greater than 20 μg uranium per liter of water. These measurements were made by contractors for DOE, ODH, and OEPA. The exposure group (25 people) was defined as participants in the FMMP who drank water from contaminated wells within the area of the South Plume, as characterized in 1991. The control group (569 people) consisted of Fernald area residents who lived within 4 to 5 miles of the Fernald site and who drank water from wells not contaminated by uranium. The age and sex distribution of South Plume residents and control residents are similar (Kammer 1998).

The biological marker β-2-microglobulin was used to measure the effect of uranium on kidney function. This marker is not specific for uranium-induced renal toxicity, because there are numerous other diseases and chemicals (e.g., chronic active and viral hepatitis, preeclampsia, rheumatoid arthritis) that cause alterations in urinary β-2-microglobulin concentrations (Kammer 1998). Concentration measurements for urinary β-2-microglobulin were not available for all South Plume residents or for the entire control group. (They were available for 24 South Plume residents and for 499 people in the control group.) Likewise, concentrations of urinary β-2-microglobulin, standardized for creatinine, were available only for 22 South Plume residents and 496 control residents. Mean urine β-2-microglobulin concentrations in the South Plume and control groups were not statistically different, and mean urine β-2-microglobulin concentrations, standardized for creatinine, were also not statistically different.

Although the findings indicate that South Plume residents did not have increased urine β-2-microglobulin concentrations compared to the control group of residents, several issues must be considered in the interpretation of these results (Kammer 1998). The most important of these is the relatively imprecise estimation of uranium exposure and the use of urinary β-2-microglobulin as a biological marker of effect.

For exposure to uranium, water measurements were used to define the boundaries of the South Plume in 1991. Actual measurements of uranium concentrations at residences within the plume (and outside the plume) would provide more precise estimates of exposure concentrations and changes in concentrations over the period of exposure. This is important because uranium concentrations varied over time, and maximum concentrations were presumably present in the 1960s, not the 1990s. In addition, such data would help minimize misclassification bias (based on exposure).

The use of urinary β-2-microglobulin concentrations as a marker of effect, and the length of time between exposure and measurement of β-2-microglobulin concentration, may have hampered the ability of this study to detect positive associations if they were present. As mentioned previously, concentrations of uranium in the South Plume varied over time. The maximum period of exposure was presumably the 1960s; concentrations decreased substantially from the 1960s to the 1990s. Many residents were provided bottled water to drink in 1984. Measurements of urinary β-2-microglobulin concentrations were made in the late 1990s, 30 years or more after maximum exposure may have occurred. The kidneys of South Plume residents may have recovered from any toxicity by the time urinary β-2-microglobulin concentrations were measured. Few studies have examined the *chronic* effects of uranium on the kidney and its ability to repair itself once exposure to uranium has ended (Kammer 1998). While urinary β-2-microglobulin is a valid marker for *acute* toxicity, it may not be appropriate for use in chronic exposure conditions.

NIOSH Activities

NIOSH has conducted various investigations involving past and current workers at the Fernald site. NIOSH is also conducting an exposure assessment of hazardous waste, decontamination and decommissioning, and clean-up workers and a retrospective exposure assessment for workers at the Fernald plant. These and other NIOSH activities are not discussed in detail in this report, which addresses health issues related to the surrounding community. Further information about NIOSH's activities at the Fernald site can be obtained by calling or writing the NIOSH contact person listed in the For Additional Information section of this report.

Community Concerns

Background

ATSDR representatives first met with members of the Fernald community in May 1992, during the initial visit to the Fernald area. Many times since then, ATSDR representatives have traveled to the Fernald area to meet with various members of the community, both in public and private meetings. The purpose of the visits was to learn more about the Fernald site and to hear from community members about their health concerns.

The most concerted efforts to compile community concerns in the Fernald community were public availability sessions (or open-house meetings) sponsored by ATSDR on December 6, 7, and 8, 1993. Four open-house meetings were held, two each in Crosby and Ross, at which concerned citizens met individually, in pairs, or in small groups with ATSDR representatives. The public availability sessions were advertised widely in local and area newspapers. The advertisements stated that the purpose of the availability sessions was to hear the community's health concerns related to the Fernald Environmental Management Project and the former Feed Materials Production Center. ATSDR representatives, with the assistance of personnel from Boston University's School of Public Health, EPA's NAREL, and CDC's NCEH, met with approximately 110 people and recorded their concerns, questions, and comments. These sessions attracted a reasonable cross-section of the community, including residents affiliated with local government, representatives of local community organizations, and former workers at the site. Many of those who attended gave their names, but some spoke anonymously. ATSDR made every effort to maintain confidentiality for the attendees; representatives of the news media were not allowed to sit in on or record conversations.

