# FINAL REPORT

## Task 6: Radiation Doses and Risk to Residents from FMPC Operations from 1951–1988 Volume I

## The Fernald Dosimetry Reconstruction Project Centers for Disease Control and Prevention Department of Health and Human Services

**January 1998**

*Submitted to the Centers for Disease Control and Prevention*
*in Partial Fulfillment of Contract No. 200-90-08037*


Radiological Assessments Corporation
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**Contributing Authors**

**George G. Killough, Hendecagon Corporation**
**Marilyn J. Case, Eagle Rock Scientific, Inc.**
**Kathleen R. Meyer, Ph.D., Keystone Scientific, Inc.**
**Robert E. Moore, Ph.D., Moore Technical Associates, Inc.**
**Susan K. Rope, Environmental Perspectives, Inc.**
**Duane W. Schmidt, Health Physics Applications**
**Bernard Shleien, Ph.D., Scinta, Inc.**
**Warren K. Sinclair, Ph.D., Warren K. Sinclair, Ph.D, Inc.**
**Paul G. Voillequé, MJP Risk Assessment, Inc.**
**John E. Till, Ph.D., *Radiological Assessments Corporation***

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**TASK 6: RADIATION DOSES AND RISK TO RESIDENTS FROM FMPC OPERATIONS FROM 1951–1988**

**TASK 6**
**RADIATION DOSES TO RESIDENTS FROM FMPC OPERATIONS FROM 1951–1988**

## GLOSSARY OF TERMS AND ACRONYMS

*Each term or acronym is in italics the first time it appears in the text.*

**Absorbed dose** is the amount of energy imparted to tissue in the body by ionizing radiation. The unit of absorbed dose is the rad (traditional system) or gray (SI system); 1 gray (Gy) = 100 rad.

**Activation products** are radionuclides that result from the absorption of neutrons by uranium and other materials present in a nuclear reactor. An example is plutonium-239 produced following neutron absorption by uranium-238 and subsequent decays of uranium-239 to neptunium-239 and then to plutonium-239.

**AD** – the aerodynamic diameter, which is the physical diameter of a particle of unit density that has the same gravitational settling velocity as the particle of interest.

**AMAD** – activity median aerodynamic diameter, a measure of particle size.

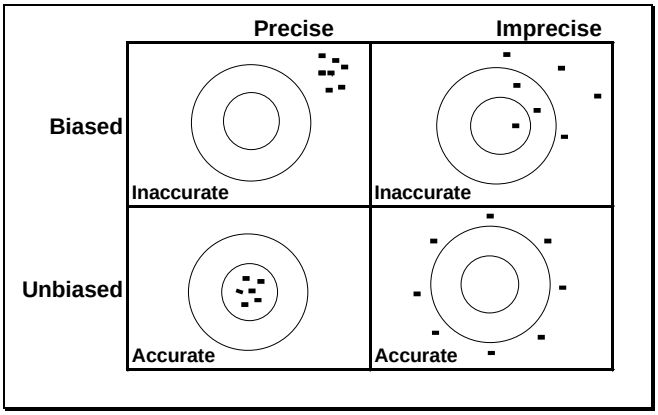
**AMS** – air monitoring stations

**Anisokinetic sampling** refers to a mismatch between the air or fluid velocity in the sampling probe and that in the stack releasing airborne effluents. It is a source of bias in effluent sampling. In contrast, *isokinetic sampling,* in which the two velocities are equal, results in an unbiased sample of the stack effluent.

**Assessment domain** is the region surrounding a facility for which radiation doses to people are calculated; for this project, a circular assessment domain with a radius of 6.2 miles (10 kilometers) centered on the FMPC production area.

**Background radiation** – is the amount of ionizing radiation to which a person is exposed from natural sources, such as terrestrial radiation due to natural sources, such as radiation from naturally occurring radionuclides in the soil or cosmic radiation originating in outer space.

**Background radioactivity**– refers to radioactive elements in the natural environment including those in the crust of the earth (like radioactive potassium, uranium and thorium isotopes) and those produced by cosmic rays.




**BEIR** – The (U.S.) National Research Council's committee on Biological Effects of Ionizing Radiations. The BEIR committee have prepared

reports to advise the U.S. government on the health consequences of radiation exposures.

**Bias** is a systematic distortion of measurements that makes the results inaccurate. Accuracy is a measure of how close a value is to the true number, or a measure of the correctness of a measurement. Precision is a measure of the exactness of a measurement.

**Building wake effects** refer to disturbed airflow that occurs when the points of contaminant release extend only a few feet or meters above the roof of the building. The building affects air dispersion when part or all of the contaminant plume is drawn down into the turbulent region near the ground in the building's wake. Calculations indicate that releases from all rooftop stacks at the FMPC would be affected by wake effects of the buildings.

**Calibration** is the use of environmental data collected under known conditions to represent model parameters outside the limits of the special conditions (for example, at other times and in other locations). In this study, we calibrated an air dispersion model by fitting the model to data representing 1986–1988 and applied the model to earlier time periods with different conditions.

**CDC** – Centers for Disease Control and Prevention, which funded the Fernald Dosimetry Reconstruction Project, is part of the U.S. Department of Health and Human Services.

**Chemical symbols** are abbreviations for different elements and compounds. Examples of elements include U for uranium, O for oxygen, N for nitrogen and F for fluorine. Examples of compounds include $UF_4$ for uranium tetrafluoride (green salt) and $UO_3$ for uranium trioxide (orange oxide).

**Computer code** is a set of alpha numeric instructions that tells a computer to so something. A computer program consists of code. When a reference is made to software consisting of 60,000 lines of code, it reefers to the commands contained in the computer programs used to estimate radiation doses.

**Contamination** refers to unwanted radioactive material or to the deposition of radioactive material in the environment or in any place where it may make surfaces or equipment unsuitable for some specific use.

**Correlation coefficient** is a statistic that describes the degree of association between two sets of data. In this study, the correlation coefficient is used to compare predicted and observed concentrations of a material at various times or locations in the environment. The correlation coefficient can range from –1 to +1. Relatively large values for the correlation coefficient are obtained when the data are positively correlated, that is, large values of the first data set (e.g. predictions) are associated with large values of the second data set (for example, observed measurements). Large negative correlation coefficients are obtained when large values of one data set are associated with small values of another data set. If the sets of data are unrelated, the correlation coefficient would be near zero (see *validation, predicted-to-observed ratio, model bias*).

**Decay (daughter) products** refer to the isotopes or radionuclides that result from radioactive decay of isotopes, such as the uranium and thorium isotopes. In most of the feed materials received by the Feed Materials Production Center (FMPC), the uranium had previously been separated chemically from the other decay products. As a result, the facility's effluents consisted primarily of uranium, and decay product radionuclides were generally present in

small quantities. In naturally-occurring uranium ores, the decay products include isotopes of uranium, protactinium, thorium, radium, radon, and radon daughter products. *Radon daughter products* that are derived from uranium are the short-lived decay products from radon-222, which include polonium-218, lead-214, bismuth-214 and polonium-214.

**Denitration** is a chemical process that took place in Plant 2/3. This process drove off nitrates by heating uranyl nitrate hexahydrate to produce uranium trioxide (or orange oxide).

**Depleted uranium** – one of three categories of uranium based on the abundance of uranium-235 relative to uranium-238. It can be compared to natural and enriched uranium. At the FMPC, depleted uranium typically contained 0.14–0.20% uranium-235. Natural uranium contains 0.72% uranium-235, while enriched uranium contains significantly more uranium-235 relative to uranium-238.

**Deposition velocity** refers to the proportion of the rate of deposition of radioactive material on the ground to the concentration of the material in the air at a specified reference height. The dry deposition velocity is a function of the particle size and density of the radioactive material (the larger the particle size, the greater the deposition velocity); the nature of the surface (for example, snow-covered, lawn, tree-covered); and meteorological variables (for example, the higher the wind speed, the higher the deposition velocity).

**Derbies** are masses of uranium metal fabricated in Plant 5. The derbies were then remelted and cast into ingots of metallic uranium.

**Deterministic effects** include health effects that most often occur relatively soon after exposure (days to weeks) such as visible erythema (reddening) on the skin, cataracts, epilation (hair loss), sterility, and bone marrow depression. These deterministic effects have threshold doses for acute exposure that must be exceeded for the effect to occur.

**Diffusion** (related to radon releases) results from the random motions of molecules. For radon in the K-65 silo head spaces, this causes movement from an area of high concentration (the silo head space) to an area of lower concentration (outside the silos). Radon (like other gases) diffuses through small cracks or even solid material like the concrete silo domes (the material must be somewhat porous).

**Direct exposure** refers to one pathway of exposure of people to radiation from the FMPC. In this exposure pathway, penetrating radiation emitted from radioactive material is partially absorbed by individuals exposed to it. The amount of exposure decreases with distance from the source. An example is gamma radiation from the K-65 silos that resulted in low-level exposure of nearby residents.

**DOE** – U.S. Department of Energy

**Dose** is a general term denoting the quantity of radiation or energy that is absorbed by the body. There are technical terms with specific definitions, such as absorbed dose, equivalent dose, and effective dose.

**Dose** Reconstruction is a scientific study that estimates doses to people from releases of radioactivity or other contaminants into the environment from a facility.

**DRF** – dose-rate factor

**Dust collector** is one type of filtration system for airborne effluents used at the FMPC to remove airborne particulate material before it was discharged through the stack to the outside. The filtering medium is similar to that used for large fiber vacuum cleaner bags.

**Effective dose** provides a measure of the dose to the whole body, taking into account the dose absorbed by each of the target organs and the sensitivity of those organs to radiation. The unit of effective dose is the rem (traditional system) or seivert (SI system); 1 seivert (Sv) = 100 rem.

**Effluent** is a gas or liquid containing contaminants that flows from a process, building, or site into the surrounding environment.

**Empirical models** are based primarily on measured data, rather than on theory or calculated values.

**Enriched uranium** – one of three categories of uranium based on the abundance of uranium-235 relative to uranium-238. It can be compared to natural and depleted uranium. At the FMPC, enriched uranium typically contained 0.95–1.25% uranium-235. Natural uranium contains 0.72% uranium-235, while depleted uranium contains less than 0.72% uranium-235. While most of the enriched uranium at the FMPC was in the above range, some processing of 2% enriched uranium occurred in the 1960s. The capability to digest 5% enriched uranium was added to Plant 1 in 1970.

**Enrichment of uranium** is a process by which the relative abundances of the isotopes of uranium are altered, thereby, producing a form of the element that has been enriched in one particular isotope and depleted in its other isotope. For example, natural or normal uranium contains 0.72% uranium-235. Enriched uranium contains more than the natural concentration of uranium-235.

**Entrainment** is a process in which the uranium-containing liquid droplets in a scrubber are carried by the exhaust air stream and are vented to the atmosphere with the exhaust gases.

**Environmental exposure** is exposure to radiation through environmental pathways.

**EPA** – U.S. Environmental Protection Agency

**Epidemiology** is the study of the incidence of specific diseases in human populations.

**Episodic releases** are actual accidental releases that were large enough to require special treatment during environmental transport and dose assessment.

**Equivalent dose** – The probability of a health effect for 1 gray of absorbed dose to an organ depends on the type of radiation that is absorbed (for example, alpha, beta, or gamma radiation). The equivalent dose for a particular type of radiation is the absorbed dose multiplied by a quality factor that adjusts approximately for the probability of health effects. The equivalent dose for any combination of different types of radiation is the sum of the equivalent doses for the respective types. The unit of effective dose is the rem (traditional system) or sievert (SI system); 1 sievert (Sv) = 100 rem.

**Exposure pathways** are ways in which people are exposed to contaminants in the environment. The key exposure pathways are air and water, with most exposures occurring by inhalation, drinking water, eating crops and other foods, and from direct irradiation.

**Fission products** are radionuclides that result from the splitting of heavy elements like uranium in a nuclear reactor. Examples are strontium-90, technetium-99, ruthenium-106 and cesium-137.

**FDRP** – Fernald Dosimetry Reconstruction Project

**FEMP** – Fernald Environmental Management Project, the new name of the FMPC beginning in 1991.

**FMPC** – Feed Materials Production Center, the name of the site until 1991.

**Geometric bias** is a type of model bias used in this study when concentrations of a material at multiple locations or times are being examined. Geometric bias is the geometric mean of the individual P/O ratios, given by the formula: Geometric bias = $\exp\{ \Sigma [ \ln (P_i / O_i) ] / n\}$, where $P_i$ is the predicted concentration at location or time i, $O_i$ is the observed concentration at location or time i, and n is the number of locations or times being compared (see *model bias, predicted-to-observed ratio, validation*).

**GM** – Geometric Mean, a measure of the central point of a distribution; typically used to describe skewed distributions (e.g. lognormal distributions).

**GSD** – Geometric Standard Deviation, a measure of the spread of a distribution. A large GSD indicates a wide range of measured or calculated values.

**Gaussian plume model** is a well-known air transport model that assumes that released materials are moved in a straight line from the source, depending upon the wind speed and direction at the time of the release. The air transport calculations for this project are based upon this model.

**Grab samples**, usually of relatively small volume, are taken at random or at preselected frequencies. These samples define the concentration of a contaminant at the specific time when they are collected and differ from continuous or proportional samples that reflect the time averaged value.

**Gray (Gy)** is the unit of absorbed dose in the newer SI system of units. It is equivalent to 100 rad. See absorbed dose.

**Great Miami River** is the major water flow near the FMPC that receives most of the FMPC liquid effluents. The river, located about 1 mile (mi) [1.7 kilometer ( km)] east and south of the FMPC, runs in a southerly direction and enters the Ohio River approximately 18 mi (29 km) downstream of Cincinnati. Upstream of the FMPC on the Great Miami River lie the communities of Fairfield, Hamilton, Middletown, and Dayton. The flow of the river at the Hamilton gauge averages 3300 cubic feet per second ($ft^3\ s^{-1}$) [93.4 cubic meters per second ($m^3\ s^{-1}$)] with a maximum of 352,000 $ft^3\ s^{-1}$ (9970 $m^3\ s^{-1}$) measured in March 1913 and a minimum of 100 $ft^3\ s^{-1}$ (2.8 $m^3\ s^{-1}$) measured in September 1941.

**Green salt** is the common name for uranium tetrafluoride ($UF_4$), the product from the Plant 4 operations that was sent to Plant 5 for conversion to uranium metal derbies.

**Gulping** refers to a process in Plant 2/3 in which orange oxide (uranium trioxide, $UO_3$) from the denitration pots was transferred by a vacuum hose to a storage hopper. It appeared that the hose was gulping the orange oxide.

**Health impact** – the likelihood of deleterious health effects occurring as the result of exposure.

**IH&R** – Industrial Hygiene and Radiation Department at the FMPC

**ICRP** – International Commission on Radiological Protection

**Ionizing radiation** is a type of radiation that has enough energy to create ions (ionized atoms that are chemically active) inside living cells. These ions can damage key substances in cells, including the DNA within the cell nucleus. Such damage can lead to cancer or other defects.

**IT** – International Technology Corporation

**K-65 silos** are large concrete tank-like structures that store residues from the extraction of uranium from ores that were processed during the early years of FMPC operations.

**kilo** is a prefix that multiplies a basic unit by 1000. For example, 1 kilogram = 1000 grams.

**LET** – linear energy transfer, the manner in which radiations deposit energy in tissue. High LET radiations deposit energy more densely and low LET radiations deposit energy more sparsely.

**Lognormal distribution** is obtained if the logarithms of a set of values are distributed according to a normal distribution.

**Mass loading** is defined as the concentration of dust or particulates in air. This value can be used to determine the concentration of a contaminant in air as a result of resuspension if the concentration of the contaminant in the surface layer of the soil is known.

**Mathematical model** is a collection of mathematical formulas used to characterize a relationship or process. For example, mathematical formulas were used in the Fernald Dose Reconstruction Project to model how uranium and other radionuclides released from Fernald traveled through the environment and were taken up by people.

**Median** is the central point of a distribution. Half of the values are larger than the median value and half are smaller.

**MMAD** – mass median aerodynamic diameter, a measure of the diameter of a sphere of unit density that has the same gravitational settling velocity as the particle. Particle size is reported as MMAD.

**Model bias** is a measure of agreement between predicted and observed concentrations of a material in the environment. In this study, the predicted-to-observed ratio (P/O ratio) is used as a measure of model bias. A P/O ratio of 1 indicates perfect agreement (no bias). A type of model bias, the geometric bias, is used when a number of times or locations are being compared (see *predicted-to-observed ratio, geometric bias, correlation coefficient, validation*).

**Monte Carlo procedure** is a method that uses computer-generated pseudo-random numbers to make calculations with statistical distributions. In this study, Monte Carlo methods

have been used to estimate statistical distributions that represent uncertainties in estimated quantities, such as radiation dose. This approach contrasts with a deterministic approach in which a calculation is based upon point estimates of the various parameters and yields a single result. The Monte Carlo calculation carries the underlying uncertainty in the parameters forward and displays it in the magnitude of the distribution of results. A statistical risk management computer program, called Crystal Ball™ (Decisioneering 1993) was used in this study for some of the uncertainty analysis.

**MTU** – metric ton of uranium; 1 MTU equals 1000 kilograms or 2200 pounds of uranium.

**Natural uranium** is one of three categories of uranium based on the abundance of uranium-235 relative to uranium-238. It can be compared with enriched and depleted uranium. Natural or "normal" uranium contains 0.72% uranium-235, enriched uranium contains more than the natural concentration of uranium-235, while depleted uranium contains significantly less than 0.72% uranium-235.

**Naturally occurring radionuclides** are radionuclides that are naturally present in the environment and are two general types: primordial and cosmogenic. Most primordial radionuclides are isotopes of the heavy elements of the three radioactive series headed by uranium-238, thorium-232, and uranium-235. Cosmogenic radionuclides are produced by interactions in the atmosphere or in the earth; three of these [tritium (hydrogen-3), carbon-14, and sodium-22] are isotopes of major elements in the body.

**NCRP** – National Council on Radiation Protection and Measurements

**NKES** – Northern Kentucky Environmental Services. In 1985, they measured distributions of particle sizes from both the inlet and outlet ducts of 15 major uranium-emitting stacks at the FMPC.

**NLO** – National Lead Company of Ohio, the contractor for the FMPC through the end of 1985.

**NOAA** –National Oceanic and Atmospheric Administration

**$NO_x$** – nitrogen oxides, such as $NO_2$ and $NO_3$.

**ODH** – Ohio Department of Health

**Orange oxide** is the common name for uranium trioxide ($UO_3$), the product from the Plant 2/3 refinery that was sent to Plant 4 for further processing.

**OSTI** – Office of Scientific and Technical Information, located in Oak Ridge, Tennessee. The national center for worldwide literature on scientific and technical energy-related matters. It provided some of the information that *RAC* used for completion of the project.

**Paddy's Run Creek** is a small intermittent stream lying along the west boundary of the FMPC that joins the Great Miami River approximately 1.9 miles (3 kilometers) south of the site. The flow in Paddy's Run, which generally exists only during January to May, averaged 2 to 4 cubic feet per second (0.065 to 0.13 cubic meters per second). Because flow in Paddy's Run is dependent on rainfall, discharges from the site to Paddy's Run generally occurred dur-

ing periods of heavy rain and runoff when the storm sewer outfall overflowed, or when runoff from the west side of the site flowed into Paddy's Run.

**Percentiles** are defined in such a way that a large set of data, arranged from its smallest to its largest value, is divided by its percentiles into 100 classes containing nearly equal numbers of data. The exact rules for defining the percentile numbers are complicated, but the effect is that approximately 5% of the data are less than or equal to the 5th percentile, and approximately 95% of the data are greater than or equal to the 5th percentile (similar statements hold for the other percentiles). The median is defined as the 50th percentile, which divides the data (approximately) into halves (if there are an odd number of data, the middle value is the median; if there are an even number, the average of the two middle values is the median). In this document, uncertainty distributions are indicated by their 5th, 50th, and 95th percentiles. Observations above the 95th percentile have only a 5% probability of occurrence, as do observations below the 5th percentile. The 50th percentile is presented as the best estimate.

**pico** is a prefix that multiplies a basic unit by 1/1,000,000,000,000 or 1 x $10^{-12}$. For example, 1 picocurie equals 1 x $10^{-12}$ curie.

**Plume** is the concentration profile of an airborne or waterborne release of materials as it spreads from its source. A plume from a coal-fired power plant, for example, may be visible for some distance from its stack, with the concentration of its components decreasing with distance from the stack and from the centerline of the plume. After the plume becomes invisible because of dilution, it continues to be diluted with increasing time and distance. Atmospheric dispersion models of this process predict concentrations within a plume far downwind and far beyond the point at which a plume becomes invisible. Similar modeling for releases from nuclear facilities can estimate the impacts of releases that occurred in the past.