In addition to the public availability sessions, ATSDR representatives have attended meetings sponsored by DOE, FRESH, NCEH, and NIOSH. ATSDR representatives have attended meetings of the Fernald Citizens Advisory Board, formerly the Fernald Citizens Task Force, since March 1994. Upon receiving an invitation from the board, an ATSDR representative has been an *ex-officio* member since December 1995. Representatives have also attended meetings of the Fernald Health Effects Subcommittee (FHES) since its inception and reviewed and compiled public concerns sent to the FHES by mail. (Refer to Appendix C.) ATSDR scientists also reviewed information collected by the community group FRESH concerning health problems reported by local residents. ATSDR staff have spoken with many community members by telephone and visited with them in their homes or at other locations.

With the assistance of personnel from NAREL, ATSDR conducted an environmental sampling program in the Fernald area. Visits to the area to collect environmental samples, including setting up radon detection canisters in residents' homes, provided ample opportunity to talk with many residents of the area, particularly those who live close to the facility.

Fernald Community Concerns

Throughout all the meetings and activities sponsored and attended by ATSDR representatives, an ongoing list of community concerns related to the Fernald site has been kept. In many cases, ATSDR representatives do not know where some of the respondents live or lived, because this information was not provided.

ATSDR grouped the community concerns under the following headings:

HEALTH CONCERNS
Cancers
Non-Cancer Effects
ENVIRONMENTAL EXPOSURES
Air
Soil
Surface Water
Groundwater
Biota
SPECIFIC POPULATIONS' CONCERNS
PROCEDURAL CONCERNS
Remediation
Lack of Trust
Emergency Response
Monitoring or Sampling
General
Recommendations by the Public

ATSDR did not record concerns that were addressed during the public availability sessions or other meetings, or concerns that had already been reported and recorded. A summary of the community concerns and ATSDR's responses is presented in Appendix C of this report. Also included in Appendix C is a summary of concerns submitted to the FHES and compiled by FRESH.

Conclusions

ATSDR scientists evaluated chemicals and radioactive materials in completed and potential exposure pathways for the Fernald site. ATSDR scientists reached the following conclusions:

- Ingestion of uranium in water from privately owned off-site wells in the South Plume poses a *public health hazard* under *past* conditions at the site. This pathway poses a health hazard because available information indicates that people drinking water from wells south of the site were exposed to contaminants at levels that could result in adverse health effects.

- Inhalation of radon and radon decay products poses a *public health hazard* under *past* conditions at the site for individuals living immediately west of the silos. This pathway poses a public health hazard because available information and the estimation of exposures from modeling this pathway indicate that people were potentially exposed to contaminants at levels that could result in adverse health effects.

- Inhalation of the combination of radon, radon decay products, and airborne uranium poses a *public health hazard* under *past* conditions at the site for individuals living immediately east/northeast of the site. This pathway poses a public health hazard because available information and the estimation of exposures from modeling this pathway indicate that people were potentially exposed to contaminants at levels that could result in adverse health effects.

- *Past* off-site exposure to airborne non-uranium chemicals was possible; however, given the lack of information concerning quantities and frequency of most releases, ATSDR is not able to determine whether the off-site community was exposed to these chemical concentrations at levels of health concern while the plant was operating. (Some releases were evaluated and were not at levels that would cause adverse health effects to the off-site community.)

- One *current* exposure pathway (inhalation of airborne non-uranium particles) was unable to be evaluated. Although the total suspended particles are not at a level of health concern for the off-site community, specific off-site airborne concentrations of some elements known to be at the site (i.e., beryllium) were not able to be determined.

- According to the information reviewed by ATSDR, *there are no other exposure pathways that pose a public health hazard to the surrounding community under current conditions at the site.*

Although human exposure to chemicals and radionuclides may have occurred via other exposure pathways, the levels and conditions of exposure (e.g., duration, frequency, route of exposure) were not sufficient to cause adverse health effects, even for the most sensitive individuals.