**Plume depletion** refers to the processes that reduce the amount of material in a plume of airborne effluent. As material in a plume falls to or, is deposited on the ground, the airborne concentration of plume gets smaller (or is depleted) as it travels downwind from the source.

**Plume rise** is related to the height of the plume coming from a stack or roof vent. The total effective release height of the plume is the sum of the physical height of the stack or roof vent and an increment that depends on other factors, such as air flow and meteorological conditions. The increment is called the plume rise. Plume rise was considered in the calculations of releases from the old solid waste incinerator.

**Predicted-to-observed ratio, or P/O Ratio**, is a measure of the agreement between predicted concentrations (P) of a material in the environment, and measured or observed concentrations (O). A P/O ratio greater than 1 indicates the model is overpredicting compared to the observed concentration of the material, whereas a P/O ratio of less than 1 indicates underprediction. A P/O ratio of 1 indicates perfect agreement (see *validation, model bias*).

***RAC*** – Radiological Assessments Corporation is the organization contracted by CDC to do the Fernald Dosimetry Reconstruction Project.

**Radionuclide** is a radioactive element, for example uranium-238 or radon-222.

**Radium** is a naturally occurring, radioactive metallic substance that occurs most commonly as an isotope with an atomic weight of 226 (radium-226). It occurs in minute quantities asso-

ciated with uranium in natural ores. Radium-222, is a naturally occurring decay product of uranium.

**Radon** is a radioactive, nonreactive gas. There are three isotopes of radon that occur in nature as members of the actinium, thorium and uranium series. Most human exposure to radon is from uranium naturally present in soil and rock. The gas is created and leaves the soil as the uranium-238 decays through several *decay products* to radium-226, then on to radon-222 gas. Radon and its own decay products (radioactive particles created as radon decays) may then be inhaled by humans. At Fernald, radon-222 has been released from the onsite storage of K-65 material, a waste from the processing of uranium ore. This material, stored in two large silos onsite, contains high concentrations of radium-226 and thus, acts as a continuous source of radon-222.

**Receptor location** is a geographic location of individuals within the assessment domain where concentrations are calculated by a model.

**Recycled uranium** is uranium that had been irradiated in nuclear reactors, where finished uranium products were used. As a result, when the uranium was recovered and returned to the FMPC, small amounts of fission and activation products were introduced into the process stream.

**Reentrainment** is a process whereby the exhaust airflow creates new droplets from liquid that had been previously collected by a screen type filter.

**Resuspension** refers to the reentry into the air of particles that were previously deposited onto the soil.

**Risk** – the probability of a deleterious health effect such as cancer, being induced by radiation.

**Scrubber** is a type of treatment system for airborne effluents that uses liquid droplets to remove particulate matter and reactive gases from airborne waste streams before they were discharged through the stack to the outside. At the FMPC, scrubbers were used in Plant 2/3 (refinery) and in Plant 8 (scrap recovery).

**Scrub liquor** is the liquid in a scrubber that cleans or scrubs the exhaust air from certain plant operations. The liquid removes reactive gases and particles in the air stream before the air stream is discharged to the atmosphere.

**South Plume** refers to the groundwater that has been contaminated by uranium from the FMPC. It extends southward from a point south of the waste pits and reflects the movement of contaminated groundwater.

**Source term** refers to the quantity, chemical and physical form, and the time history of contaminants released to the environment from a facility.

**SSOD** – Storm Sewer Outfall Ditch, the drainage ditch that runs south from the FMPC production area near the storm sewer lift station to Paddy's Run.

**Stochastic effects** include health effects that occur randomly at all radiation dose levels, including the lowest doses, with a frequency dependent on the dose, and usually at long intervals after exposure.

**TBE**– tracheobronchial epithelium, a layer of cells that lines the conducting portion of the airway.

**Thorium** is a naturally occurring, radioactive metallic substance that occurs most abundantly as an isotope with an atomic weight of 232 (thorium-232). Thorium is used as a source of nuclear energy, in sun-lamp and vacuum-tube filament coatings, and in alloys.

**TLD** – thermoluminescent dosimeter, which is a device that measures the amount of radiation from cosmic rays, naturally occurring radioactivity in the soil and from the K-65 silos.

**Tracheobronchial epithelium** (TBE) is a layer of cells that lines the conducting portion of the airway.

**TRU** – transuranic nuclides refer to isotopes heavier than uranium that are created by neutron capture by heavy elements.

**UAP** – uranium ammonium phosphate

**$UF_4$** – uranium tetrafluoride, or green salt, was the product from Plant 4 that was sent on to Plant 5 for conversion to derbies.

**UNH** – uranyl nitrate hexahydrate was an intermediate step in the denitration process in Plant 2/3; nitrates were removed from UNH to produce uranium trioxide ($UO_3$, or orange oxide).

**$UO_2$** – uranium dioxide

**$UO_3$** – uranium trioxide, often called orange oxide, was produced in the Plant 2/3 refinery and was sent to Plant 4 for further processing.

**$U_3O_8$** – uranium oxide, the most common oxide of uranium found in typical ores. $U_3O_8$ is extracted from the ore during the milling process. The ore typically contains only 0.1% $U_3O_8$. The yellow-cake, the product of the milling process, contains about 80% $U_3O_8$

**$UO_2(NO_3)_2$** – uranyl nitrate was a product of the digestion phase in the Plant 2/3 refinery.

**Uncertainty** is the term used to describe the level of confidence in a given estimate based on the amount and quality of the evidence (data) available. Uncertainties in the results of this study arise primarily from uncertainties in existing measurements, absence of some kinds of measurements because of less extensive environmental monitoring in earlier periods, lack of knowledge about some physical processes and operational procedures, and the approximate nature of mathematical models used to predict the transport of released materials.

**Uncertainty range** is defined in this project as the range of releases, doses or risks that fall between the 5th and 95th percentile, thus, containing 90% of all possible values.

**Uranium** is a naturally occurring, radioactive metallic substance which, in natural ores, has an atomic weight of approximately 238. The two principle natural isotopes are uranium-235 (0.7%) and uranium-238 (99.3% of natural uranium). Natural uranium also includes a minute amount of uranium-234. Uranium is used chiefly in nuclear bombs and as a fuel in power reactors.

**UNSCEAR -** United Nations Scientific Committee on the Effects of Atomic Radiation

**USGS** – U. S. Geological Survey

**Validation** is the comparison of predicted concentrations of a material in the environment, based on source term reconstruction and environmental transport models, with historical measured concentrations (see *predicted-to-observed ratio, model bias, correlation coefficient*).

**Wake cavity** is a volume of turbulent air formed by airflow around and over a building.

**Waste disposal pits** are excavations in the ground that were used to store solid and liquid wastes from the FMPC. There were six pits in all located west of the production area: three were used for the disposal of dry solid wastes only and three were used to hold liquid wastes. The pits ranged in depth from 13 feet (Pit 2, dry) to about 30 feet (Pit 5, wet), and in volume from 9,000 $yd^3$ (Pit 6, dry) to 227,000 $yd^3$ (Pit 3, wet).

**WMCO** – Westinghouse Materials Company of Ohio, the FMPC site contractor from 1986 through 1992.

**Washout ratio** is the ratio of concentrations of the material in precipitation and in air. The atmospheric plume moving from the FMPC is affected by the process of wet deposition, in which rain, snow or sleet removes, or washes out, radioactive material from the air.

## EXECUTIVE SUMMARY

The Feed Materials Production Center (FMPC), a 1000-acre site located about 25 km northwest of Cincinnati, Ohio, processed uranium concentrates and uranium compounds recycled from other stages of nuclear weapons production, as well as some uranium ore and thorium. Particulate releases were primarily uranium (natural, depleted, and slightly enriched). In addition, two large silos, containing radium-bearing residues from uranium extraction, were emission sources of radon and its decay products. Radiological Assessments Corporation (RAC) was contracted by the Centers for Disease Control and Prevention (CDC) to carry out the Fernald Dosimetry Reconstruction Project to evaluate the doses and health impacts on the public from radionuclides released from the FMPC to the environment from 1951 through 1988.

Since the project began in 1990, RAC has estimated the quantities of radioactive materials released to air, surface water and in groundwater (*RAC* 1991, Voillequé et al. 1995), and developed the methodology and mathematical models to describe the environmental transport of the materials, which are needed to calculate the resulting radiation doses and health impacts (Killough et al. 1993; Killough et al. 1996). They did a thorough evaluation of historic environmental monitoring data to verify that the estimates of releases and transport are reasonable (Shleien et al. 1995; Killough et al. 1996). The Task 2/3 Source Term and the Task 4 Methodology reports were reviewed by the National Research Council Committee on an Assessment of CDC Radiation Studies (NAS/NRC 1992, 1994). In addition, the NAS Committee reviewed a draft version of this task 6 report and published their findings (NAS/NRC 1997). Radiological Assessments Corporation (*RAC*) published a detailed response (Killough et al. 1997). A summary of the issues raised by the NAS review and RAC's comments to them is included in this report as an addendum.

Thorough review of historical records and extensive interaction with former and current employees and residents have been the foundation for reconstructing routine operations, documenting accidents, and evaluating unmonitored emission sources. The release estimates reported here are median values of releases with associated uncertainties (5th and 95th percentile values) that were calculated as an integral part of the estimates. Uncertainties for each source vary widely depending upon the type of information found to support the methods to reconstruct the releases. The largest releases of uranium to air and water occurred in the 1950s and 1960s. Radon releases from the K-65 storage silos remained elevated through most of the 1970s. The predicted total quantities of radon and radon decay products released through 1988 are 170,000 Ci (5th to 95th percentile range of 110,000 to 230,000 Ci), and 130,000 Ci (5th to 95th percentile range of 87,000 to 190,000 Ci), respectively.

The quantity of uranium released to surface water was much less than that released to air. Total amounts released to the atmosphere (dust collectors, Plant 2/3 scrubbers, Plant 8 scrubbers and miscellaneous sources) were 310,000 kg uranium, with the 5th to 95th percentile range of 270,000 to 360,000 kg. For releases of uranium in liquid effluents, the median release estimate to the Great Miami River during this time period is 82,000 kg (5th to 95th percentile range of 71,000 to 94,000 kg), while that to Paddy's Run is 17,000 kg, with the 5th to 95th percentile range of 14,000 to 20,000 kg.

The methodology that describes the movement of *radionuclide*s in the environment around the FMPC translates release estimates into concentrations of radioactivity in the environment. These environmental concentrations form the basis of the radiation dose calculations. Screening calculations showed that atmospheric pathways dominate the total dose from FMPC releases. Accordingly, the local meteorology, effluent particle size and chemical form, and wet and dry deposition were particularly important in this study. The airborne pathways model, described in this report, was developed specifically for the project to include:

- Multiple release points within the FMPC production area
- Physical and chemical characteristics of release sources
- Dispersion as a function of distance from the source
- Particle size distributions and dry and wet deposition of particulates on the ground
- Resuspension of material previously deposited on the ground
- Runoff and leaching of deposited material from the soil
- Decay of radon-222 and formation of decay products as the release moves downwind from the release point to a receptor location.

A simple dilution and transport model was chosen as the method for assessing the transport and dispersion of radioactive materials in surface water. This method accounted for dilution and transport of the material in either the Great Miami River or Paddy′s Run Creek.

The environmental monitoring records provided important confirmation that the estimates of environmental releases and transport were reasonable. Although the environmental data are important to consider in developing methods for dose reconstruction, they are not complete enough, either temporally or spatially, to rely on exclusively for assessment of the exposure to surrounding populations from FMPC effluents. Rather, these data were used primarily to provide a quality check of the source term estimates for uranium and radon and to calibrate or validate the transport models.

Exposure scenarios for nine hypothetical residents of the FMPC area accounted for the effect of location, distance, diet and lifestyle on the radiation dose estimates. For some exposure scenarios, individuals spent their entire lives in the area (scenarios 1, 2, 3, 5, 6 and 7), while others left after completing high school (scenarios 4 and 9). These realistic buyt hypothetical scenarios can help individuals determine general dose ranges for themselves by finding a lifestyle profile that most closely matches their background.

The study shows that inhalation is the most important pathway of exposure and radon is the source of most of the dose. The decay products of radon contribute more than 90% except for scenario 4 (75%) of the cumulative lung dose, depending on the scenario. The organs receiving the highest radiation doses, in order, were lung, bone surfaces, red marrow, kidney, and liver. The doses to the testes and the ovaries were less than those to the other organs. The dose to the embryo/fetus from radionuclides taken into the body of a pregnant woman was estimated as the dose to the uterus of the woman during the nine-month gestation period, and is not a committed dose. After birth, the individual would continue to accumulate dose from radioactivity incorporated in tissues from placental transfer during gestation.

For all scenarios, the dose to the lung from releases of radon was much higher (98% of the total) than the dose to other organs from uranium. Residency time and distance from the site are the most important variables in determining the dose received by an offsite resident.

Next in importance are the direction from the site and drinking well water containing uranium. The person in scenario 1 had the highest median dose to the tracheobronchial portion of the lung from radon releases of 3.6 Sv (360 rem), with an uncertainty range of 0.98 to 14 Sv (98 to 1400 rem). For scenario 1, the natural background radiation dose to the tracheobronchial epithelium would have been about 0.95 Sv (95 rem).

The subject of scenario 3, whose source of drinking water was a uranium-contaminated well, had the highest median dose to bone surface of 0.26 Sv (26 rem), with an uncertainty range of 0.13 to 0.33 Sv (13 to 33 rem). (****** vs. 78 rem in draft report***). However, the dose to the bone surface was significantly less than the dose from radon releases to the tracheobronchial epithelium of 2.6 Sv (260 rem), with an uncertainty range of 0.89 to 10 Sv (89 to 1000 rem).

There is a risk of fatal cancer, almost entirely lung cancer, for all scenarios. For the scenarios with the greatest exposure, these risks are by no means negligible. The median lifetime cancer risk identified for the scenario providing the largest dose (scenario 1) is about 1.3%. This would increase the natural risk of fatal cancer in a lifetime (20%) to 21.3%, i.e., by about 7%. This risk for scenario 1 is about the same as the estimate of risk from natural background sources; that is, the exposure in scenario 1 is equivalent to about a doubling of the risk of background radiation in a leftime.

The risk estimates made for the representative individuals in the nine scenarios are subject to a variety of uncertainties. For scenario 1, the median estimate of the risk of fatal cancer from the FMPC releases is 1.3%, but a risk as low as 0.24% (the 5th percentile value) or as high as 9.6% (the 95th percentile value) are possible risks at the levels indicated because of the uncertainties involved. The 95th percentile risk for scenario 1 would increase the natural risk of cancer by almost 50% and is equivalent to the risk of a smoker from tobacco smoke alone.

There is no likelihood of direct tissue damage to the lung or any other organ to individuals in any of the scenarios from the radiation. The chemical toxicity from uranium in the human kidney is low except for scenarios 1, 3, and 6 where mild effects in the kidney are possible and there is a small chance of more severe effects. However, it should be pointed out that severe effects are inferred rather than directly known from experience. An episodic (accidental) release of uranium hexafluoride in 1966 could have produced a maximum concentration in the kidney, which was above the level where more serious kidney effects might be observed. The doses to the gonads are very low and the resulting genetic risk is very small.

***Each acronym and term defined in the glossary of this report appears in italics the first time it is used in the text***

## INTRODUCTION TO THE FERNALD DOSIMETRY RECONSTRUCTION PROJECT

The Fernald Dosimetry Reconstruction Project (*FDRP*) was conducted to estimate radiation *doses* to people who lived near the Fernald (Ohio) Feed Materials Production Center (*FMPC*) during its years of operation from 1951 to 1988. The Fernald area is generally rural and the land is mainly used for farming and raising cattle. The population within a 6.2 mile (mi) [10-kilometer (km)] radius of the center of the site has increased from approximately 10,000 in 1950 to over 23,000 in 1990. The study was conducted by Radiological Assessments Corporation (*RAC*) for the Centers for Disease Control and Prevention (*CDC),* U.S. Department of Health and Human Services. The project has been a comprehensive one with a number of key events that have added depth and breadth to the original work, and have extended the project beyond its original scope. We have

**This project deals with the past, not the present or future. The focus is on exposure of area residents to releases of radioactive materials from the FMPC from 1951 to 1988.**

- reconstructed the *source terms*, or release estimates, instead of validating the site-generated source terms, as originally proposed,
- carefully reviewed and discussed the *radon* source term because our results indicated that radon emissions from the site contributed significantly to doses to the residents,
- worked closely with the CDC in adapting the atmospheric dispersion and radiation dose models to obtain final dose estimates in a form compatible with a possible epidemiological, or health effects, study,
- interacted with the National Research Council's (NRC) Committee on an Assessment of CDC Radiation Studies, who reviewed our source term (Voillequé et al. 1995) and environmental transport and dose methodology reports (Killough et al. 1993).
-

The NRC committee also reviewed a draft of this Task 6 report and published its opinions (NRC 1997). RAC disagreed with important parts of the review document and published a detailed response to disputed findings (Killough et al. 1997). At a subsequent meeting of the NRC committee with RAC and CDC staff, RAC clarified many points in the draft report that the NRC committee had misunderstood. The committee sent a letter to the CDC acknowledging that it had "mistakenly criticized RAC in some parts of its report." The letter, which is reproduced in the Addendum at the end of this volume, requested some changes in the report and further investigation of some questions. The Addendum summarizes the disposition of these requests.

The FMPC is a 1000-acre [405-hectare (ha)] site located about 15 mi (25 km) northwest of Cincinnati, Ohio (Figure 1). Ground was broken on May 16, 1951, construction was completed by 1954, and the production area operated through 1988. The FMPC converted *uranium* feed materials (uranium concentrates, uranium compounds recycled from other stages of nuclear weapons production, and some uranium ores) to uranium metal ingots for machin-

ing or for extrusion into tubular form. Production reactor fuel cores and target elements also were fabricated.

Although uranium processing was the primary activity at the FMPC, lesser amounts of *thorium* were processed intermittently during the mid-1950s and from 1964 through 1980. Some *recycled uranium* feed materials were processed beginning in late 1962. The recycled uranium had been irradiated in nuclear reactors, where finished uranium products were used. When spent fuel from the reactors was processed before being returned to the FMPC, the uranium was not completely separated from other radioactive contaminants called *fission and activation products.* As a result, when the uranium was returned to the FMPC, small amounts of fission and activation products were introduced into the process stream. However, particulate releases to the environment were primarily uranium.

Besides uranium and thorium, significant quantities of *radon* and its *decay products* were also released from the site. Radon is a chemically inert radioactive gas, which produces a decay chain of radioactive isotopes of *polonium*, *bismuth*, and *lead.* These decay products of radon (sometimes called *daughters*) deliver radiation dose to the respiratory tract when they are inhaled. Two large concrete storage silos (the K-65 silos), containing *radium*-contaminated materials, were the principal emission sources of radon and its *decay products*.