Recommendations

On the basis of the information reviewed, ATSDR recommends the following:

- Analyses of boundary monitoring for airborne particles should include specific analyses for certain chemicals which had been used at the site and which have very low concentrations that can be of health concern (i.e., beryllium). The analyses should also include a determination of particle size to aid in inhalation calculations and to show compliance with EPA standards.

- DOE should continue monitoring groundwater on the eastern site boundary and in the South Plume. These are the predominant directions of the groundwater movement under the site and the movement of two groundwater plumes (currently one is on site and one is off site). This monitoring should include analyses for contaminants that may be drawn into the South Plume (from sources other than the Fernald facility) due to groundwater remedial activities.

- If additional pertinent information becomes available, an in-depth assessment of *past* exposure to airborne non-uranium chemical contaminants and chemical contaminants in privately owned residential wells near the Fernald facility should be conducted. Preliminary analyses of data from residents participating in the Fernald Medical Monitoring Program (FMMP) suggest a higher than expected occurrence of kidney cancer; however, it is not known whether any of the observed increases in adverse health effects are related to chemicals and radioactive releases from the Fernald site.

- Monitoring for radon and radon daughters should be continued in the Fernald area during remedial activities at the site, particularly those involving the K-65 silos. Additional sampling locations should be added (especially on the west fenceline) during remediation of the silos.

Additional Considerations

Further evaluation of possible risk factors for adverse health effects among participants in the Fernald Medical Monitoring Program should be considered.

Public Health Action Plan

The Public Health Action Plan for the Fernald Closure Project (formerly the Fernald Environmental Management Project and the Feed Material Production Center) describes public health actions taken or planned to be taken by ATSDR, DOE, or responsible state agencies at, and in the vicinity of, the site based on the recommendations of this public health assessment. The purpose of the plan is to ensure that this public health assessment not only identifies public health hazards, but that it also provides a plan of action designed to mitigate and prevent adverse human health effects resulting from exposure to hazardous substances in the environment. Public health actions completed, being implemented, or planned are as follows:

Public Health Actions Taken:

- In 1984, DOE formally established a groundwater monitoring program for private well owners near the site. Property owners could request their residential wells to be sampled for uranium, and if the samples showed uranium concentrations above background levels, the owners were offered bottled water for household use. In 1996, DOE supplied some residences including those in the South Plume area with public water. In 1996, DOE also discontinued sampling private wells but routinely samples a network of monitoring wells.

- In ATSDR's 1995 health consultation on radon and radon decay products emissions from the silos, ATSDR highlighted some issues with DOE's monitoring program: (1) the continuous monitors were unreliable when used outside their optimal temperature range; (2) DOE did not obtain duplicate hourly radon measurements, and (3) back-up monitors were not maintained to replace inoperable detectors. Since then, DOE began using thermal jackets for their monitors and has adequate back-up and procedures for response to inoperable monitors.

- ATSDR independently monitored for radon at several off-site locations from 1993 until October 2003.

- DOE concurs with ATSDR's recommendation for continued radon monitoring during remedial activities at the site. DOE's radon monitoring program will continue until the site is fully remediated. Recently DOE has increased the number of radon monitors at the property fenceline, purchased replacement radon monitors capable of operating in colder environmental conditions, and inspects monitors daily to ensure proper operation. OEPA also has three continuous radon monitors located at or near the site.

- DOE is performing groundwater monitoring and remedial activities for a portion of the Great Miami Aquifer that has been impacted at levels exceeding the FEMP's final cleanup levels. FEMP's Integrated Environmental Monitoring Plan, which is modified every two years,

describes the locations of the monitoring wells, the sampling program, the extraction wells, the re-injection wells, and the treatment processes for the water. DOE will continue this program until the site remedial activities are complete.

- ATSDR's Division of Health Education and Promotion has co-sponsored with the University of Cincinnati College of Medicine and Mercy Hope Partners three educational programs for health care professionals in the Cincinnati area.

Public Health Actions Planned:

- No in-depth assessment of *past* exposures to airborne non-uranium chemical contaminants or non-uranium chemical contaminants in privately-owned residential wells near the site is planned at this time. If additional pertinent information becomes available, this assessment should be considered.