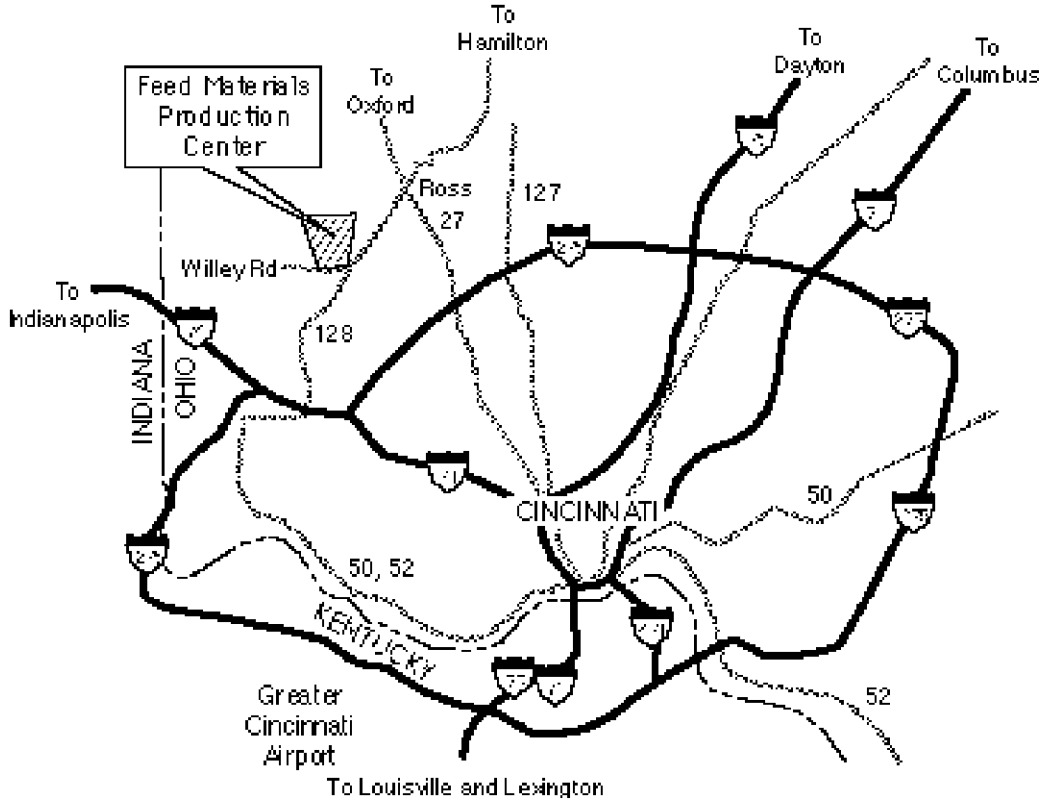



**Figure 1.** Location of the Fernald Feed Materials Production Center.

Exposure of the public to radioactive materials resulted from both planned and unplanned releases to the environment. This study has been mainly concerned with radiation dose resulting from this exposure and health risks associated with the dose. However, uranium is also known to have toxic chemical effects in the body, primarily the kidneys, when the uranium concentration in the body exceeds threshold levels. The study examines this non-radiological exposure and its potential health effects

**The pathways approach begins with learning what kinds and how much radioactive materials were released from a facility and ends with estimating the health impacts these releases had on the residents in the area.**

A pathway of exposure must be known for radiation doses to be estimated (Figure 2). For this project, the pathway analysis begins at the FMPC site by determining the source term, that is, how much and what kinds of radioactive materials were released from which locations onsite. The analysis ends with final doses and risks, accompanied by uncertainties in the estimates. The pathways approach includes determining the way materials were dispersed from the FMPC to the environment; identifying the movement and uptake of material through the food chain; identifying the amount of the material in various environmental media (air, soil, water, vegetation, milk, food); calculating the radiation doses to the people in the area by taking into account their consumption of contaminated food and water and their exposure to contaminated media, and estimating the risks or health impacts of the releases to the residents in the area. On the basis of this pathway approach, the project has been divided into six tasks that are described below and shown in Figure 2.

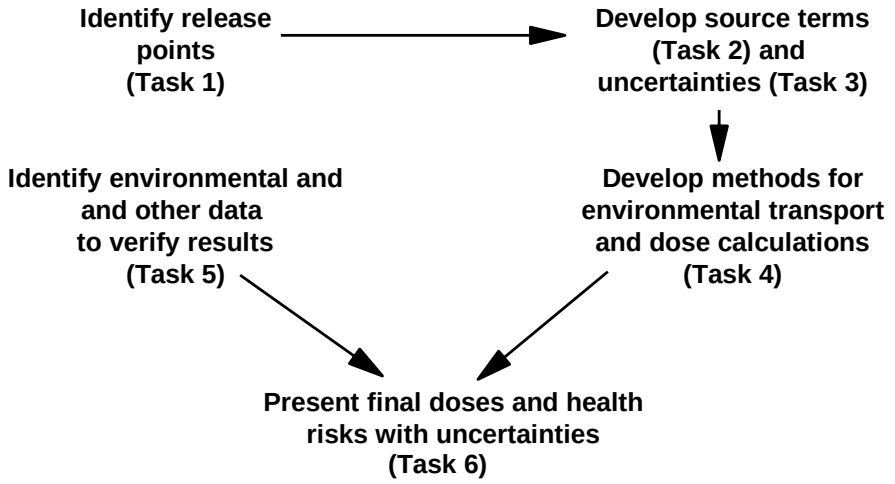



**Figure 2.** The pathway approach used in the Fernald Dosimetry Reconstruction Project. Radioactive materials were released from numerous locations onsite and were transported from the facility to the environment, where people may have been exposed. We described the transport of materials in the environment using mathematical models. We checked the performance of the models by comparing their predictions with historical measurements of environmental samples. Finally, we calculated radiation doses and estimated the health risks to people living offsite from past releases.

**Overview of Project Tasks**

The goal of Task 1 was to verify and confirm all significant points of release of radionuclides to the environment from the FMPC since operations began in 1951. This confirmation was based on documentation from the original contractor, National Lead Company of Ohio, Inc. (*NLO*), and the contractor in the late 1980s, Westinghouse Materials Company of Ohio (*WMCO*). The goals of Tasks 2 and 3 of the project were to estimate the quantity and characteristics of uranium and other radionuclides released from the facility and to determine the *uncertainties* associated with these release estimates. The release estimates are collectively known as the source terms.

The goal of Task 4 was to develop mathematical models that describe the movement of radionuclides released from the site in the environment around the FMPC and to relate the models to all available environmental data. The environmental transport models allow us to translate release estimates into concentrations of radioactivity in environmental media where people live, using available site-specific data. The study emphasizes the validation of the models used to make dose calculations. Task 5 focused on finding and evaluating available environmental monitoring data as well as other information such as the size of particles in airborne effluents. The information developed in Task 5 was used to develop our methods and to verify that the final results are consistent with the environmental data.

Task 6 calculates and reports radiation doses and associated risks of health effects to people living in the FMPC area. The dose that an individual may have received from operations at the FMPC depends upon numerous factors, including where the individual lived, attended school, and worked within the *assessment domain* (the region surrounding a facility for which radiation doses to people are calculated), how much time was spent there, the percentage of time spent outdoors and indoors, dietary habits, and age at the time of exposure.

This Task 6 report provides a comprehensive account of the project and includes estimates of radiation doses and health risks resulting from past releases. Task 6 uses information developed in the other project tasks to estimate radiation doses to people living within a 6.2-mi (10-km) circle about the FMPC, which is the assessment domain. This report summarizes some of the key points of the other tasks that have already been reported in detail (RAC 1991, Voillequé et al. 1995, Killough et al. 1993, Shleien et al. 1995). More recently developed methods are described fully in this final Task 6 report. The project synopsis that follows describes

**A kilometer (km) is equivalent to 0.6 mile (mi). The assessment domain for this study is 10 km, or approximately 6 mi from the center of the FMPC.**

- The information used to support our findings
- The types and amounts of radioactive materials released from the FMPC to air and water
- The mathematical models used to estimate environmental concentrations
- The use of environmental data to verify the reasonableness of the final results
- The conversion of the radiation doses to estimates of the risk of health effects.

This report is intended to make the information that it contains accessible and useful to readers with a variety of interests and backgrounds. The main body of the report (Volume I) emphasizes presentation graphics and less technical descriptions of release estimates, pathways, mathematical models, and validation procedures. This part of the report avoids equations, complex tables, and details that may not be necessary for the general reader. More specialized and detailed discussions are given in the appendices in Volume II. The titles of the appendices are listed here for the reader's reference:

A. The Garden Model for Produce and Animal Products
B. A Model of Uranium, Thorium, and Decay Products in Soil
C. Use of Soil Data to Confirm Magnitude of Airborne Releases and Deposition of Uranium Over Time
D. Variation in Concentration of Airborne Particulates Over Time
E. Radionuclide Concentrations for Water Pathways
F. Estimates of Concentrations of Uranium and Decay Products in South Plume Wells
G. Gamma Radiation Dose from Waste Storage Silos
H. Particle-Size Distributions for Dust Collectors
I. Dosimetric Methods
J. Specifications of the Nine Exposure Scenarios
K. Dose Estimates for Members of the Public Residing Near the Feed Materials Production Center
L. Determination of Air Sampler Collection Efficiency
M. Uncertainty and Calibration of the Air Dispersion Model
N. Detailed Validation Results
O. Investigation of Relatively High Background Uranium Measurements at Distant Environmental Monitoring Stations
P. Previous Studies of Uranium and Radon Releases at the FMPC
Q. Followup of Issues Related to Radon Source Term
R. Toxicity to the Kidneys from Natural Uranium
S. Lifetime Risks of Fatal Cancer for Individual Scenarios at the Feed Materials Production Center
T. Episodic Releases

**Sources of Information for the Project**

A major effort in the FDRP has been searching for and reviewing the thousands of existing documents related to operating the FMPC since 1951. It has been our practice to trace relevant information back to original sources whenever possible. In the Task 1 report (RAC 1991), we outlined five general approaches taken to locate information:

1. Conducting site visits to the FMPC
2. Investigating records and scientific literature pertaining to the FMPC
3. Retrieving and reviewing documents from NLO using their computer database of document titles
4. Examining engineering diagrams, site blueprints, historic photographs, and maps

5. Participating in discussions with current and former long-time employees.

Because we realized the importance of retrieving documents from a wide range of sources, considerable time was spent identifying types and locations of reports and records pertinent to this project. We visited many locations around the country to review documents that might provide background information on FMPC operations (Figure 3). Generally, the documentation of FMPC operations and releases comes from NLO, the former operator of the site; WMCO, the site operator from January 1, 1986, through 1992; the U.S. Department of Energy (*DOE*), and sources independent of the FMPC, such as state agencies, technical libraries, and universities. Appendix A of our Task 2/3 report (Voillequé et al. 1995) provides a detailed description of the sources and locations of documents used for the project.

**A cornerstone of a complete dose reconstruction is making full use of available measurements. Original sources of information, rather than summaries, are best when they can be located.**

While not all the original records are still available, many original documents remain in the files at the FMPC facility, in the library of the NLO offices, and in storage facilities used by WMCO. Many hours were spent examining original plant documents, particularly those related to *effluent* discharge measurements and operations and laboratory procedures. The information sources can be categorized as follows:

- Process descriptions for the various facilities
- Plant operating procedures
- Effluent sampling procedures
- Daily and monthly reports of liquid effluent discharges
- Monthly reports of airborne effluent discharges
- Original analytical data sheets recording sample concentrations
- Plant operating process logbooks
- Nuclear materials control reports
- Daily sump discharge logbooks
- Topical reports related to effluent characteristics
- Reports of ventilation system tests and evaluations
- Incident reports
- Accident investigation reports
- Letter reports of operational problems
- Production records for specific processes.

The recollections of long-time employees and retirees from the FMPC provided information on processes and procedures that were used routinely since facility start-up, and identified sources and locations of documentation. We have maintained a collection of all documentation that we reviewed since the FDRP began in 1990 (Appendix A, Voillequé et al. 1995).

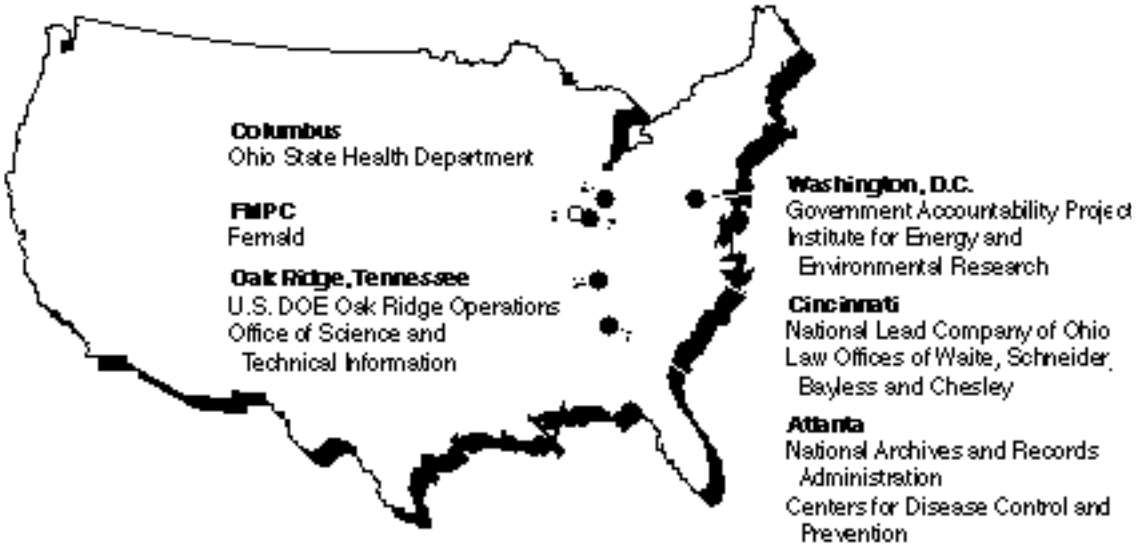



**Figure 3**. Locations visited to obtain FMPC-related documentation and information.

**Overview of FMPC Operations and Production History**

**Production area.** The FMPC processed uranium compounds as well as some uranium ore and thorium in the production area (Figure 4). Uranium-rich feed materials (ore, concentrates, or recycled feed) arrived at the FMPC from other locations. The same basic processing method was employed throughout all years of operation. From Plant 1 (the Sampling Plant), the materials passed to Plant 2/3 (the Refinery), where the uranium in feed materials was converted to uranium trioxide ($UO_3$), which is often called *orange oxide* because of its color.

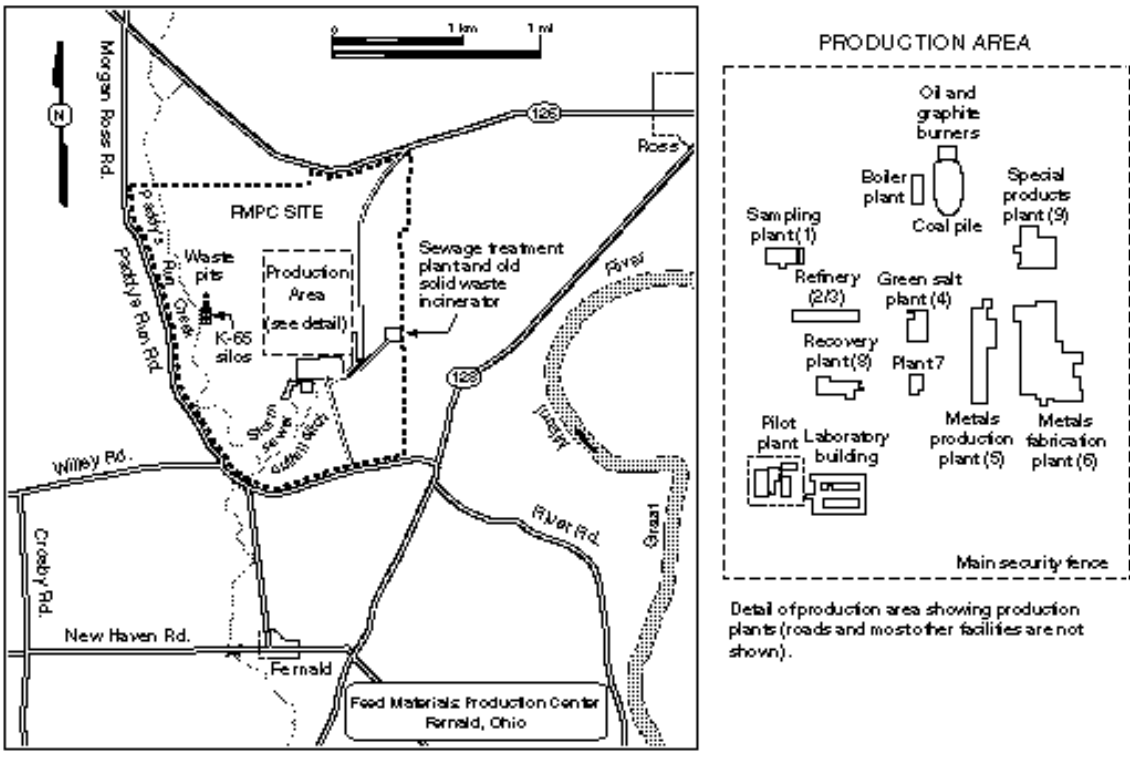



**Figure 4**. Diagram of the main facilities at the FMPC through 1989. The production area is centrally located on the FMPC site. The K-65 silos are large, concrete domed structures that store radium-containing waste from the uranium ore extraction during the early years of site operations. The six waste pits, north of the K-65 silos, were used for the disposal of dry solid wastes and for liquid wastes. The Great Miami River flows to the east of the site. *Paddy's Run Creek*, a small intermittent stream at the western boundary of the site, flows into the Great Miami River about 3 mi (5 km) to the south of the FMPC.

Waste materials from these processes were treated in various ways depending on the physical form of the waste. The *K-65 silos* contain residues from extracting uranium during processing in the early 1950s; these residues have a high radium content, which produces radon gas. Liquid processing wastes were treated at the General Sump before being sent to the *waste disposal pits*. Liquids from the waste pits, along with runoff and effluent from the sewage treatment plant were eventually piped to the Great Miami River. Most solid waste materials were sent directly to the waste pits. Flammable solids were burned in the incinerator near the eastern edge of the facility, or in the burn pit near the waste pits. The FMPC also operated a graphite burner from 1965 to 1984, an oil burner from 1962 until 1979, and an incinerator for liquid organic wastes that was installed in 1983. For more information, see our source term report (Voillequé et al. 1995).

The magnitude and extent of production activities conducted from 1951 to 1988 provide some guidance in understanding the amounts of materials that might have been released to the environment. Material was received, processing occurred, and products were shipped on a fairly regular schedule. During Fiscal Years (FYs) 1952 through 1980, the FMPC received about 362 thousand metric tons of uranium (*MTU*) (796,400,000 pounds) and shipped about

358 thousand MTU (787,600,000 pounds) to offsite locations (Audia 1977, FMPC 1988). Plant production records also specified the level of *enrichment* of processed uranium, which relates to the concentration of uranium-235 relative to uranium-238. Approximately 54% of the materials received and shipped were *natural uranium* (0.72% uranium-235); about 20% were enriched uranium (typically, 0.95–1.25% uranium-235 at the FMPC); and some 26% were *depleted uranium* (typically 0.14–0.20% uranium-235 at the FMPC). Uranium shipments from the FMPC tended to follow the pattern of materials received at the site during most of the years of operation.

The amounts of material released to the environment depended on the processes used. As a result, knowledge of specific production rates for different processes is helpful for estimating releases of radioactive materials from the facilities onsite. Figure 5 summarizes the total quantities of uranium produced in each plant for 1952 through 1988. The quantities are given in units of metric tons of uranium.

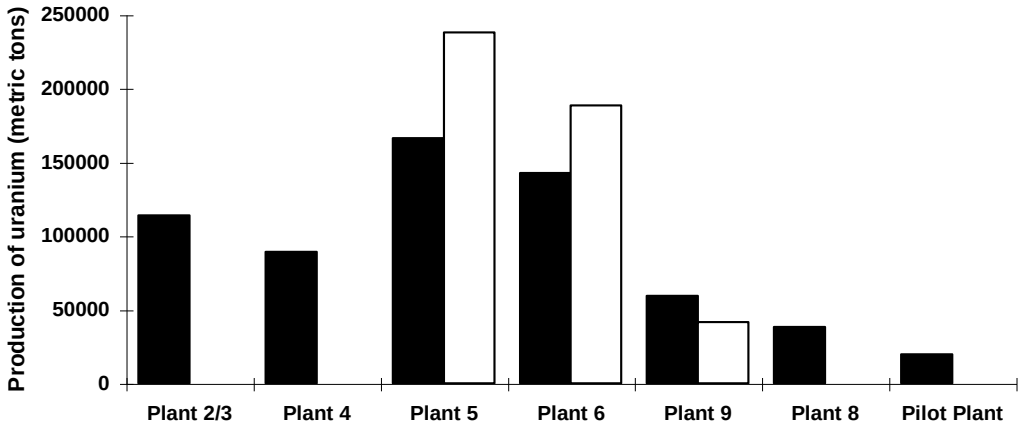



**Figure 5**. FMPC plant production for 1952 through 1988. Each plant produced a different product: uranium trioxide in Plant 2/3, uranium tetrafluoride in Plant 4, metal *derbies* (dark bars in figure) and ingots (white bars) in Plant 5, machined (dark) or rolled products (white) in Plant 6, machined products (dark) and uranium ingots (white) in Plant 9, uranium recovered from scrap materials in Plant 8, and uranium tetrafluoride in the Pilot Plant. Plant 7 produced uranium tetrafluoride for only two years, from June 1954 through May 1956, before it was closed and subsequently dismantled.

The most important radioactive material from the FMPC production facilities was uranium, but other radionuclides were also released in smaller quantities from these facilities. Thorium production was limited to a few facilities and to brief time periods; it was estimated to have been only

**Thorium was processed for brief time periods at the FMPC. During 1954-1955, thorium metal was processed, and thorium scrap metals were machined. Much of the thorium production took place in the Pilot Plant, a facility that was used for process testing and experimental operations.**

about 0.4% of the uranium production. The radionuclides released with uranium at the FMPC can be separated into three categories: (1) *naturally occurring radionuclides* found in uranium ores, (2) fission products, such as strontium-90, technetium-99, ruthenium-106, and cesium-137, and (3) activation products such as neptunium-237, plutonium-238, and plutonium-239,240. The naturally occurring radionuclides in uranium ores include protactinium-234m and isotopes of uranium, radium, and thorium.

Radioactive decay of uranium and thorium isotopes produces a series of other radionuclides, collectively referred to as decay products. In most of the material received by the FMPC, the uranium had been chemically separated from the other decay products. Some of the uranium received at the FMPC was recycled; that is, it had been irradiated in a nuclear reactor at another location, and contained fission and activation products as contaminants in the uranium. Recycled uranium was not processed at the FMPC before October 1962, so releases of fission and activation products did not occur before that time.

**Radon from waste storage silos.** Radon has been released from large quantities of a particular kind of waste material stored onsite, called K-65 material. The waste resulted from processing a certain type of uranium ore in the 1950s at the FMPC. This material contains high concentrations of the radionuclide, radium-226, and thus, acts as a continuous source of radon-222. The K-65 material at the FMPC has been stored primarily in large concrete storage silos, called the K-65 Silos, located near the west side of the site (Figures 4 and 11). The silos have had problems of deterioration, almost since the time of construction. Because of these problems, repairs and improvements to the silos were made from the 1960s through the 1980s. The most important change that affected the release of radon from the silos was made in 1979. This was the sealing of cracks and fissures that penetrated the silo domes. The addition of an exterior foam layer on the silo domes in 1987 further reduced the emissions of radon. Changes in the rate of radon release coincided with the times of these and other changes to the silos. As a result, releases of radon and its decay products from the silos were estimated differently in different periods according to the changes that had occurred.

**Estimating Releases from the FMPC**

Radionuclides were released from the FMPC to air, surface water, and groundwater. Radionuclides reached the groundwater from the migration of surface water into the ground, primarily following discharge to Paddy's Run Creek. The quantities of radionuclides that were released from the FMPC were calculated for releases to air, surface water, and in groundwater (Voillequé et al. 1995) based on thorough searches of records documenting operations and effluent and environmental monitoring. In some cases, data from which estimates could be derived were not available. To fill these gaps, we used statistical methods that quantify a possible range of values that could have existed. We focused our efforts and resources on the radioactive materials that were likely to be the largest contributors to radiation dose to those living offsite through a process called screening.

In 1985, a study at the FMPC examined particulate materials removed from stack releases by scrubber spray and trapped by filter arrangements called *dust collectors*. The materials were analyzed for uranium and other radionuclides, that included 14 decay, fission, and activation products (Boback et al. 1987). The results of the measurements in air, reported as radionuclide concentrations per kilogram uranium, formed the basis for judging the relative importance of the uranium and decay, fission, and activation products to offsite radiation doses. We used a screening methodology developed by the National Council on Radiation Protection and Measurements (*NCRP*) (NCRP 1989) to calculate the relative contribution to the radiation dose of the various radionuclides in airborne particulate releases. The results showed that the release of uranium was by far the most important contributor to the potential dose from airborne releases from the uranium production area, contributing 85% of the dose (Voillequé et al. 1995, Appendix D). The thorium isotopes, thorium-232 and thorium-230, contributed 5% and 4%, respectively. The other radionuclides contributed less than 1% of the dose. The screening calculations clearly showed the relative unimportance of other radionuclides in particulate releases compared to uranium, and it helped to concentrate our efforts and resources on the most important contributors to dose.

**Screening calculations indicated that doses from the inhalation of materials released from the FMPC would be 100 to 1000 times higher than doses received from drinking water from the Great Miami River near the FMPC**.

**Detailed dose calculations for airborne releases considered doses from uranium isotopes (uranium-234, uranium-235 and uranium-238), thorium isotopes from thorium production, and decay products of these radionuclides, as well as from radon-222 and its decay products.**

For surface water pathways, the relative importance of uranium and decay, fission, and activation products to offsite doses was based on a similar NCRP methodology (NCRP 1995). For liquid releases, the radium isotopes were found to be of primary importance (Voillequé et al. 1995, Appendix D). Screening calculations suggested that doses from the inhalation pathway would be at least 1000 times higher than doses from ingesting river water. This screening process emphasized the need for focusing our resources most heavily on the release estimates and dose evaluation methods for airborne releases.

Our particular approach to reconstructing releases to air and water depended on the type and quality of data available. Table 1 summarizes the types of information we used to reconstruct releases from the different sources at the site. As shown in Table 1, the release estimates, or source terms, are based on original measurement data, on mathematical modeling, and on a combination of types of information. The specific types and amounts of information used to reconstruct the releases helped determine the magnitude of the uncertainties for each type of release. Source terms derived primarily from effluent monitoring data have smaller uncertainties than source terms that are calculated using models. For example, surface water releases were based on daily effluent monitoring data. Consequently, uncertainties about these estimates are smaller than uncertainties associated with the Plant 8 scrub-

ber releases, which were based on mathematical modeling and measurements of related parameters.

**Table 1. Types of Information Used for Uranium and Radon Release Estimates by Decade**[a]

| Source term | 1950s | 1960s | 1970s | 1980s |
|---|---|---|---|---|
| Air | | | | |
| Dust collectors | 2 | 2 | 2 | 1 |
| Plant 2/3 scrubbers | 3 | 3 | 3 | 3 |
| Plant 8 scrubbers | 3 | 3 | 3 | 3 |
| K-65 silos (radon) | 3 | 3 | 3 | 3 |
| Surface water | 1 | 1 | 1 | 1 |
| Groundwater | NA [b] | 3 | 3 | 1 |

[a] 1 Estimates are based primarily on original effluent or environmental monitoring data.
2 Estimates are based on effluent monitoring data from periodic reports and normalized or representative release rates for unmonitored periods or sources. Factors, such as sampling *bias*, that affect measurements are considered.
3 Estimates are based on mathematical modeling and measurements of related parameters.
[b] NA = not applicable, no exposure.

For all source terms, the most important characteristics of the radionuclide release for the study included the

- Nature of release (routine or *episodic releases*)
- Magnitude or size of the release
- Time of release (day or night, season of the year)
- Radionuclides released.

For radionuclide releases to the atmosphere, two other factors were important in determining the radiation doses to people living offsite: (1) the chemical form of the discharge and (2) the physical characteristics, primarily the size distribution, of the released particles. These factors are important in determining how the materials are metabolized in the human body. Soluble compounds are readily absorbed into the blood and are rapidly distributed throughout the body. Chemical forms that are insoluble in body fluids tend to be retained in the lung for a long time and are only gradually transported to other tissues.

Knowledge of particle size distribution is important for calculating the amounts of radioactive material that were deposited on the ground following a release. Particle size is also important for estimating the radiation dose to the lungs from inhaling the particles.

**Uncertainties in Estimating Releases and in Model Calculations**

Scientific investigations often result in estimates of quantities that are not precise, and it is common practice for investigators to provide some estimate of the uncertainty, or level of confidence, they have in their estimates. Determining the uncertainties associated with the release quantities and mathematical models (Task 3) has been an important part of the FDRP, and uncertainties were evaluated at each step of the project. To do this, we ordinarily used a *Monte Carlo procedure,* where random samples from a distribution of possible inputs

were used for multiple calculations, and the calculated values were combined to estimate the uncertainty distribution of the result.

A Monte Carlo procedure is used to estimate releases because of normal variability in measurements or lack of knowledge about the parameters on which the estimates depend. This approach is different from calculations based on point estimates of parameters that yield a single, or deterministic, result. For more information about the uncertainty of source term estimates, see Voillequé et al. (1995). Estimating the uncertainties associated with the environmental transport models was also a critical part of the Task 6 work. The distribution of values for these parameters was determined from original site-specific measurements, from scientific literature, or, in the absence of other data, by subjective judgment of experts. For more information on uncertainty analysis used in environmental transport and dose calculations, see the appropriate appendices in this report or Killough et al. (1993).

## RELEASES TO THE ATMOSPHERE

Atmospheric releases from FMPC operations originated from monitored and unmonitored effluent release points from the buildings where uranium was processed, from the K-65 silos where waste residues were stored, and from outside areas such as the waste pits and incinerators. Radioactivity in some airborne effluents from the FMPC production area was partially removed by dust collectors (filters) or *scrubbers* (chemical sprays) before the effluents were discharged to the air.

Dust collectors employed bag filters to remove airborne particles from exhaust streams. To reduce worker exposure, ventilation air for process areas was ducted to dust collectors designed to remove airborne particulate material (Ross and Boback 1971). The dust collectors recovered valuable uranium that would otherwise be lost. Many of the dust collectors were sampled to estimate the amounts of uranium released to the environment. A typical sampling system is shown in Figure 6. Air was drawn from the duct and the uranium particles were trapped by the filer, which was periodically removed and taken to the laboratory for analysis.

**There were a total of 131 stack release points: 125 from production plants and six from non-production areas, including cooling towers, the graphite burner, the liquid organic incinerator, the new solid waste incinerator, the oil burner and the old solid waste incinerator. We estimated releases from dust collector and scrubber exhaust stacks, and from several unmonitored sources, like the K-65 silos.**

Scrubbers cleaned the effluent stream using sprays of either acid or caustic solutions to remove particulate matter from the air stream being discharged to the atmosphere (Figure 7). Most of the liquid, or scrub liquor, was collected in a separator chamber and returned to a reservoir. To inhibit the escape of uranium-containing liquid droplets, mist-eliminating systems were used. Although the mist eliminators trapped most of the droplets, some agglomerated into larger droplets and escaped back into the exhaust gas stream in a process called *entrainment*. Exhaust air from scrubbers of Plant 2/3 and Plant 8 contained particles that were not collected by the scrubbers as well as droplets of entrained scrub liquor.

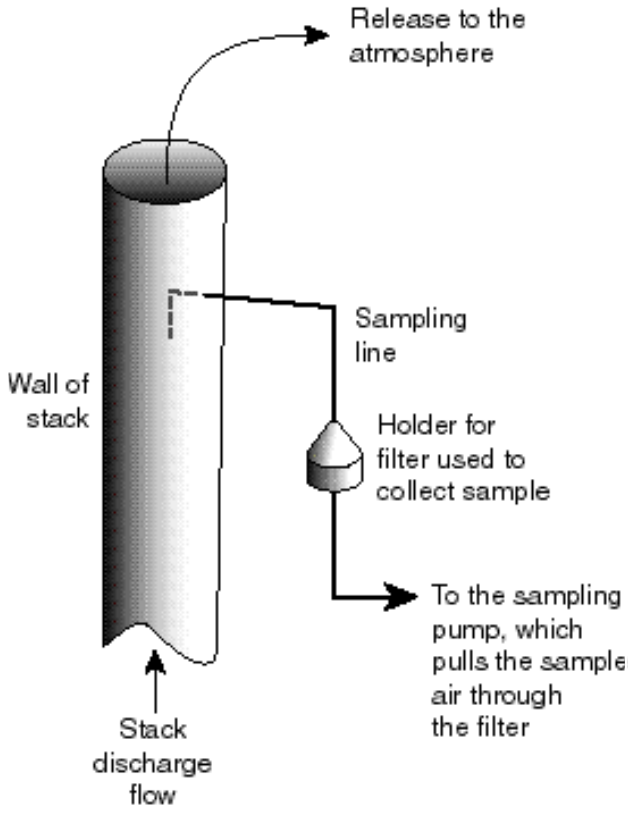



**Figure 6.** Schematic diagram of the dust collector stack sampling system.

Releases of uranium from unmonitored sources (incinerators, building ventilation, laboratory hoods, unmonitored process emissions, and waste pits) and accidental releases (fires, spills, and episodic releases) were carefully evaluated, but the quantity of material released was found to be much less than from the dust collectors and scrubbers. In addition, radon releases were calculated for the K-65 silos, located near the west side of the site, and for drummed K-65 material temporarily stored on the Plant 1 pad in the early 1950s.

Releases of radon and radon decay products were calculated for two locations. The most important source was the K-65 silos located near the waste pits on the west side of the site. Releases from drums of K-65 material, which were stored temporarily on the Plant 1 pad in the early 1950s, were also estimated.

**Releases of Uranium from Dust Collectors**

The onset and growth of the dust collector effluent sampling program are documented in monthly industrial hygiene reports (Voillequé et al. 1995). Periodic sampling of roughly 10 dust collector exhaust stacks occurred as early as 1953; however, a continuous sampling program did not begin until April 1955. The sampling program was initiated in 7 stacks in Plants 4 and 5 and grew fairly rapidly to encompass 30 stacks 6 months later. There were further increases in the program to a maximum of 50 sampling systems in May 1958. With the shutdown of systems in Plant 1 and in the Pilot Plant at the start of 1960, there was a decline to 44 samplers for dust collector exhausts. With reductions in plant production and

staff in later years, the intervals between sample analyses were greater and routine reports contained less detail.

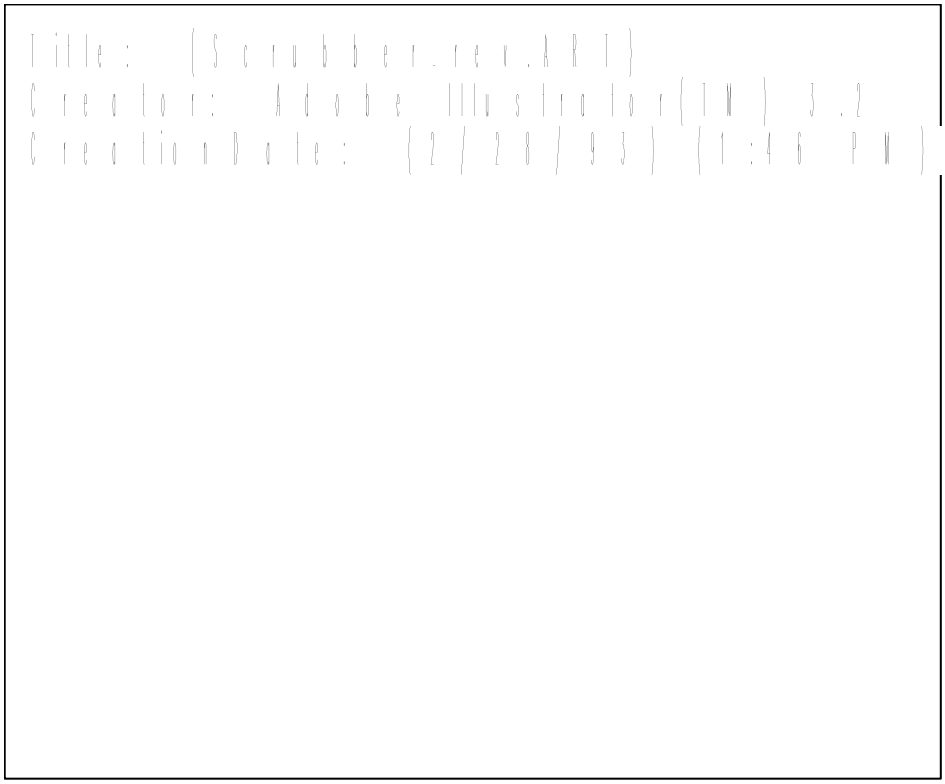



**Figure 7.** Schematic diagram of a scrubber. Exhaust gas entering the scrubber is forced through a liquid spray into a Venturi tube. The gas then passes through a separator chamber and into the outlet duct. The spray entrains most particles into liquid droplets. Most of the liquid (or scrub liquor) is collected in the separator chamber and returns to a reservoir from which it is recycled. The scrub liquor of the Plant 2/3 and Plant 8 scrubbers was changed periodically and uranium was recovered from it. To inhibit the escape of the uranium-containing droplets mist-eliminating systems were used. The figure indicates a wire mesh mist eliminator in the outlet duct (as in Plant 2/3), which would trap most droplets. Some of the trapped liquid was reentrained into the gas stream as large agglomerates and escaped to the atmosphere. Evaporation of the liquid produced relatively large solid particles.

Estimates of uranium releases from individual dust collectors at the FMPC were tabulated from the original records, which were usually monthly reports of the measurements. Production began at the FMPC in 1951, but release estimates were not reported for most production plants until 1953 or later (Boback et al. 1987). The reports also did not include estimates of releases that were undetected by the analytical procedure or not measured because a sampling system was temporarily out of service. To develop a more representative estimate of the source term to air for the dose reconstruction, we estimated the unmonitored and undetected releases. For undetected releases, we used one-half of the smallest reported amount of material on a filter that had been recorded for all sampling periods. This kind of

procedure was necessary because early analytical data sheets did not indicate the minimum detectable amount of material on a filter. For unmonitored releases, we used releases measured during previous or subsequent sampling periods to estimate the release. In addition, we considered the following factors that may have affected the accuracy, or bias, in the effluent measurements:

- Nonrepresentative sampling that may have occurred when particles were not uniformly mixed in the exhaust at the location of the sampler
- *Anisokinetic sampling* that may have occurred when there was a mismatch between the flow rate in the sampling line and that in the stack
- Losses of particles in the sampling line that may have occurred when particles were deposited on the walls of the line or impacted in bends in the lines between the probe and the collection filter.

Overall, corrections for unmeasured releases and for sampling bias led to revised release estimates that were approximately 50% higher than original, site-based estimates of dust collector releases (Voillequé et al. 1995). Table 2 shows that the *median,* or 50th percentile, estimate of total releases from the FMPC dust collectors for 1951–1988 was about 140,000 kg uranium. The 5th and 95th percentiles of the estimate were 120,000 and 170,000 kg uranium, respectively. Most releases of uranium from the dust collectors occurred during the 1950s. Principal contributors to the releases during that decade were Plants 4, 7, and 5. (Note that in Table 2, the medians for the 1950s, 1960s, 1970s, and 1980s do not add up to the median given in the bottom line for the entire period 1951–1988. This is not an error, but rather a statistical effect. If we were considering arithmetic means, the sum would be exact, but this is not the case for percentiles.)

The physical and chemical characteristics of the uranium releases were developed as part of the source term (Voillequé et al. 1995). The chemical form of the materials discharged affects the particle density, the transport and deposition of released uranium, and the estimate of radiation dose.

**Table 2. Summary of Uranium Release Estimates for All FMPC Dust Collectors**

| | Median, or 50th percentile value | Other percentiles in distribution of release estimates (kg U) | | | |
|---|---|---|---|---|---|
| Period | (kg U) | 5th | 25th | 75th | 95th |
| 1950s | 120,000 | 96,000 | 110,000 | 130,000 | 150,000 |
| 1960s | 21,000 | 18,000 | 19,000 | 22,000 | 24,000 |
| 1970s | 3,100 | 2,500 | 2,800 | 3,400 | 3,800 |
| 1980s | 2,100 | 1,700 | 1,900 | 2,400 | 2,700 |
| 1951–1988 | 140,000 | 120,000 | 130,000 | 160,000 | 170,000 |

**Releases of Uranium from Plant 2/3 Denitration Operations**

Effluents (mainly fumes of nitrogen oxides) from the *denitration* process in Plant 2/3 (where the uranium in feed materials was converted to uranium trioxide) were treated by a wet scrubber before discharge to the atmosphere (Figure 7).

The orange oxide produced by denitration was transferred by a vacuum process called *gulping*. The transfer line carried the uranium trioxide to two cyclone separators that collected most of the product and fed it into a storage hopper. Particles of orange oxide not collected by the cyclones were carried in the exhaust air to the scrubber. Releases of uranium from the scrubber exhausts were not sampled until recently. In June 1988, an investigation found that releases from Plant 2/3 processing activities were the source of higher offsite air concentrations that had been observed (Investigation Board 1988). This inquiry led to special measurements of the discharges during gulping and routine scrubber operations. Routine scrubber operations proved to be an important component in the reduction of the total site releases (Semones and Sverdrup 1988).

We calculated releases from the Plant 2/3 scrubber system using models of (a) scrubber penetration by particles and (b) mist entrainment, which were based on the recent effluent measurements (Semones and Sverdrup 1988). Monte Carlo techniques were used to sample uncertainty distributions of the possible input parameters used in the calculations and to develop a distribution of release estimates, from which median values were obtained. The parameters considered in calculating the release estimates were scrubber outage fraction, scrub liquor concentration, entrainment release factor, uranium trioxide production, amount of uranium trioxide in a denitration pot, time to transfer the uranium trioxide to a storage hopper (gulping time), and gulping release rate (Voillequé et al. 1995). Original plant operating logbooks from 1969, 1970, and 1973 and shift foremen's logbooks for 1956–1962 and 1967 provided supporting information on parameters important to calculate releases as a result of gulping operations.

The highest releases from the Plant 2/3 scrubbers were estimated for the period 1957–1961. The median release estimate from this source for the entire period of FMPC operation was 66,000 kg uranium (Table 3). Fifth and 95th percentiles for the estimate are 56,000 and 78,000 kg uranium, respectively. The small particles of uranium trioxide that penetrated through the scrubber compose about 40% of the release. The larger fraction (approximately 60%) would have been uranyl nitrate in entrained droplets (Voillequé et al. 1995).