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References

[ATSDR] Agency for Toxic Substances and Disease Registry. 1992. Public health assessment guidance manual. Atlanta: US Department of Health and Human Services.

[ATSDR] Agency for Toxic Substances and Disease Registry. 1995a. Health consultation for milk produced near the Fernald Environmental Management Project. Atlanta: US Department of Health and Human Services; 1995 Jun.

[ATSDR] Agency for Toxic Substances and Disease Registry. 1995b. Health consultation for the K-65 silos. Atlanta: US Department of Health and Human Services; 1995 May.

[ATSDR] Agency for Toxic Substances and Disease Registry.1996a. Health consultation for nonpotable use of uranium contaminated groundwater near the Fernald Environmental Management Project. Atlanta: US Department of Health and Human Services; 1996 Feb.

[ATSDR] Agency for Toxic Substances and Disease Registry. 1996b. Health consultation for consumption of produce grown near the Fernald Environmental Management Project. Atlanta: US Department of Health and Human Services; 1996 Jan.

[ATSDR] Agency for Toxic Substances and Disease Registry. 1997. Toxicological profile for manganese. Atlanta: US Department of Health and Human Services; 1997 Sep.

[ATSDR] Agency for Toxic Substances and Disease Registry. 1998. Toxicological profile for arsenic. Atlanta: US Department of Health and Human Services; 1998 Aug.

[ATSDR] Agency for Toxic Substances and Disease Registry. 1999a. Demographic analysis for populations residing within one mile, five kilometers, and ten kilometers of the Fernald site. Atlanta: US Department of Health and Human Services Public Health Service; Spatial Analysis Activity.

[ATSDR] Agency for Toxic Substances and Disease Registry. 1999b. Toxicological profile for uranium. Atlanta: US Department of Health and Human Services Public Health Service; 1999 Sep.

[ATSDR] Agency for Toxic Substances and Disease Registry. 1999c. Toxicological profile for ionizing radiation. Atlanta: US Department of Health and Human Services; 1999 Sep.

Boback , et al. 1987. History of the Feed Material Production Center Radiological Discharges, FMPC-2082. Oak Ridge, TN: US Department of Energy; 1987.

Calabrese E, Stanek E. 1998. Soil ingestion estimation in children and adults: A dominant influence in site-specific risk assessment. *ELR News and Anal* 28: 10,660–10,671.

[CDC] Centers for Disease Control and Prevention. 1998. Estimation of the impact of the former Feed Materials Production Center (FMPC) on lung cancer mortality in the surrounding community. Atlanta: US Department of Health and Human Services Public Health Service; 1998 Dec.

[CDC] Centers for Disease Control and Prevention. 2000. Screening level estimates of the lifetime risk of developing kidney cancer, female breast cancer, bone cancer, and leukemia resulting from the maximum estimated exposure to radioactive materials released from the Former Feed Materials Production Center (FMPC). Atlanta: US Department of Health and Human Services Public Health Service; 2000 Mar.

Diamond GL. 1989. Biological consequences of exposure to soluble forms of natural uranium. *Rad Protect Dosim* 26: 23–33.

[DOE] US Department of Energy. 1972–1999. Environmental monitoring annual reports for 1972–1995, and integrated site environmental reports for 1997–1999. Fernald Environmental Management Project (and Feed Materials Production Center).

[DOE] US Department of Energy. 1993a. Fernald Environmental Management Project– contamination at the fire training facility removal action work plan and closure plan information and data package. Cincinnati: 1993 Sep.

[DOE] US Department of Energy. 1993b. Fernald Project cleanup report. Cincinnati: 1993 Oct.

[DOE] US Department of Energy. 1993c. CERCLA/RCRA background soils study, Fernald Environmental Management Project, Fernald, OH: 1993 Mar.

[DOE] US Department of Energy. 1994. Remedial investigation report for operable unit 5, Fernald Environmental Management Project. FEMP-05RI-5. Cincinnati: 1994 Oct.

[DOE] US Department of Energy. 1995. Final report–contamination at the fire training facility removal action no. 28, Fernald Environmental Management Project. 1995 May.

[DOE] US Department of Energy. 1996. Record of decision for remedial actions at Operable Unit 5, Fernald Environmental Management Project. Cincinnati: 1996.

[DOE] US Department of Energy. 1997. DOE's integrated environmental monitoring plan. 2505-WP-0022, Rev 0. Cincinnati:1997 Aug.

[DOE] US Department of Energy. 1998a. Final–Explanation of significant differences for operable unit 4 silo 3 remedial action at the Fernald Environmental Management Project.
FEMP-OU4-ESD-O final; 1998 Jan.

[DOE] US Department of Energy. 1998b. Draft final–Waste pits remedial action project (WRAP) remedial design package, Vol 1, Fernald Environmental Management Project. FN/MSD; 1998 Jun.

[DOE] US Department of Energy. 1998c. DOE's integrated environmental monitoring plan. 2505-WP-0022, Rev 1 (draft). Cincinnati: 1998 Oct.

[DOE] US Department of Energy. 1999. Chronic beryllium disease prevention program; final rule. 10CFR850 (final), *Fed Reg* Vol 64, No 235: 1999 Dec 8.

[DOE] US Department of Energy. 2000a. Fernald report. US Department of Energy, Fernald Environmental Management Project. Cincinnati: 2000 Jan.

[DOE] US Department of Energy. 2000b. Attachment 3 (1991 Laboratory data for biota) to letter signed by J. Reising, Associate Director, Environmental Management, DOE Ohio Field Office, dated July 13, 2000.

[DOE] US Department of Energy. 2000c. 1999 integrated site environmental report, 1999 environmental summary (appendices A–E), Fernald Environmental Management Project. 51350-RP-0010, final; 2000 May.

[DOE] US Department of Energy. 2001a. Addendum to the CERCLA/RCRA background soil study, FEMP, Fernald, Ohio. 21120-RP-0003, Rev.1, final; 2001 May.

[DOE] US Department of Energy. 2001b. 2000 integrated site environmental report, 2000 environmental summary, Fernald Environmental Management Project. 51350-RP-0015, Rev 0, final; 2001 May.

[DOE] US Department of Energy. 2001c. FEMP Waste Pit Remedial Action Project. 10500-RP-0008, final; 2001 Mar.

[DOE] US Department of Energy. 2001d. Incident report on waste pit dryer stack. 2001 Apr.

[DOE] US Department of Energy. 2002a. 2001 Site Environmental Report, Fernald Environmental Management Project, Fernald, OH. 51350-RP-0019, Rev 0, final; 2002 May.

[DOE] US Department of Energy. 2002b. Fernald Closure Project Integrated Environmental Monitoring Plan, Fernald, OH. 2505-WP-0022, Rev 3A, draft final; 2002 Oct.

[DOE] US Department of Energy. 2003a. Fernald Closure Project weekly report. 2003 Feb 25.

[DOE] US Department of Energy. 2003b. Fernald Environmental Management Project weekly report. 2003 Jan 14.

[DOE] US Department of Energy. 2003c. IEMP mid-year data summary report for 2003, Fernald Closure Project. 51350-RP-0023, final; 2003 Nov.

[DOE] US Department of Energy. 2003d. 2002 site environmental report, Fernald Closure Project. 51350-RP-0022, final; 2003 May.

[DOE] US Department of Energy. 2004. Quarterly report on dryer stack. 2004 Jul.

Doisy EA. 1973. Effects of deficiency in manganese upon plasma levels of clotting proteins and cholesterol in man. Trace element metabolism. In: Animals 2, 2nd ed. Hoekstra WG, Suttie JW, Ganther AE, Mertz W, eds. Baltimore: University Park Press, 668–670.

Domingo JL, Llobert JM, Tomas JM, et al. 1987. Acute toxicity of uranium in rats and mice. *Bull Environ Contam Toxicol* 39:168–174.

[EPA] US Environmental Protection Agency. 1989. Risk assessment guidance for Superfund, Vol. 1, human health evaluation manual (part A). EPA/540/1-89/002. Washington, DC: 1989.

[EPA] US Environmental Protection Agency. 1999. Exposure factors handbook. EPA/600/C-99/001. US Environmental Protection Agency, Office of Research and Development. (CD-ROM).

[EPA] US Environmental Protection Agency. 2000a. US Environmental Protection Agency's National Primary Drinking Water Regulations; radionuclides; final rule. 40 CFR Parts 9, 141, and 142 (FR Vol 65, No 236). 2000 Dec 7.