**Table 3. Summary of Uranium Release Estimates for Plant 2/3 Scrubbers**

| | Median, or 50th percentile value | Other percentiles in distribution of release estimates (kg U) | | | |
|---|---|---|---|---|---|
| Time period | (kg U) | 5th | 25th | 75th | 95th |
| 1950s | 24,000 | 18,000 | 21,000 | 26,000 | 32,000 |
| 1960s | 19,000 | 14,000 | 17,000 | 21,000 | 25,000 |
| 1970s | 22,000 | 17,000 | 20,000 | 25,000 | 29,000 |
| 1980s | 980 | 730 | 850 | 1,100 | 1,600 |
| 1953–1988 | 66,000 | 56,000 | 62,000 | 71,000 | 78,000 |

**Releases of Uranium from Plant 8 Scrubbers**

Plant 8 recovered uranium from various residues and used furnaces to eliminate combustible material and to oxidize uranium metal. Ten air-scrubbing systems (Figure 7) were installed to remove acidic gases and particles from exhaust air streams by contact with droplets of caustic liquid. Six of the scrubbers handled hot exhaust gases from the rotary kiln and

various furnaces, and four scrubbers treated ventilation air collected above the digestion and other process tanks. The furnace exhaust scrubbers were the largest sources of uranium releases. Two important limitations affect our release estimates for the Plant 8 scrubbers:

1. The exhausts from these systems were not sampled regularly.
2. There were no reported measurements of the sizes of the particles or liquid droplets released to the atmosphere.

Review of plant memoranda on Plant 8 scrubber performance and analytical data sheets that contained data on scrubber efficiency showed that uranium collection efficiencies were not constant. They varied with time for a particular scrubber, and some scrubbers were much more efficient that others. We used available plant records, Plant 8 production (uranium recovery) data, memoranda, and analytical data sheets to estimate parameters needed to model scrubber performance during the years 1953–1981. Data collected in the 1980s on short-term measurements of release rates from the various stacks were used to calculate releases from 1981 through 1988. For both time periods, simple models were used to estimate the releases from individual scrubbers. The distributions of all of the parameters used in the Monte Carlo calculations were broad because of variability in the release rates, lack of knowledge, and limited historic data.

The median release estimate from the Plant 8 scrubbers for the entire operating period was 81,000 kg uranium, with 5th and 95th percentiles equal to 56,000 and 130,000 kg, respectively (Table 4). The table illustrates the importance of scrubber releases during the 1960s, when Plant 8 production was highest.

**Table 4. Summary of Uranium Release Estimates for Plant 8 Scrubbers**

| Time period | Median, or 50th percentile value (kg U) | Other percentiles in distribution of release estimates (kg U) | | | |
|---|---|---|---|---|---|
| | | 5th | 25th | 75th | 95th |
| 1950s | 29,000 | 17,000 | 23,000 | 37,000 | 53,000 |
| 1960s[a] | 47,000 | 30,000 | 39,000 | 57,000 | 78,000 |
| 1970s | 1,700 | 1,000 | 1,400 | 2,100 | 2,700 |
| 1980s | 1,400 | 980 | 1,200 | 1,600 | 2,000 |
| 1953–1988 | 81,000 | 56,000 | 69,000 | 95,000 | 130,000 |

[a] In making these estimates we assumed that the bypass for the uranium ammonium phosphate (*UAP*) scrubber operated 10% of the time between September 1963 and April 1966.

Figure 8 compares the quantities of uranium released annually from the dust collectors, the Plant 8 scrubbers, and the Plant 2/3 scrubbers. The dust collectors dominated the releases in the 1950s, with 120,000 kg of uranium discharged. A maximum of 54,000 kg of uranium was released in 1955 alone. In the 1960s, the Plant 8 scrubbers dominated the releases, with approximately 47,000 kg of uranium released during that decade, compared to 21,000 and 14,000 kg of uranium from the dust collectors and Plant 2/3 scrubbers, respectively. In the 1970s, the Plant 2/3 scrubbers released more uranium than the dust collectors and Plant 8 scrubbers. In the 1980s, the dust collectors contributed most to the total uranium releases (2,100 kg of uranium), although the magnitude of all releases in the 1980s was significantly less than in previous times.

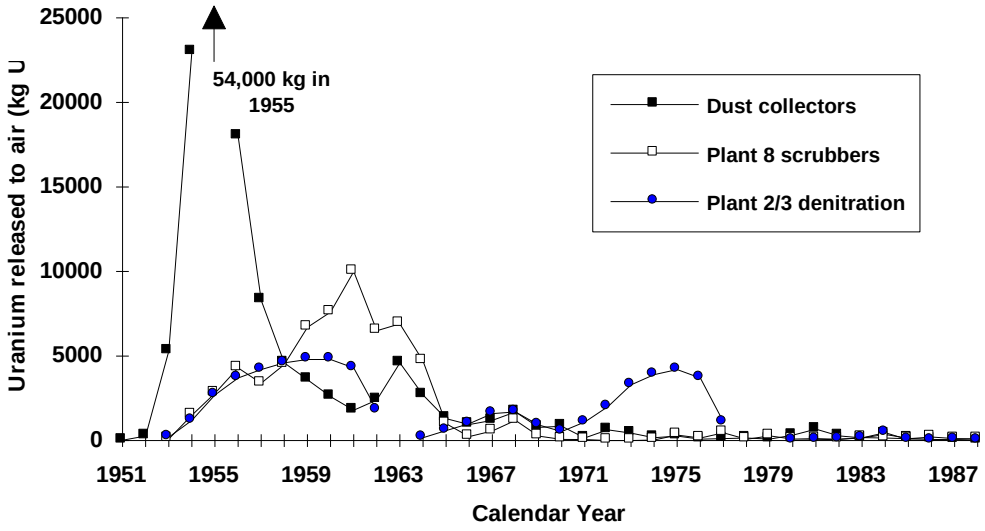



**Figure 8.** The median, or 50th percentile, estimates of annual releases of uranium to the atmosphere from the dust collectors, the Plant 8 scrubbers, and the Plant 2/3 denitration operations. The relative importance of each of these sources to the total atmospheric uranium release changed with each decade.

**Miscellaneous Unmonitored and Accidental Releases of Uranium**

This project included a thorough evaluation of atmospheric releases of uranium from unmonitored sources (five waste incinerators, building roof vents and open doors or windows, laboratory hoods, unmonitored process emissions, and waste pits) and accidental releases (fires, spills, and episodic releases). Episodic releases are actual accidental releases that were large enough to require special treatment during environmental transport and dose assessment. The criteria used to define these events were

- The event under consideration caused the release rate from the FMPC to increase by 10 times or more above the value that would otherwise have been observed, and
- The duration of the high release rate caused by the event was less than 10 days.

Dose consequences of episodic releases are described in Appendix T of Volume II. One episodic release of uranium hexafluoride could have been significant in terms of chemical toxicity. Radiation doses from episodic releases were small in comparison to cumulative doses fromroutine releases. Nonroutine releases from other events, such as spills, fires, and leaks of gaseous uranium hexafluoride and uranyl nitrate, were estimated on the basis of the event's frequency of occurrence. Figure 9 illustrates the relative importance of the various sources of uranium releases to air. Figure 9a is plotted on a logarithmic scale to show the uncertainty distributions clearly. Figure 9b is plotted using a linear scale, which provides a clear visual picture of the relative magnitudes of these sources.

**Figure 9a**. Vertical scale is logarithmic.

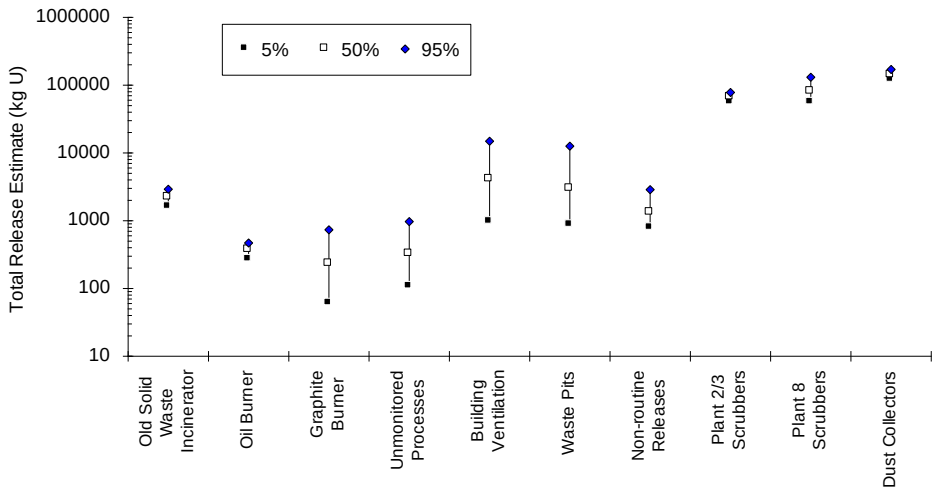



**Figure 9b**. Vertical scale is linear.

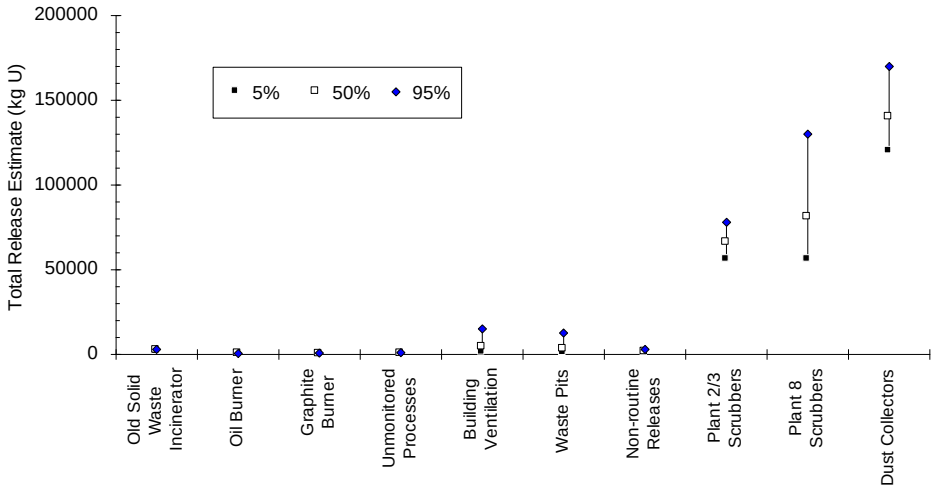



**Figure 9.** Relative importance of estimated releases from miscellaneous unmonitored sources of atmospheric uranium compared with releases through scrubbers and dust collectors. The 50th percentile point represents the median. The 5th and 95th percentile points encompass a 90% probability interval on the total estimates.

Although releases from these nonroutine events were believed to be relatively minor compared with the dust collectors and scrubber emissions, the documentation to support that conclusion was limited in most cases, and some of the methods that were used in the past to estimate releases needed improvement. Our detailed assessments of these releases are presented in Appendix K of the Task 2/3 report, where we documented the magnitude of these sources with uncertainties (Voillequé et al. 1995). These release estimates are included as part of the total atmospheric source term.

Most of the miscellaneous unmonitored sources were not releasing uranium to the atmosphere over the entire production history of the FMPC. The waste pits, building ventilation, and the old solid waste incinerator were the largest unmonitored sources of uranium to the atmosphere. However, the releases from these miscellaneous unmonitored sources were minor relative to the major sources of uranium emission from the FMPC.

In summary, uranium releases from the dust collectors (Table 2), the Plant 2/3 scrubbers (Table 3), and the Plant 8 scrubbers (Table 4) have dominated the uranium releases from the FMPC. Releases of uranium from the miscellaneous unmonitored sources (Figure 9) were minor relative to the major sources of uranium emissions. Figure 10 shows that airborne releases were highest in the 1950s, with over 60,000 kg (132,000 lb) released in 1955. Releases of uranium gradually declined in the 1960s and 1970s, averaging about 500 kg (1100 lb) per year during the 1980s. Table 5 lists the median estimates of total uranium releases by year from the major sources, with uncertainty bounds.

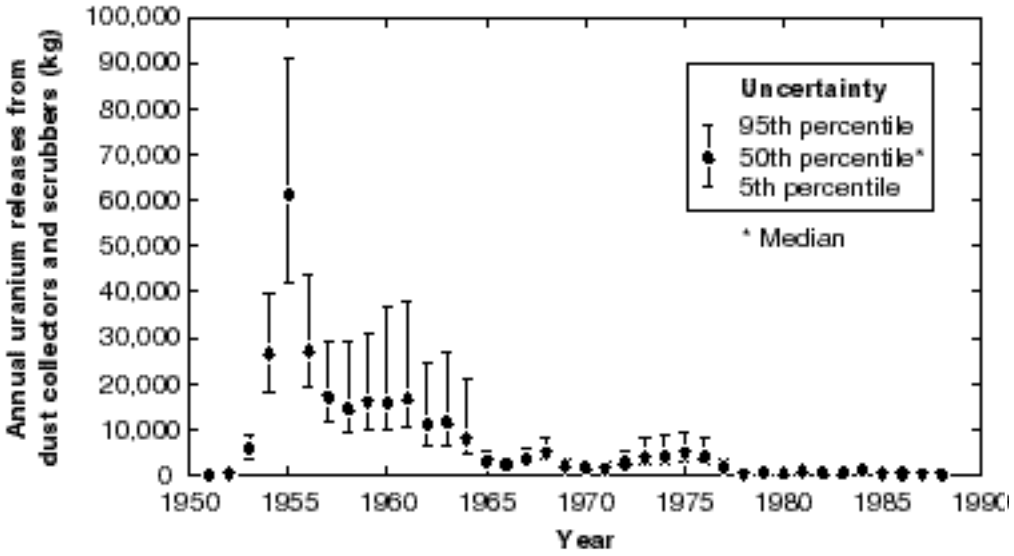



**Figure 10.** Summary of annual uranium release estimates for all dust collectors and scrubbers. The median estimate is encompassed by the 5th and 95th percentile points, which indicate a 90% probability distribution on the total estimates.

**Table 5. Annual Uranium (kg) Release Estimates[a]**

| Year | Median, or 50th percentile value | Other percentiles in distribution of release estimates | | | |
|---|---|---|---|---|---|
| | | 5th | 25th | 75th | 95th |
| 1951 | 22 | 11 | 17 | 29 | 45 |
| 1952 | 274 | 136 | 205 | 364 | 550 |
| 1953 | 5747 | 3786 | 4835 | 6862 | 8958 |
| 1954 | 26563 | 18370 | 22810 | 30818 | 39913 |
| 1955 | 61309 | 41729 | 51989 | 72276 | 91163 |
| 1956 | 27181 | 19007 | 23530 | 31905 | 43563 |
| 1957 | 17006 | 11552 | 14414 | 20334 | 29011 |
| 1958 | 14413 | 9430 | 11777 | 17996 | 28947 |
| 1959 | 15916 | 10026 | 13030 | 19884 | 30677 |
| 1960 | 15780 | 9705 | 12698 | 20557 | 36869 |
| 1961 | 16790 | 10250 | 13505 | 21973 | 38102 |
| 1962 | 11063 | 6466 | 8742 | 14224 | 24572 |
| 1963 [a] | 11567 | 6294 | 8871 | 15474 | 26693 |
| 1964 [a] | 7864 | 4494 | 5983 | 10995 | 20795 |
| 1965 [a] | 3127 | 2218 | 2680 | 3739 | 5491 |
| 1966 [a, b] | 2316 | 1668 | 2012 | 2734 | 3716 |
| 1967 | 3569 | 2541 | 3088 | 4273 | 5894 |
| 1968 | 4892 | 3406 | 4158 | 5865 | 8216 |
| 1969 | 2101 | 1467 | 1802 | 2486 | 3358 |
| 1970 | 1543 | 1035 | 1307 | 1850 | 2382 |
| 1971 | 1391 | 888 | 1138 | 1795 | 2849 |
| 1972 | 2664 | 1791 | 2250 | 3324 | 5242 |
| 1973 | 3793 | 2452 | 3133 | 4874 | 7894 |
| 1974 | 4138 | 2618 | 3396 | 5342 | 8739 |
| 1975 | 4873 | 3164 | 4050 | 6225 | 9432 |
| 1976 | 3977 | 2492 | 3193 | 5158 | 8280 |
| 1977 | 1711 | 1044 | 1407 | 2191 | 3297 |
| 1978 | 225 | 151 | 192 | 271 | 364 |
| 1979 | 372 | 141 | 244 | 554 | 991 |
| 1980 | 449 | 281 | 369 | 559 | 796 |
| 1981 | 915 | 544 | 739 | 1133 | 1559 |
| 1982 | 456 | 325 | 397 | 521 | 646 |
| 1983 | 525 | 353 | 448 | 627 | 818 |
| 1984 | 1022 | 740 | 888 | 1198 | 1632 |
| 1985 | 303 | 200 | 257 | 371 | 496 |
| 1986 | 301 | 179 | 240 | 400 | 598 |
| 1987 | 239 | 151 | 198 | 294 | 441 |
| 1988 | 148 | 74 | 103 | 205 | 335 |
| **1951-1988 Totals** | **310.000** | **270,000** | | | **360,000** |

[a] From FMPC dust collectors and scrubbers.
[b] In making these estimates it was assumed that the bypass for the UAP scrubber in Plant 8 operated 10 percent of the time between September 1963 and April 1966.
[c] Does not include an episodic release of 750 kg uranium which was evaluated separately.

**Radon Releases**

Radon releases have resulted from the onsite storage of K-65 material, a waste that remained after extraction of uranium from ores. The K-65 material contains high concentrations of radium-226 and thus acts as a continuous source of radon-222. The K-65 material at the FMPC has been stored primarily in large concrete storage silos, called the K-65 silos, located near the west side of the site. Some of this waste was stored temporarily in drums on the Plant 1 storage pad in the early 1950s (1951–1953). Figure 11 shows the location of the K-65 silos (as well as the two other waste storage silos) and the Plant 1 storage pad. Release estimates were calculated for the K-65 silos and for the drums of K-65 material on the Plant 1 pad. Appendix J of our Task 2/3 report (Voillequé et al. 1995) described the radon releases in considerable detail; much of the following information is summarized from that appendix.

**The significant radon releases are from K-65 material stored onsite. This material is a waste residue from the extraction of uranium from ore. The K-65 material was originally slurry (solid and liquid mixture), but it became drier over time in storage in the silos.**

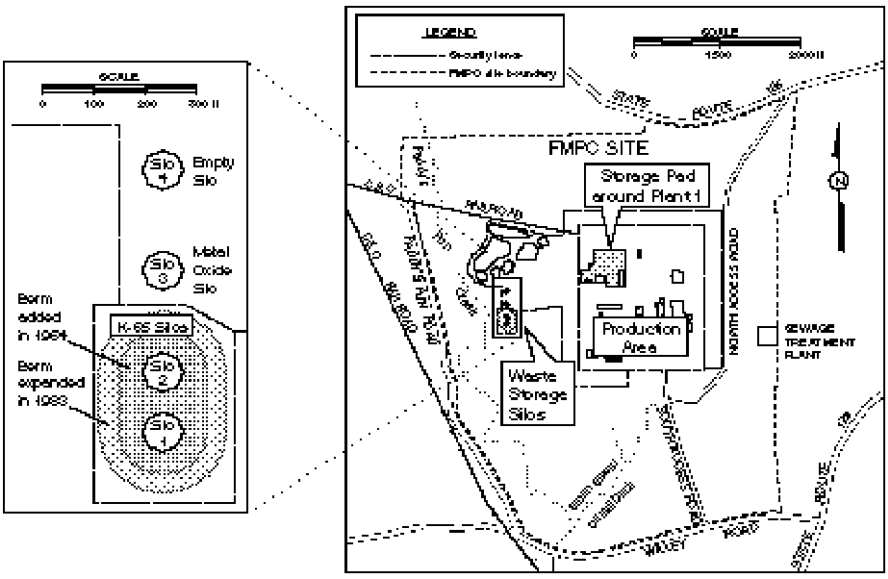



**Figure 11.** Locations of significant radon releases from K-65 material on the FMPC site. Drums of K-65 material were stored on the pad around Plant 1 in the early 1950s. The K-65 material was stored in the two K-65 silos starting in 1952 and continuing to the present.

The following other sources of radon releases from the FMPC were investigated, in the Task 2/3 report, and were found to be insignificant compared to the K-65 silos:

- Uranium ore stored in silos near Plant 1 before processing
- Processing of Belgian Congo uranium ores in Plant 2/3
- Dumping K-65 material from drums into a slurry tank and filling the silos with the slurry
- Other ore processing wastes stored in the metal oxide silo (Silo 3)

- Waste pits north of the silos area.

The drummed K-65 material that was stored on the Plant 1 pad was a relatively minor source of radon releases when compared to the K-65 silos, but this drummed material was an important source for the early 1950s. This drummed material stored near Plant 1 was eventually transferred to the silos. We calculated quantities of radon released from two sources: (1) the two K-65 silos and (2) the drummed K-65 material stored on the Plant 1 pad. Because the K-65 silos are by far the most important radon source, we focus attention in this report on them. Additional details about the radon source terms are provided in the Task 2/3 report (Voillequé et al. 1995).

**Some of the K-65 material stored at the FMPC was from uranium ore processing at Fernald. Most, however, was from similar Atomic Energy Commission work performed at Mallinckrodt Chemical Works in St. Louis, Missouri.**

**The ore was from the Belgian Congo and had a very high content of uranium and decay products, including radium. The radium was the source of the radon and decay products that were released from the silos.**

**Characteristics of and historical changes to K-65 silos.** The K-65 silos were constructed in 1951–1952 to store K-65 material (Catalytic circa 1950s, NLO 1962, Grumski 1987, Shanks and Vogel 1988). The physical characteristics of the silos are shown in Figure 12. The K-65 silos have had deterioration problems almost since the time of construction. Significant cracking in the walls and seepage of the contents was noted from the 1950s (Wunder 1954 and Martin 1957). Because of these problems, repairs and improvements to the silos were made from the 1960s through the 1980s. Figure 13 shows the silos as they appeared in 1965 (DOE 1965).

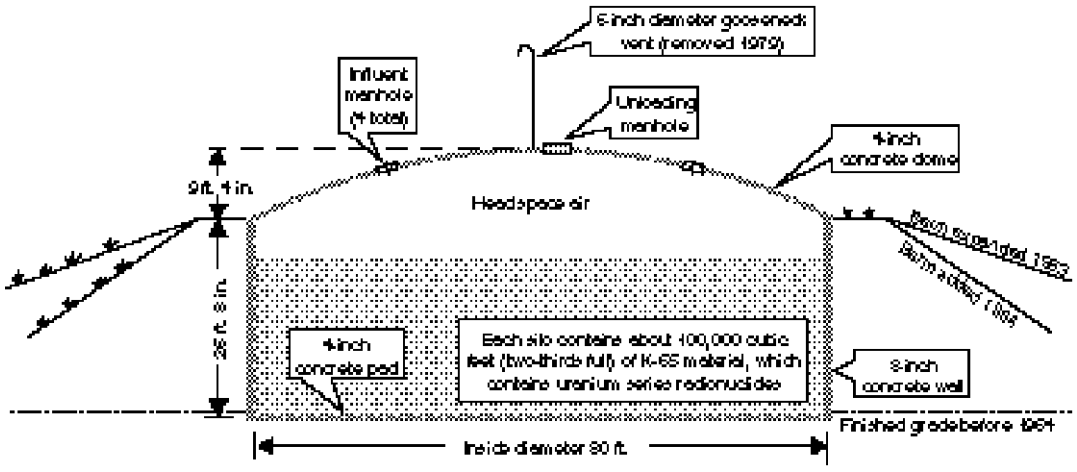



**Figure 12.** Cross-section view of a K-65 silo. Not shown are a number of 2-inch (in) diameter sounding pipes that also penetrate the silo domes.

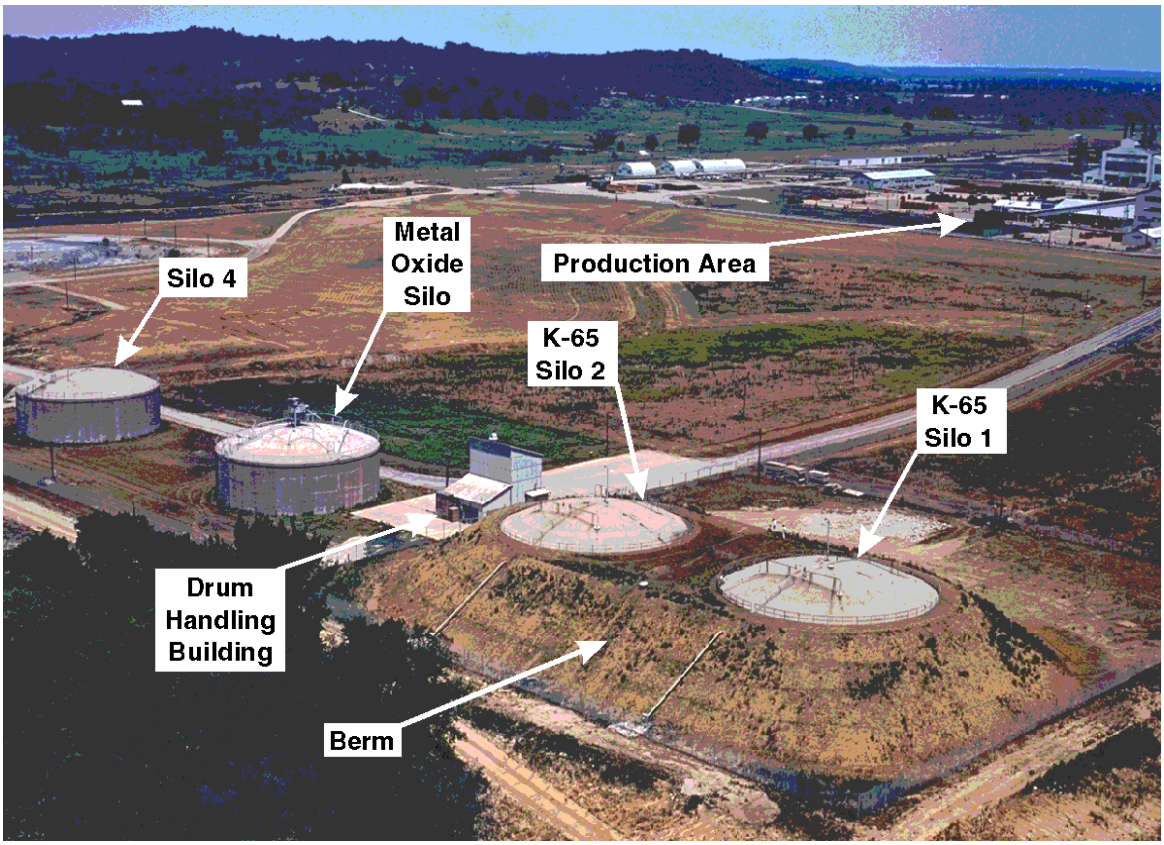



**Figure 13.** The waste storage silos in 1965, viewed from the southwest (labels added to image digitized from DOE 1965). The appearance of the silos would have been similar to this for the period 1964 (after installation of berms) to 1979 (before removal of piping and sealing of penetrations). The drum handling building was removed in 1983 when the berms were enlarged (Geesner 1983).

Not all of the changes to the silos would have had a significant effect on the releases of radon. For example, earthen berms were built around the silos in 1964, a time when radon releases occurred primarily through openings in the silo domes. Therefore, adding the berms would not alter the releases. The most important change in the radon emissions was due to sealing the openings in the domes in 1979. The most important operational changes are described in Table 6.

**A cornerstone of historical dose reconstruction is making full use of available site-specific data. Because of the lack of such data to support the conventional model for calculating radon releases, we developed and adapted other models to calculate the radon source term.**

**Table 6. Summary of Key Historical Changes to the K-65 Silos**

| Operational changes | Effect on radon releases |
|---|---|
| Filling of silo 1 begins (July 1952) | Radon releases increase as K-65 material is added to silos |
| Silo 1 full, filling of silo 2 begins (June 1953) | Releases continue to increase |
| Both silos are full and are decanted (water is removed) (September 1958) | Releases are now high, and continue at roughly constant level |
| Berm added around silos (May 1964) | No significant change expected |
| Major openings in silo domes are sealed and vent pipes are removed (end of June 1979) | Sealing produces major decrease in radon release rate |
| Radon Treatment System installed (late November 1987) and foam layer applied to silo domes (December 1987) | Foam layer insulates silo air space, reducing temperature change inside, which further reduces radon releases |
| Bentonite (clay) layer added on top of K-65 material in silos (November 1991) | Another major decrease in radon releases (outside the time frame of this project) |

Changes in the rate of radon release were assumed to have coincided with the times of these changes to the silos. As a result, we estimated radon and radon decay product releases from the silos separately for several periods based on the changes. These periods are shown in Table 7.

**Table 7. Time Periods for Reconstruction of Radon Releases from K-65 Silos**

| Nominal period | Description | Precise time period |
|---|---|---|
| 1952–1953 | Operational period of silo 1 | Mid-July 1952 to mid-June 1953 |
| 1953–1958 | Operational period of silo 2 | Mid-June 1953 to mid-September 1958 |
| 1959–1979 | Before sealing penetrations in silo domes | Mid-September 1958 to June 1979 |
| 1980–1987 | After sealing penetrations in domes | July 1979 to December 1987 |
| 1988 | After addition of foam layer | 1988 |

**Methods for estimating radon and radon decay product releases.** Release estimates for radon and radon decay products were complicated by a lack of data describing characteristics of the material in the silos and by the structural changes over the years. There are no direct measurements of release quantities for radon and radon decay product releases. In addition, until the 1980s there were very few measurements of parameters that could be used indirectly to calcu-

**Unlike uranium releases from building stacks, there were no direct measurements of radon released from the silos. Radon releases had to be estimated from related information and by developing models of how releases might have occurred.**

late radon releases. The most important data sets for reconstructing the radon releases are shown in Figure 14. Details about these data sets are discussed in the Task 2/3 report (Voillequé et al. 1995). Some minor changes to the calculations of radon releases are described in Appendix Q of this report (Volume II).

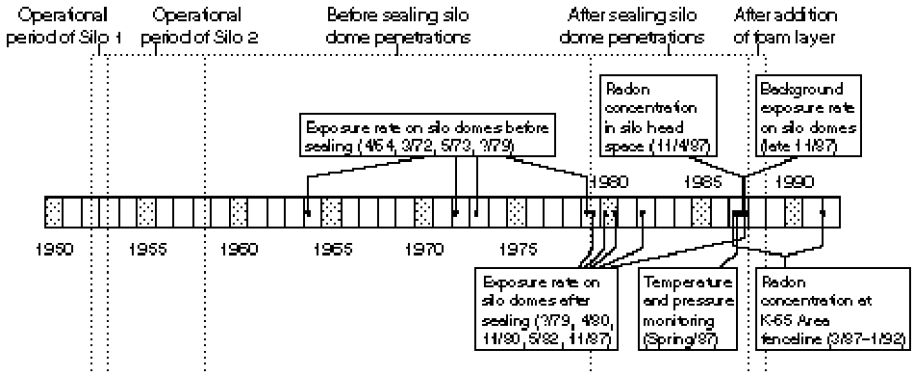



**Figure 14.** Timeline showing key measurement data related to reconstructing radon source term. Dates were not available for two sets of measurements of exposure rate on the silo domes (indicated as ?/79). Documentation for the ?/79 measurements indicates that they were taken before and after the dome penetrations were sealed.

The two main mechanisms for release of radon from the K-65 silos were (1) air exchange and (2) *diffusion*. Air exchange refers to the movement of air containing radon between the silo headspaces and the outside atmosphere. This exchange occurred through openings in the silo domes, including the open vent pipe, leaks around manhole covers, cracks in the domes, and other penetrations. For 1980–1987, the air exchange was driven by temperature changes of the headspace air, which caused the air to expand and contract. Diffusion releases were from diffusion of radon in the headspaces through the concrete domes into the atmosphere.

**Our calculations indicate that releases through air exchange are much more important than through diffusion. For 1980–1987, air exchange releases were more than 80% of the total; for 1959–1979 they contributed an even higher fraction.**

Because of the limited availability of data, models were used to estimate the quantities of radon and radon decay products that were released. A conventional model for estimating radon releases from radium-226-bearing material involves calculations of the quantity of radon formed in the material and the subsequent diffusion of the radon through the material to the outside air. The conventional model was not the primary model employed in this study because site-specific measurements of the radon diffusion coefficient and radon emanation fraction had not been made. Instead, our preferred method was the development and adaptation of other models to calculate air exchange, diffusion, and total radon releases from the data that were available. Figure 15 compares our preferred method to the conventional method. The conventional method was used for a secondary calculation to provide a limited check of our primary calculation. Details of the models can be found in the Task 2/3 report (Voillequé et al. 1995).

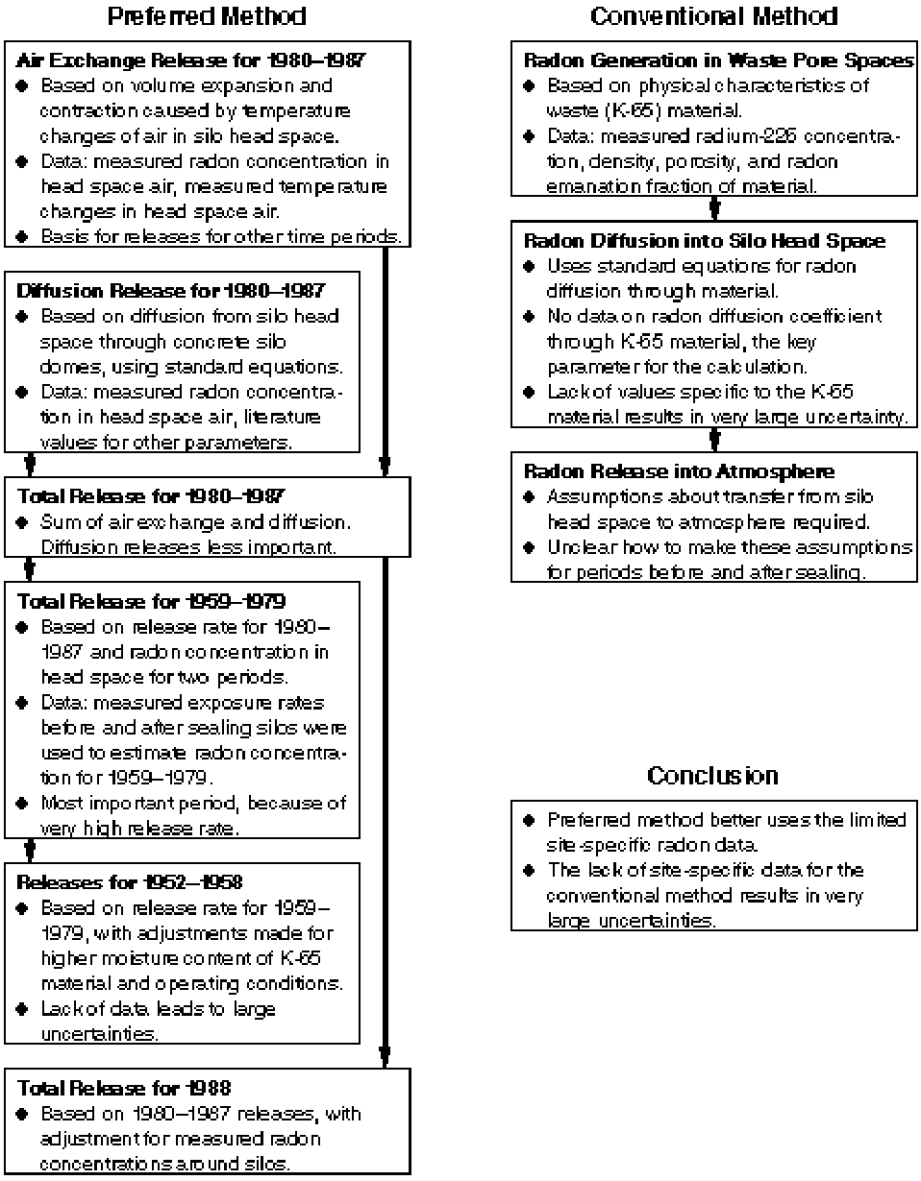



**Figure 15.** Comparison of the preferred and conventional methods for calculating radon releases from the K-65 silos.

As described earlier, a Monte Carlo procedure was used to estimate uncertainty in the results of the radon source term calculations. To evaluate uncertainty in the results, distributions of values were assigned to each parameter involved in the radon source term calculation. Table 8 summarizes the important factors in these calculations. Additional details about the uncertainty distributions assigned to each parameter are given in the Task 2/3 report (Voillequé et al. 1995).

**Table 8. Important Parameters in the Radon Source Term Calculations**

| Period | Parameters in calculation [a] |
|---|---|
| 1980–1987: air exchange releases | ● Measurement of radon concentration in silo head space air.<br>✈ Daily temperature change of headspace air, based on measurements.<br>● Volume of head space air. |
| 1980–1987: diffusion releases | ● Measurement of radon concentration in silo head space air.<br>● Radon diffusion length and porosity of silo dome concrete.<br>● Silo dome concrete thickness. |
| 1959–1979 | ● Calculated releases for 1980–1987.<br>● Measurement of radon concentration in silo head space air.<br>✈ Exposure rate measurements on silo domes, before and after sealing of silos. |
| 1952–1958 | ● Calculated releases for 1959–1979.<br>✈ Assumed factors to account for operating conditions and moisture content in K-65 material. |
| 1988 | ● Calculated releases for 1980–1987.<br>● Measured radon concentrations in air around the K-65 silos. |
| All: radon decay product releases | ● Calculated radon releases.<br>● Radon decay product equilibrium fraction in headspace air.<br>● Fractional release factor for radon decay products. |

[a] Uncertainty has been accounted for in all parameters of the radon source term calculations. Uncertainties in the parameters marked (✈) contribute the most to the overall uncertainty of radon release quantities.

**Recent followup of issues related to radon releases.** Given the importance of doses from radon releases, it was reasonable to review radon related aspects of the project. The Centers for Disease Control and Prevention (CDC) held a Radon Review Meeting in December 1995. Peer reviewers and staff from the CDC and the *RAC* Team attended and discussed radon source terms and air dispersion modeling for radon and decay products emanating from the K-65 silos. Also, with the CDC's initiative, CDC and *RAC* staff performed additional document searches at the Fernald site to help ensure that all important data or documentation related to the radon source terms has been located and evaluated for use on the Project. The document searches located some data that had not been previously evaluated, and as a result some of the calculations related to the radon source term have been revised. Appendix Q of this report describes the additional document searches, changes to the calculations, and other issues related to radon releases.

In addition, since the release of the Task 2/3 report (Voillequé et al. 1995), we noticed that an important detail was neglected in the calculation of the radon release for 1988 from the K-65 silos. The radon source term for 1988 has been revised; Appendix Q of this report describes the revision. This change affects only 1988 and is not significant to the overall releases.

Finally, in a letter related to its review of a draft of this Task 6 report, the National Research Council's (NRC) Committee on an Assessment of CDC Radiation Studies accepted the preferred model of radon release from the K-65 silos as credible but requested that this final report contain a simplified step-by-step summary of the model. The committee also requested that uncertainties for the 1951–1979 releases be adjusted, that the relationship between the equilibrium ratios of decay products in the headspace and the pre-1979 gamma exposures be elaborated, and that the air exchange mechanism between the silo headspaces and the atmosphere before the domes were sealed be investigated further. The last request, concerning the air exchange mechanism, was considered infeasible in the current project. The disposition of the remaining requests is reported in the Addendum to this volume and in Appendix Q of Volume II.

**Radon source term results.** Based on these methods, we estimated that the historical annual radon releases from the K-65 silos were the greatest for 1959–1979, with a median release rate of 6200 curies per year (Ci $y^{-1}$). Figure 16 compares the radon releases for the different periods. After 1979, when the silo penetrations had been sealed, the radon release rate decreased significantly. Radon releases from the drums of material stored near Plant 1 were insignificant contributors to the total radon releases for the period 1951–1988, but they were important contributors to the releases during 1951 and 1952.

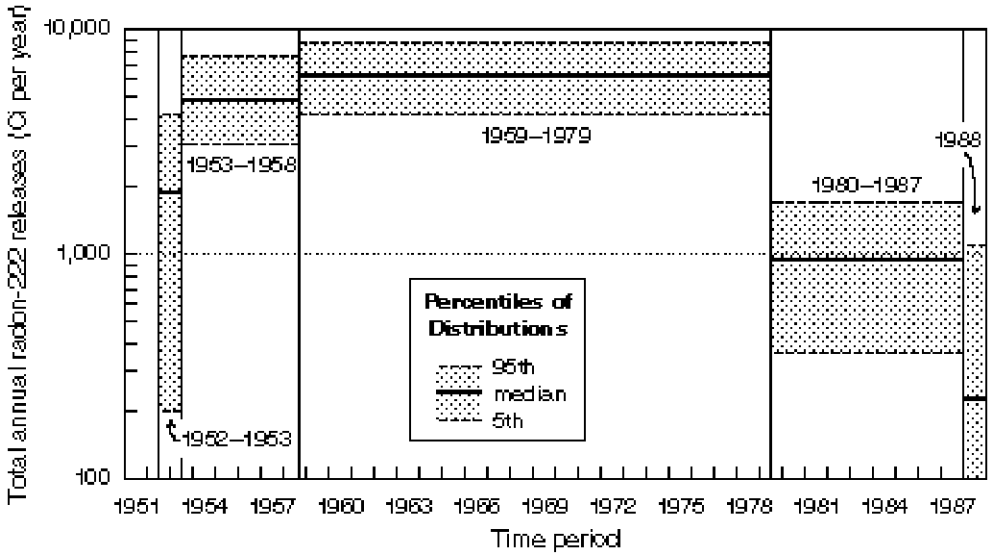



**Figure 16.** Changes in the estimated radon releases rates from the K-65 silos with time. The gray band shows the 90% probability interval. The solid centerline represents the median release rate, bounded by the 95th percentile (upper dashed line) and 5th percentile (lower dotted line). The 1959–1979 period is the most important with a very high release rate for a long period of time. The sealing of penetrations of the silo domes in 1979 resulted in a significant decline in radon emissions. The addition of an exterior foam layer on the silo domes in 1987 further reduced the emissions of radon. Note the logarithmic scale for annual releases.