[EPA] US Environmental Protection Agency. 2000b. SCREEN3 model and executable source code and SCREEN3 user's guide. Available at http://www.epa.gov/ttn/scram.

[EPA] US Environmental Protection Agency. 2001a. EPA's NPL site narrative for Feed Materials Production Center (USDOE). Available at http://www.epa.gov/superfund/sites/npl/nar1251.htm. Last accessed August 6, 2001.

[EPA] US Environmental Protection Agency. 2001b. EPA's Region 5 Superfund Division, NPL fact sheets for Ohio: Feed Materials Production Center (USDOE). Available at http://www.epa.gov/R5Super/npl/ohio/OH6890008976.htm. Last accessed August 6, 2001.

[EPA] US Environmental Protection Agency. 2002. US Environmental Protection Agency's National Ambient Air Quality Standards (NAAQS). Available at http://www.epa.gov/airs/criteria.html. Last updated November 15, 2002.

[EPA] US Environmental Protection Agency. 2003. EPA's Region 5 Superfund Division, NPL fact sheets for Ohio: Feed Materials Production Center (USDOE). Available at http://www.epa.gov/R5Super/npl/ohio/OH6890008976.htm. Last accessed October 20, 2003.

[EPA] US Environmental Protection Agency. 2004. EPA's National Pollutant Discharge Elimination System fact sheets for states' statistics. Available at http://cfpub.epa.gov/npdes/statestats.cfm. Last updated April 13, 2003.

[FCAB] 2002. Fernald Citizens Advisory Board full board meeting minutes for October 12, 2002 meeting.

Fluor Daniel Fernald. 2000. Fluor Daniel Fernald Interoffice Memorandum (Subject: Beryllium at the FEMP) to D. Carr from R. Vogel dated January 18, 2000.

Friedman BJ, Freeland-Graves JH, Bales CW, et al. 1987. Manganese balance and clinical observations in young men fed a manganese-deficient diet. *J Nutr* 117: 133–143.

[GAO] US General Accounting Office. 1994. Nuclear health and safety consensus on acceptable radiation risk to the public is lacking. GAO/RCED-94-190; 1994 Sep.

[GAO] US General Accounting Office. 2000. Radiation standards - scientific basis inconclusive, and EPA and NRC disagreement continues. GAO/RCED-00-152; 2000 Jun.

Garbe P. 1999. Feasibility assessment for a community-based epidemiologic study of lung cancer and radiation exposures near the Former Feed Materials Production Center. Presentation to the Fernald Health Effects Subcommittee. 1999 Sep.

Gilman AP, Villeneuve DC, Secours VE, et al. 1998. Uranyl nitrate: 91-day toxicity studies in the New Zealand white rabbit. *Toxicol Sci* 41: 129–137.

Goldhaber S. 2003. Trace element risk assessment: essentially vs toxicity. *Reg Tox & Pharm* 38: 232-242.

[HCES] Hamilton County Department of Environmental Services. 2004. ATSDR's review of air permits issued to FMPC and FEMP from 1985 through 2003 by Southwestern Ohio Air Pollution Control Agency. 2004 Jan.

Hodge HC. 1953. Resume of chapters 18 to 28. In: Voegtlin C and Hodge HC, eds. Pharmacology and toxicology of uranium compounds. National Nuclear Energy Series, Division VI, Vol 1, Part IV. New York: McGraw-Hill.

[ICRP] International Commission on Radiological Protection. 1979. Limits for intake of radionuclides by workers. ICRP Publication 30, Part 1. Oxford: Pergamon Press.

[ICRP] International Commission on Radiological Protection. 1991. 1990 Recommendations of the International Commission on Radiological Protection. ICRP Publication 60. Oxford: Pergamon Press.

[ICRP] International Commission on Radiological Protection. 1993. Age-dependent doses to members of the public from intake of radionuclides: Part 2, ingestion dose coefficients. ICRP Publication 67. Oxford: Pergamon Press.

[ICRP] International Commission on Radiological Protection. 1994a. Protection against radon-222 at home and at work. ICRP Publication 65, Annals of ICRP 23(2). Oxford: Pergamon Press.

[ICRP] International Commission on Radiological Protection. 1994b. Human respiratory tract model for radiological protection. ICRP Publication 66, Annals of ICRP 24(1–3). Oxford: Elsevier Science Ltd.