The predicted total quantities of radon released from the FMPC for the entire period, 1951–1988, are summarized in Table 9. Radon releases from the drummed K-65 material stored on the Plant 1 pad are relatively small contributors to the total. We also calculated radon releases from the K-65 silos using the conventional method (described in Appendix J of the Task 2/3 report, with a brief review and revisions described in Appendix Q of this report). To compare the results of the preferred and conventional methods on the same basis, we calculated the radon release rate that would occur if the K-65 material was not covered by the silos; we called the result of this calculation an "unconstrained release rate." The unconstrained release rate is an upper bound on actual releases because the silos inhibit, or constrain, release of radon into the atmosphere. We have also calculated a "concentration-based minimum," which is a minimum unconstrained release that would have to exist to support the radon concentration measured in the silo head space in 1987 (see Appendix Q for details). Because the concentration-based minimum is based on the measured radon concentration, it is consistent with the preferred method calculations of the unconstrained release. But, the concentration-based minimum is independent of the conventional method calculations, and so can provide an additional comparison.

**The total amount of radon released from the K-65 silos for 1951–1988 is estimated to be in the range 110,000–230,000 Ci. Over 90% of the total was released before 1980.**

**Table 9. Summary of Estimated Radon and Radon Decay Product Release Quantities (Ci) from the FMPC for the period 1951–1988[a]**

| | 5th | median | 95th |
|---|---|---|---|
| *Radon-222 released from each source:* | | | |
| K-65 silos | 77,000 | 160,000 | 340,000 |
| Drummed K-65 material stored on Plant 1 pad | 54 | 730 | 3,600 |
| Both sources | 78,000 | 160,000 | 340,000 |
| *Total radon decay products released:* | | | |
| Polonium-218 | 74,500 | 156,000 | 327,000 |
| Lead-214 | 65,600 | 138,000 | 289,000 |
| Bismuth-214[b] | 59,500 | 126,000 | 265,000 |

[a] The median is our central estimate of the release. The 5th and 95th percentiles define a 90% probability interval.

[b] An equal amount of polonium-214 is assumed, because bismuth-214 and polonium-214 would have been in a kinetic state known as *secular equilibrium*.

The calculated unconstrained radon release rates are compared in Figure 17. The central estimates of the conventional and preferred methods are similar. However, the uncertainty in the conventional calculation results is much greater. The 90% probability interval (5th to 95th percentiles) of the preferred estimates lies within the 50% probability interval (25th to 75th percentiles) of the estimates made by the conventional method. And, more than 25% of

the distribution of results of the conventional method estimates are less than the concentration-based minimum, and thus inconsistent with the measured radon concentration in the silo head space. From these comparisons, we conclude that the results of the conventional method should be considered less reliable than results of the preferred method. However, the overlap in uncertainty ranges supports the conclusion that the conventional method results provide some corroboration of the preferred method results. Additional discussion of these comparisons is provided in Appendix Q of this report (Volume II).

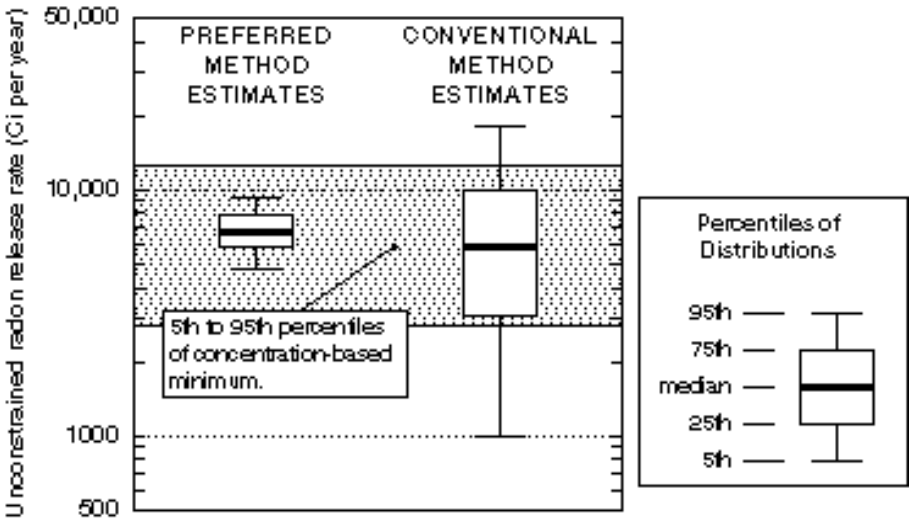



**Figure 17.** Comparison of estimated unconstrained radon releases from the K-65 silos using three methods: (1) our preferred method (our primary calculations) (2) the conventional method (used as a check), and (3) a concentration-based minimum unconstrained release rate. The unconstrained radon release rate is the release rate estimated to occur if the K-65 material was open to the atmosphere. The concentration-based minimum release rate is the minimum required to sustain the radon concentration in the silo headspaces (this concentration was measured in 1987). The band shown is the estimated 90% probability interval (5th to 95th percentile band) of this minimum. The conventional method estimates that are less than the concentration-based minimum are inconsistent with the measured head space radon concentration. Note logarithmic scale.

Table 10 compares our results with previous estimates of the emissions of radon from the K-65 silos. This comparison of release rates is only for a relatively short period and during the lowest releases. The other studies did not report uncertainties associated with the release rate estimates.

**Table 10. Comparisons of Our Preferred Method Estimates of Radon Release Rates (Ci $y^{-1}$) from K-65 Silos to Release Rates from Other Studies**

| | Percentiles of our estimates | | | Results of other studies | |
|---|---|---|---|---|---|
| Period, release pathway | 5th | Median | 95th | Value | Reference |
| 1980–1987, diffusion | 56 | 140 | 320 | 60 [a] | Borak 1985, IT 1989 |
| 1980–1987, air exchange | 250 | 740 | 2000 | 1023 [a] | IT 1989 |
| 1980–1987, total | 340 | 880 | 2400 | 1083 [a] | IT 1989 |
| 1988, total | 36 | 220 | 1300 | 1150 [b] | Hamilton et al. 1993 |

[a] These results were considered by IT (1989) to apply to the complete period 1953–1984, but we believe that the conditions and parameters used to develop the estimates were only valid for the period July 1979–1987.

[b] This result was the average release rate calculated for 1989–1990. We compare it to our results for 1988 because we believe conditions of the silos were unchanged for 1988–1991.

### Gamma Radiation Emitted from the Waste Storage Silos

Waste materials stored in the K-65 silos (Silos 1 and 2) and the metal oxide silo (Silo 3) emit gamma radiation, which is a source of direct radiation exposure of people around the FMPC. Direct radiation exposure refers to exposures of people outside the FMPC boundary to radiation from radioactive sources remaining on the FMPC site.

In Appendix J of the Task 2/3 report (Voillequé et al. 1995) we determined that these three silos were the only significant sources of such direct radiation exposures. Appendix J of the Task 2/3 report also provides detailed information characterizing the radiation sources of the silos. Calculations of exposure rate due to the radiation emitted from the silos are described in Appendix G of this report (Volume II). When compared to radon releases and particulate uranium releases, direct radiation emitted from the silos is a relatively insignificant source of doses to people around the site (see Appendices G and K of this report).

## RELEASES TO SURFACE WATER

There were three main sources of liquid waste releases from the FMPC: (1) process water via the clearwell portion of the waste pit, (2) sanitary sewage, and (3) runoff (storm sewer) water. The discharged radionuclides in wastewater were either in solution or in suspension as finely divided particles. Figure 18 shows that liquid effluent streams from FMPC were released to the offsite environment at two locations: (1) the combined effluent, which discharged through Manhole 175 into the Great Miami River at a point almost directly east of the plant site, about 1.2 mi (2 km) downstream from the town of Ross and (2) the storm sewer outfall, which discharges into a branch of Paddy's Run Creek onsite (Figure 18).

**Uranium concentrations and volumes were measured regularly in liquid effluent released to the river and to the storm sewer outfall ditch, and these measurements were used to reconstruct these source terms.**

### Uranium Released to the Great Miami River and Paddy's Run Creek

The discharge flow of liquid waste from the FMPC to the Great Miami River was continuously measured, and a composite sample was collected and analyzed for uranium daily. We used these daily measurements to reconstruct releases to surface water (Voillequé et al. 1995). When specific information was not located for a particular month, we used an average value based on the other months in the same year. Sources of uncertainty for these estimates of uranium losses through Manhole 175 to the Great Miami River came primarily from the analytical errors in (a) measuring effluent flow and (b) sampling and measuring uranium concentrations in the water.

Discharges to Paddy's Run came from the storm sewer outfall ditch overflow and from runoff from the west side of the facility. Under normal operations, runoff water collected in the storm sewer system passed through the storm sewer lift station before discharging through Manhole 175 to the Great Miami River (Figure 18). Because the storm sewer lift station was not connected to production facilities, the uranium in runoff was assumed to be from ground deposition, leaks and spills (Ross 1972). When the capacity of the storm sewer lift station was reached, water overflowed through the outfall ditch to Paddy's Run. The volume of storm water that overflowed was related to rainfall amounts and patterns.

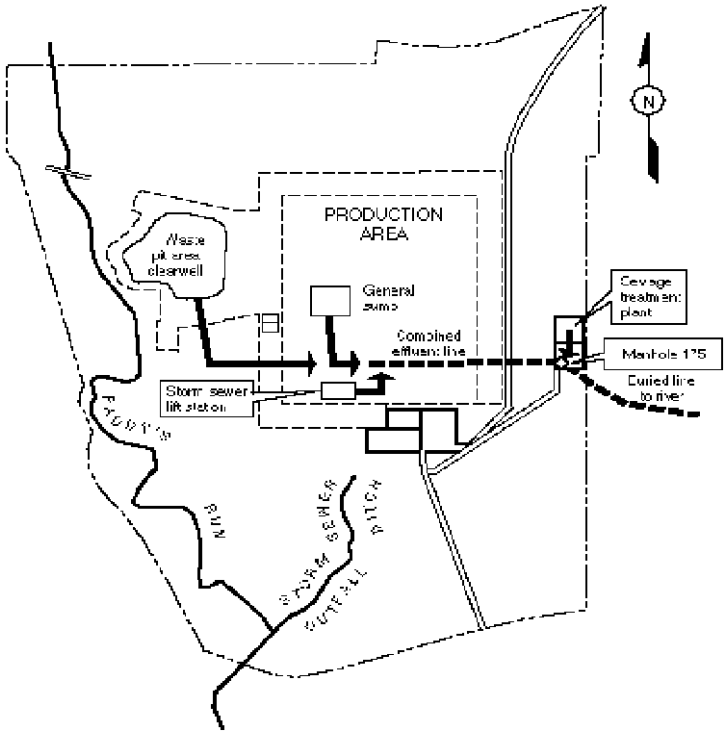



**Figure 18.** Liquid effluent flow and discharge points from the FMPC site.

We based estimates of uranium losses from the storm sewer outfall ditch to Paddy's Run on analytical data sheets and monthly reports that listed the individual overflow events. Three major components contribute to the uncertainty associated with estimating uranium losses to Paddy's Run. One component is the analytical errors associated with determining the uranium concentration and the water flow rate before discharge to Paddy's Run. The second component of uncertainty involves time periods when rainfall, and consequently ru-

noff, were quite high and the capacity of the storm sewer lift station flow meter at Paddy's Run was exceeded. The third component involves unmeasured losses to Paddy's Run Creek north of its confluence with the storm sewer outfall ditch. The weir and sampling location were in the outfall ditch above the confluence with Paddy's Run Creek.

Figure 19 shows the annual uranium release estimates to the Great Miami River and to Paddy's Run Creek for all years in the period 1952–1988. The magnitude of the uranium releases to the river peaked in 1961 with 7300 kg uranium. From 1974 to 1988, the annual releases were below 1000 kg. The uranium losses to Paddy's Run show much more month-to-month variation than do the uranium loss estimates to Manhole 175. During the early 1960s, however, the average quantity of 500 kg uranium discharged through Manhole 175 to the Great Miami River each month was five times greater than the average quantity of 100 kg of uranium lost to Paddy's Run.

With the exception of the early 1950s, there is generally more uncertainty in the release estimates to Paddy's Run than to the Great Miami River because the frequency of sampling at the storm sewer outfall ditch was less than at Manhole 175, and there were unmonitored discharges to the creek. Nevertheless, estimates of uranium releases in liquid discharges to surface water are relatively well known, and uncertainties are generally smaller compared to releases to air.

The chemical form of uranium in liquid effluents is not known with certainty. Several chemical forms of uranium may have been present in solution in liquid waste streams during this period. Uranium solubility is highly dependent on its valence state (+4 and +6), with the +6 valence state being much more soluble. The relative solubility of uranium in liquid releases depends upon the pH and level of suspended solids in the liquid wastes. Daily measurements of total suspended solids and pH were made on 24-hour composite effluent samples at Manhole 175 beginning in 1956 (NLCO 1956).

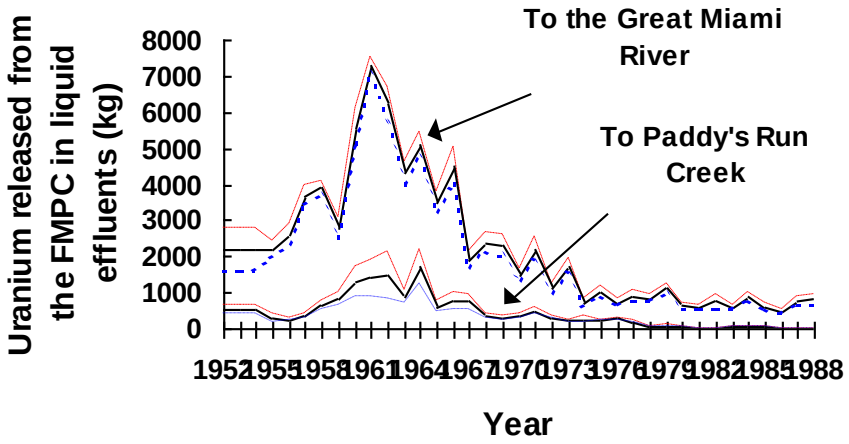



**Figure 19**. Uranium losses to the Great Miami River through Manhole 175 and to Paddy's Run Creek from the FMPC from 1952–1988. The 95th percentile (top, broken line) and the 5th percentile (lower, dotted line) describe the uncertainty of each estimate.

**Other Radionuclides Released in Liquid Effluents**

Other radionuclides released at various times over the years include decay, fission, and activation products of uranium, thorium, and recycled uranium. As discussed earlier in this report (see *Overview of FMPC Operations and Production History*, page 7), uranium, radon, and radon decay products clearly dominate the potential dose from atmospheric releases. However, other radionuclides are important for surface water pathways. Releases of thorium and one of its decay products, radium-226, occurred when thorium was processed at the site in 1954–1957 and 1964–1988. Relative concentrations of thorium with respect to uranium were measured in the mid-1950s and again beginning in 1967. Beginning in 1976, the concentrations of plutonium, neptunium, radium and the fission products, cesium-137, ruthenium-rhodium-106, technetium-99, and strontium-90 were measured in liquid effluents to the Great Miami River.

We based our release estimates for thorium, radium-226, radium-228, and fission and activation products to surface water on ratios of the releases of these radionuclides to the releases of uranium. These ratios, computed for years when measurements were made, provide a basis for estimating the release of the other radionuclides for years when they were not measured. Annual average concentrations of radium, thorium, and the fission and activation products in liquid effluents were reported by the FMPC in historic release reports (Boback et al. 1987) and in annual environmental monitoring reports beginning in 1976. We considered the variability of the release ratio from year to year in deriving the uncertainty associated with the estimated releases of these other radionuclides. Table 11 summarizes our release estimates for radionuclides in liquid effluents from the FMPC for 1952–1988.

**Table 11. Summary of Total Estimates of Radioactive Materials Released from the FMPC in Liquid Effluents For 1952–1988**

| Material released to the Great Miami River[a] | Median value or 50th percentile | Uncertainty range (5th to 95th percentile) |
|---|---|---|
| | Quantity (kg) | Quantity (kg) |
| Uranium | 82,000 | 71,000 to 94,000 |
| Uranium (to Paddy's Run Creek) | 17,000 | 14,000 to 20,000 |
| Thorium | 5,800 | 3800 to 9400 |
| | Activity (Ci) | Activity (Ci) |
| Radium-228 | 2.7 | 0.33 to 20 |
| Radium-226 | 18 | 15 to 22 |
| Plutonium-239, 240 | 0.0088 | 0.0019 to 0.033 |
| Plutonium-238 | 0.00028 | 0.00016 to 0.0034 |
| Neptunium-237 | 0.0044 | 0.0011 to 0.018 |
| Cesium-137 | 0.54 | 0.14 to 1.9 |
| Ruthenium-106 | 0.056 | 0.014 to 0.22 |
| Technetium-99 | 300 | 110 to 800 |
| Strontium-90 | 6.0 | 1.5 to 24 |

[a] Except as noted

Our median estimate of uranium released to the Great Miami River for all years is 82,000 kg. The 5th to 95th percentile uncertainty range is 71,000–94,000 kg. Some estimates of uranium in liquid wastes have been made by others annually (Boback 1971) or in summary reports evaluating the past discharge history of the facility (Rathgens 1974, Boback et al. 1985). These estimates of uranium to surface water from 1951 through 1984 range from 74,000–77,000 kg (Boback et al. 1987, Galper 1988) and fall within the uncertainty range of our estimates. Releases of radium-226 occurred throughout the history of the site. The total release estimate for radium-226 is 18 Ci (uncertainty range 15 to 22 Ci).

The total median release estimate for uranium to Paddy's Run Creek via the storm sewer outfall ditch and runoff is 17,000 kg of uranium, which is 20% of the direct releases to the river. The 5th to 95th percentile uncertainty range is 14,000–20,000 kg of uranium.

**RELEASES TO GROUNDWATER**

Uranium contamination of groundwater outside the FMPC has been known since late 1981, when the first samples of water from private wells were analyzed. The most notable offsite uranium *contamination* in groundwater is south of the site; it is referred to as the *South Plume* (Figure 20). The principal source of uranium contamination in the South Plume is the historical liquid effluent releases to Paddy's Run Creek and to the storm sewer outfall ditch (DOE 1990). There are additional known areas of groundwater contamination on the FMPC site, but only the South Plume area extends beyond the site boundary. An evaluation of the groundwater plumes underlying the FMPC at the end of 1988 indicated that three offsite wells were contaminated (DOE 1990).

Results of monthly uranium measurements in contaminated well number 15 for 1982 through 1988 were used as the basis for calculating radiation doses to a hypothetical individual for these years. (Well 15 had the highest measured uranium concentrations of the monitored offsite wells.) For the period before 1981, during which the wells were not monitored, we developed an *empirical model* to estimate uranium concentrations in the contaminated wells. This empirical model is based on uranium measurement data in the well for 1982–1992 and on the calculated quantities of uranium released to Paddy's Run and to the storm sewer outfall ditch during 1952–1988. It also depends on transit time ranges developed by previous groundwater studies. The model estimates the uranium in well 15 during each of the years before it was monitored. Details of the model are described in Appendix M in the Task 2/3 report (Voillequé et al. 1995). Appendix R of this Task 6 report (Volume II) uses a slightly different approach to reconstruct a history of contamination in well 15, along with uncertainty estimates. The results of the two approaches are similar. The reconstruction described in Appendix R was used to estimate levels of uranium in the kidneys of an individual who regularly drank water from well 15 for 38 years. Table 12 shows the estimates of uranium concentrations in well water from the South Plume, which we used for dose calculations. Groundwater model results suggest that uranium contamination in the groundwater did not reach the offsite wells before 1968 (Voillequé et al. 1995).