[ICRP] International Commission on Radiological Protection. 1995a. Age-dependent doses to members of the public from intake of radionuclides: Part 3, ingestion dose coefficients. ICRP Publication 69. Oxford: Pergamon Press.

[ICRP] International Commission on Radiological Protection. 1995b. Age-dependent doses to members of the public from intake of radionuclides: Part 4, inhalation dose coefficients. ICRP Publication 71. Oxford: Pergamon Press.

[ICRP] International Commission on Radiological Protection. 1996. Age-dependent doses to members of the public from intake of radionuclides: Part 5, compilation of ingestion and inhalation dose coefficients. ICRP Publication 72. Oxford: Elsevier Science Ltd.

Iregren A. 1990. Psychological test performance in foundry workers exposed to low levels of manganese. *Neurotoxicol Teratol* 12: 673–675.

[IRIS] Integrated risk information system. 1998. US Environmental Protection Agency. Online.

[IT] International Technology Corporation. 1986. Final interim report—air, soil, water, and health risk assessment in the vicinity of the FMPC. Fernald, OH: 1986 Nov.

Kammer JW. 1998. Effects of uranium contaminated drinking water on urinary β-2-microglobulin concentration. A thesis submitted to the Division of Research and Advanced Studies of the University of Cincinnati in partial fulfillment of the requirements for the Master of Science.

Killough GG, Case MJ, Meyer KR, et al. 1998a. Fernald Dosimetry Reconstruction Project, Task 4—environmental pathways—models and validation. Radiological Assessments Corporation Report No. CDC-3. Neeses, SC: 1998.

Killough GG, Case MJ, Meyer KR, et al. 1998b. Fernald Dosimetry Reconstruction Project, Task 6—radiation doses and risk to residents from FMPC Operations from 1951–1988. Vol I and II (final report). Radiological Assessments Corporation Report No. CDC-1. Neeses, SC: 1998 Sep.

Klaassen CD, et al, eds. 1996. Casarett and Doull's toxicology: the basic science of poisons. New York: McGraw-Hill, Health Professions Division; 1996.

Leggett RW. 1989. The behavior and chemical toxicity of U in the kidney: a reassessment. *Health Phys* 57(3): 365–383.

Leggett RW. 1994. Basis for ICRP's age-specific biokinetic model for uranium. *Health Phys* 67: 589–601.

Leifer RZ, Jacob EM, Marschke SF, Pranitis DM. 2002. Atmospheric Aerosol Measurements of ^{238}U and ^{232}Th at Fernald, Ohio, and Implications on Inhalation Dose Calculations. *Health Phys* 83(6): 892–900.

Maynard EA, Hodge HC. 1949. Studies of the toxicity of various uranium compounds when fed to experimental animals. In: Voegtlin IC, Hodge HC, eds. Pharmacology and toxicology of uranium compounds. National Nuclear Energy Series (VI). New York: McGraw-Hill; 309–376.

McDonald-Taylor CK, Singh A, Gilman A. 1997. Uranyl nitrate-induced tubule alterations in rabbits: a quantitative analysis. *Toxicol Pathol* 25(4): 381–389.

Mergler D, Huel G, Bowler R, et al. 1994. Nervous system dysfunction among workers with log-term exposure to manganese. *Environ Res* 64(2): 151–180.

Morris SC, Meinhold AF. 1995. Probabilistic risk assessment of nephrotoxic effect of uranium in drinking water. *Health Phys* 69: 897–908.

[NCRP] National Council on Radiation Protection and Measurements. 1984. Evaluation of occupational and environmental exposures to radon and radon daughters in the United States, NCRP Report No 78. Bethesda: 1984 May.

[NCRP] National Council on Radiation Protection and Measurements. 1999. Recommended screening limits for contaminated surface soil and review of factors relevant to site specific studies, Report No. 129. Bethesda: 1999 Jan.

[NEPI] National Environmental Policy Institute. 1998. Bioavailability: Implication for Science/Cleanup Policy (PhaseI); 1998 Dec.

[NRC] National Research Council. 1989. Recommended daily allowances, 10th rev. Washington, DC: National Academy of Sciences, Food and Nutrition Board; 1989, 1994.

[ODH] Ohio Department of Health. 1988. Ohio Department of Health study of radioactivity in drinking water and other environmental media in the vicinity of the US Department of Energy's Feed Materials Production Center and Portsmouth Gaseous Diffusion Plant. 1988 Dec.

[OEPA] Ohio Environmental Protection Agency. 2001. 2001 annual report to the public on the Fernald Environmental Management Project. (no date)

Ortega A, Domingo JL, Llobert JM, et al. 1989. Evaluation of the oral toxicity of uranium in a 4-week drinking water study in rats. *Bull Environ Contam Tox* 42: 935–941.

Pavlakis N, Pollock NA, McLean G, et al. 1996. Deliberate overdose of uranium: toxicity and treatment. *Nephron* 72(2): 313–317.

Pinney S. 1999a. Fernald medical monitoring program update. Presentation at the Fernald Health Effects Subcommittee meeting, March 3, 1999.

Pinney S. 1999b. Cancer incidence in participants of the Fernald medical monitoring program. Presentation at the Fernald Health Effects Subcommittee meeting, June 23, 1999.

Pinney S. 2000. Personal communication with Brenda K. Weis, Agency for Toxic Substances and Disease Registry (ATSDR). 2000 Feb.

Roels HA, Lauwerys R, Buchet JP, et al. 1987. Epidemiological survey among workers exposed to manganese: effects on lung, central nervous system, and some biological indices. *Am J Ind Med* 11: 307–327.

Roels HA, Ghyselen P, Buchet JP, et al. 1992. Assessment of the permissible exposure level to manganese in workers exposed to manganese oxide dust. *Br J Ind Med* 49(1): 25–34.

Rothstein A. 1949. Uranium dioxide. In: Voegtlin C, Hodge HC, eds. Pharmacology and toxicology of uranium compounds. National Nuclear Energy Series: Manhattan Project Technical Section, Division VI, Vol. 1. New York: McGraw-Hill.

[SED] Site-wide environmental database. 1998. US Department of Energy, Fernald Environmental Management Project. Electronic transfer to the Agency for Toxic Substances and Disease Registry (ATSDR). (Updated 2004)

Shleien B, Rope SK, Case MJ, et al. 1995. Fernald Dosimetry Reconstruction Project, Task 5—review of historic data and assessments for the FMPC. Radiological Assessments Corporation (RAC) Report No. CDC-4. Neeses, SC: 1995.

Spoor NL, Hursh JB. 1973. Protection criteria. In: Hodge HC, Stannard JN, Hursh JB, eds. Handbook of experimental pharmacology. Berlin: Springer-Verlag; 241–270.

Stokinger HE, Baxter RC, Dygent HP, et al. 1953. In: Voegtlin C, Hodge HC, eds. Toxicity following inhalation for 1 and 2 years. National Nuclear Energy Series: Manhattan Project Technical Section, Div VI, Vol 1. New York: McGraw-Hill.

Tannenbaum A, Silverstone H. 1951. Studies on acquired tolerance to uranium. In: Tannenbaum A, ed. Toxicology of uranium. National Nuclear Energy Series, Div IV, Vol 23, New York: McGraw-Hill.

Tseng WP, Chu HM, How SW, et al. 1968. Prevalence of skin cancer in an endemic area of chronic arsenicism in Taiwan. *J Natl Cancer Inst* 40: 453–463.

Tseng WP. 1977. Effects and dose-response relationship of skin cancer and blackfoot disease with arsenic. *Environ Health Perspect* 19:109–119.

[USGS] US Geological Survey. 1984. Occurrence of Uranium in Ground Water in the Vicinity of the US Department of Energy Feed Materials Production Center, Fernald, Ohio. Columbus, OH: US Department of the Interior. 1984.

Voilleque PG, Meyer KR, Schmidt DW, et al. 1995. Fernald Dosimetry Reconstruction Project, Tasks 2 and 3—radiologic source terms and uncertainties. Radiological Assessments Corporation (RAC) Report No. CDC-5. Neeses, SC: 1995 Jun.

Weis BK, Susten AS. 1999. Groundwater contamination by PCE and TCE: ATSDR's approach to evaluate public health hazard. Presented at the Annual Meeting of the American Society for Civil Engineers. Washington, DC: 1999 Jul.

Zhao LS, Zhao SF. 1990. Nephrotoxic limit and annual limit on intake for natural U. *Health Phys* 58: 619–623.