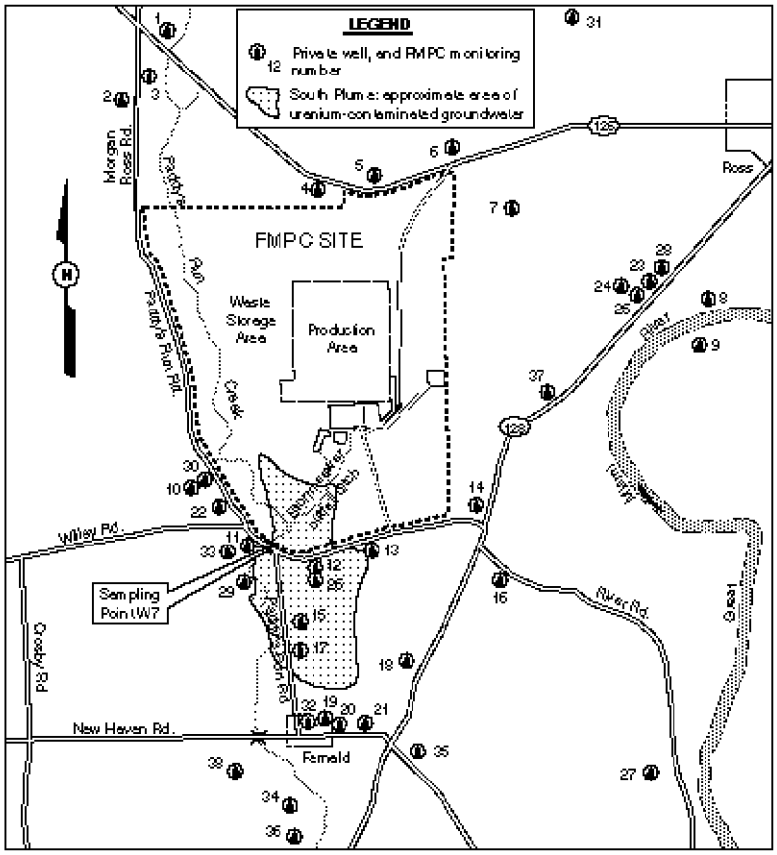



**Figure 20.** Approximate area of uranium contamination in the South Plume by 1991, and locations of the private wells around the FMPC sampled in the FMPC routine monitoring program. Sampling Point W7 samples the surface water in Paddy's Run Creek at the Wiley Road bridge.

**Table 12. Estimates of Annual Average Uranium Concentrations in Well Water from the South Plume (pCi $L^{-1}$)**

| Year | Concentration | Year | Concentration | Year | Concentration |
|---|---|---|---|---|---|
| 1951–1967[a] | 0 | 1975 | 490 | 1983 | 290 |
| 1968 | 180 | 1976 | 580 | 1984 | 220 |
| 1969 | 230 | 1977 | 620 | 1985 | 200 |
| 1970 | 230 | 1978 | 620 | 1986 | 190 |
| 1971 | 230 | 1979 | 570 | 1987 | 200 |
| 1972 | 240 | 1980 | 510 | 1988 | 190 |
| 1973 | 290 | 1981 | 460 | | |
| 1974 | 370 | 1982 | 320 | | |

[a] The concentration is applied to each year in this range.

## TRANSPORT OF RADIONUCLIDES IN AIR

The third step in the process of determining the radiation doses to people near the FMPC is developing procedures for describing the transport of released radionuclides through environmental pathways (see Figure 2). This step relies on environmental transport models, by which we mean mathematical descriptions of the dispersion and distribution in the environment of the radionuclides released from the facility. Such models depend on mathematics, statistics, chemistry, biology, and other sciences, and they ordinarily have to be programmed for a computer. The models permit us to simulate the movement of radionuclides in the air and water around the FMPC.

The Task 4 report for this project (Killough et al. 1993) reviews meteorological data and models of processes that are related to predicting air concentrations of radionuclides released to the atmosphere from the FMPC. Some of these processes are deposition of airborne radioactivity on the ground and *resuspension* of previously deposited radioactive particles from the soil back into the air. Some of the air dispersion methods described in the Task 4 report have been modified in the light of subsequent information and experience, and those modifications are reflected in this Task 6 report and its appendices.

Two principal types of radioactive materials were released from the FMPC, each requiring a different approach to air dispersion modeling:

1. Uranium, thorium, and smaller amounts of decay products, fission and activation products, and transuranics (TRU) that were released to the air, principally as particulates, from rooftop stacks in the production area (referred to generically as “uranium”)
2. Radon-222 and its decay products (referred to generically as *radon*), which were released to the atmosphere from the K-65 silos west of the production area and, in the early 1950s, from the drums stored near Plant 1.

Calculating air concentrations at downwind *receptor locations*, however, requires consideration of a number of properties and processes, which include:

- Multiple release points within the FMPC production area
- Physical and chemical characteristics of released material
- Diffusion as a function of distance from the source
- Particle size distributions and dry and wet deposition of particulates on the ground
- Resuspension of previously deposited material
- Runoff and leaching of deposited material from the soil
- Decay of radon-222 and formation of decay products as the release moves downwind from the K-65 silos to a receptor location.

Appendix M gives an extended discussion of these properties and processes and refers to the Task 4 report (Killough et al. 1993) for additional details. We give a somewhat less detailed summary here. Figure 21 suggests many of the properties and processes that enter into atmospheric dispersion.

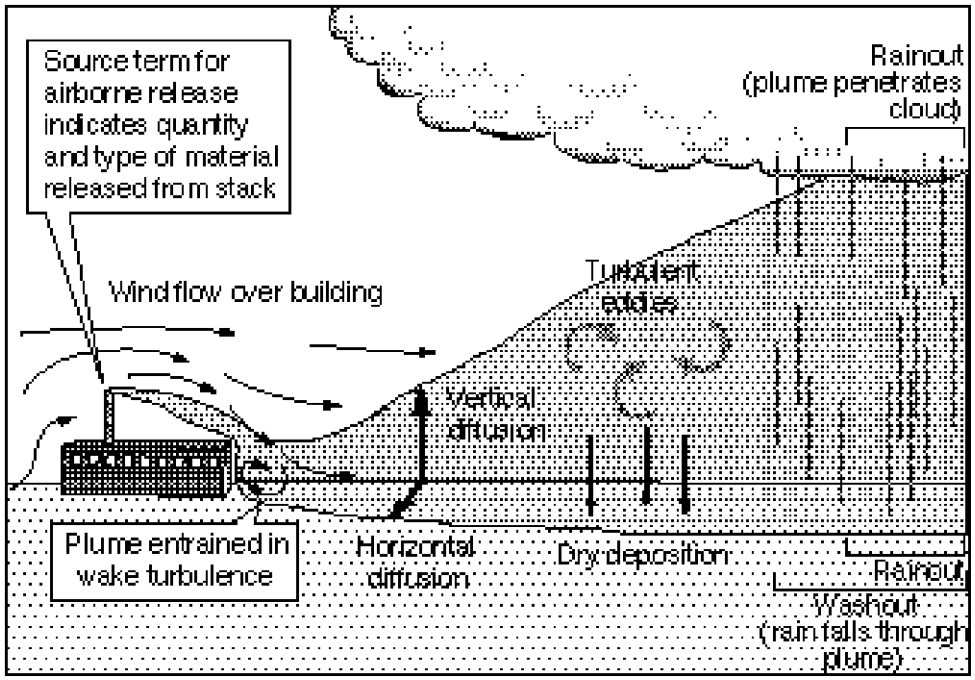



**Figure 21.** Physical processes associated with atmospheric dispersion. Diffusion disperses the plume horizontally and vertically, diluting its concentration as the released material moves downwind. Highest diffusion rates are associated with unstable air. Wet deposition consists of washout (plume is beneath the raincloud) and rainout (rain penetrates the plume), where the particles provide condensation nuclei for raindrops. Dry deposition is complex and consists of several stages. Turbulent eddies bring particles and gas molecules into contact with ground and vegetation surfaces. Near the ground, molecular diffusion accelerates deposition of very small particles. Gravitational fall dominates deposition for large particles.

**Dispersion as a Function of Distance from the Source**

The FMPC production area is a complex array of release sources with different characteristics that affect dispersion. Airflow over and among the numerous closely spaced clusters of buildings creates turbulence that sets up complicated dispersion and mixing patterns. Analysis of these release sources suggests that most releases from rooftop stacks at the FMPC would be drawn down into regions called *wake cavities* on the downwind sides of the buildings and would behave approximately like ground-level releases with enhanced vertical and horizontal dispersion (Killough et al. 1993, Appendix I). We have air monitoring data for uranium collected during the years 1986–1988 for various distances out to more than 3.7 mi (6 km) from the center of the FMPC production area. When these data are converted to estimates of diffusion, they give a coherent picture of atmospheric transport of the uranium released from the production area (Figure 22).

Diffusion at a receptor point is related to the pattern of air concentration that results from the horizontal and vertical spreading of the airborne material as it moves downwind from the source to the receptor. The diffusion curve shown in Figure 22 is based on a ground-level area-source *Gaussian plume model* for an area of approximately 0.25 $mi^2$ (0.64 $km^2$), which is approximately the area of the part of the production area where the release sources

are located. The plotted points and the solid curve in Figure 22 take into account depletion of the plume from deposition. Deposition is particularly important for modeling uranium released from the FMPC production area because of the large particles from the Plant 2/3 and Plant 8 scrubbers. These large particles are subject mainly to gravitational fall and have a higher probability of depositing near the point of release. Because particle-size distributions vary among release sources, we modeled more than 30 release components separately to account for their different deposition characteristics and properties.

We *calibrated* the air dispersion model shown as a curve in Figure 22 to air monitoring data from 1986–1988, which are represented by the plotted points. We applied the calibrated model to estimating air concentrations for other periods of time with different prevailing conditions, assuming that its use would be valid within certain adjustments and estimates of uncertainty. Figure 22 shows the 5th and 95th percentiles of the distributions of two kinds of uncertainty. The gray band around the model curve represents the first kind, which is uncertainty of the calibration process. The second kind of uncertainty, which is related to applying the model to our specific problem, is shown by the dashed curves above and below the model curve. This uncertainty is due to many factors that the model does not explicitly take into account, such as terrain irregularities and uneven heat exchange between air and soil at different locations in the assessment domain. The data for 1986–1988 were chosen for the calibration because in earlier years there were fewer sampling stations, and those stations did not represent the desired range of distances from the source. Also, the calibration depends on the release rates of uranium, and these release rates are more accurately known for the middle and late 1980s than for earlier periods.

The uncertainties shown in Figure 22 are propagated through the calculations of atmospheric transport of uranium from the FMPC production area. They are combined with other uncertainties in the dose estimates.

The second principal type of radioactive releases to air from the FMPC is radon gas and its decay products. The K-65 silos west of the production area present a simpler source for radon release than the buildings of the production area do for the release of uranium. However, the data that can be used to infer atmospheric diffusion of the released radon show somewhat greater scatter than the uranium data. Interpreting the radon air monitoring data is also more difficult because the release rate from the sealed K-65 silos during the period when the data were collected (1980–1988) was not the same for day and night. The data represent averages for 24 hours, and special mathematical techniques were required for calibrating the nighttime and daytime versions of the model to the 24-hour data. For periods before the 1980s, when the K-65 silos were vented to the atmosphere through penetrations in the domes, the release rates for day and night would have been approximately the same. However, for these earlier periods, there are few air monitoring data for radon and none that could be used to calibrate a model. An added complication in the analysis was estimating and subtracting the contribution of naturally occurring radon (sometimes called *background*) from the measured concentrations, so that our calibration would only account for radon that was released from FMPC storage sites.

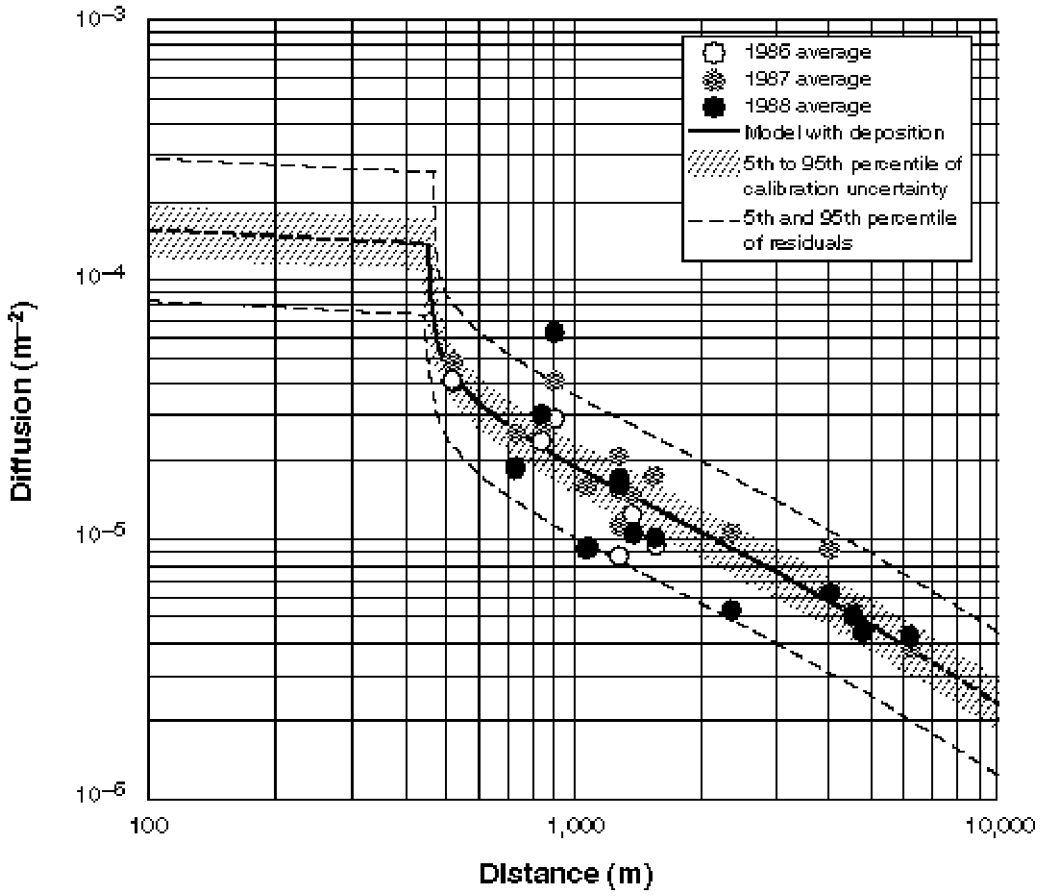



**Figure 22.** Uranium air monitoring data for 1986–1988 fitted by a Gaussian diffusion model for a ground-level circular area source with a radius of 450 m (area 0.64 $km^2$). The data reflect the reduction of air concentrations from deposition of uranium from the plume. The model simulates this reduction with a surface depletion method. The gray band represents uncertainty associated with model calibration, and the dashed curves above and below the model curve are the 5th and 95th percentiles of the residual distribution, which is interpreted as uncertainty from many factors that vary from one location to another, such as terrain irregularities and exchange of heat between the atmosphere and the soil.

Figure 23 shows the diffusion curve for releases of radon and decay products from the K-65 silos and a band indicating 5th and 95th percentiles of calibration uncertainty. The curve is based on a Gaussian diffusion model for a ground-level circular area source with a radius of 50 m. The calibration uncertainty band is wider than the band for the uranium diffusion curve (Figure 22) because of the greater scatter in the radon data. Predictions of radon diffusion beyond 2000 m are extrapolations and are subject to uncertainty that increases with the distance beyond the range of the data (shown by the widening of the gray 5th–95th percentile band beyond about 2000 m).

In a manner similar to that of uranium, the diffusion calibration uncertainties indicated by the gray band in Figure 23 are propagated through the calculations, combined with other uncertainties, and appear as a component of the uncertainty of each radon dose estimate.

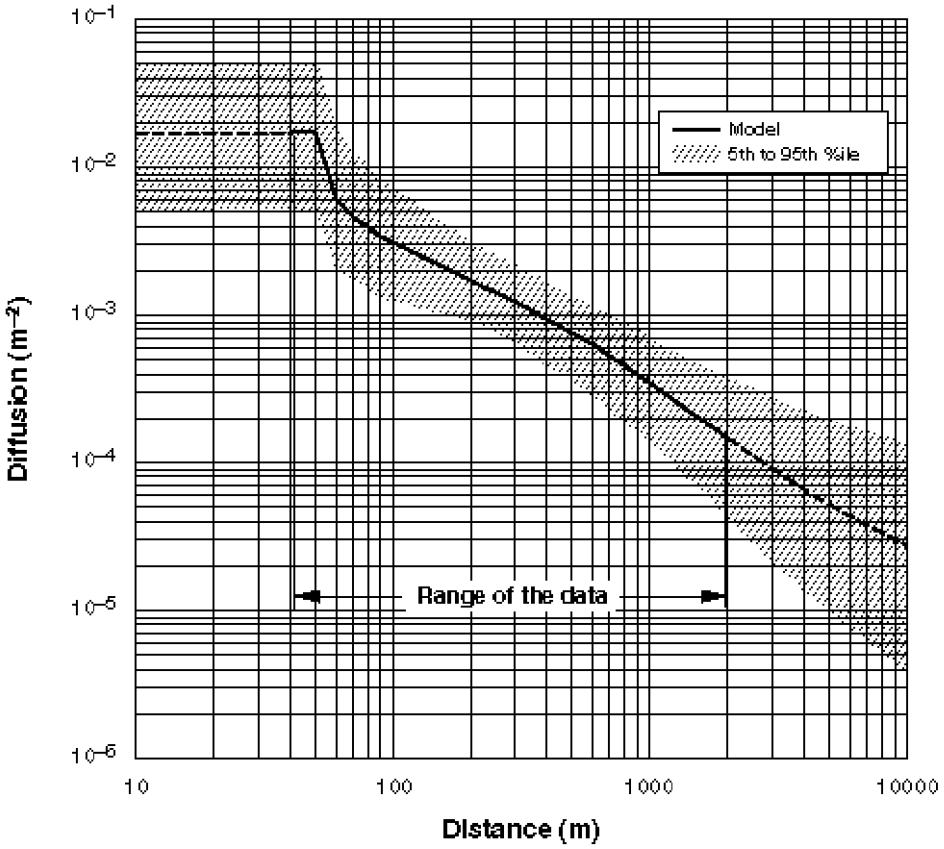



**Figure 23.** Diffusion as a function of distance for radon released from the K-65 silos. The curve is based on a Gaussian diffusion model for a ground-level circular area source with a radius of 50 m. The gray band represents uncertainty associated with calibration of the model to two sets of radon air monitoring data collected during the 1980s. This curve is an average of the daytime and nighttime calibration curves that are based on the 1980s radon monitoring data. It was used for the period before the K-65 silos were sealed in 1979.

### Physical and Chemical Characteristics of Release Sources

As noted previously, there are more than 30 release sources for uranium in the production area. For each source, the dose calculation takes into account the chemical and physical properties of the released radioactivity. Chemical properties include the type of chemical compound of a radionuclide, which is important for determining the rate of movement from the lungs or gastrointestinal tract into the blood and organs of the body. The biological retention time of a radionuclide by an organ depends on the chemical element represented by the radionuclide. Together with the radiological half-life, this retention time determines the amount of radionuclide remaining in the organ as a function of time. Calculations based on this information determine the dose to lungs, gastrointestinal tract, and specific organs.

The physical property of primary interest is the distribution of aerodynamic diameters (*ADs*) of particulate radionuclides that are inhaled. The principal regions of the respiratory tract are the nasal-pharynx, tracheo-bronchial region, and pulmonary region. Particles greater than about 20 micrometers (μm) in AD are considered non-respirable and contribute

negligibly to dose by inhalation. For smaller particles, the AD distribution determines the deposition pattern of the particles among the regions of the respiratory tract. This deposition pattern is important for calculating retention times of the radioactivity in the lungs and its movement to other organs.

**Particle Size Distributions and Dry Deposition**

The AD distribution of the particles in a release is also important for calculating concentrations of radionuclides in air and soil because of the strong relationship between dry deposition and particle diameter. Dry deposition also depends on wind speed (the higher the wind speed, the higher the dry *deposition velocity*); atmospheric stability; and the nature of the ground surface (for example, snow-covered, lawn, or tree-covered). The diffusion models shown in Figures 22 and 23 are averages over different atmospheric stability categories. Atmospheric stability is a parameter in the model, and the results of the model predictions are averaged with the frequencies of the six stabilities given by the FMPC meteorological data. This approach is particularly important for the uranium transport because the rate of deposition of the particulate material is determined by the product of a dry deposition velocity and the air concentration of the radionuclide near the ground. Each of these factors depends on the stability and wind speed near the ground. As we have indicated, the dry deposition velocity also depends on the AD of the particles. The calculation separates the particles released from each source into 13 discrete ADs and deposits the particles of each size at the appropriate rate for the prevailing wind speed and stability.

The only data for AD distributions of particles from the FMPC dust collectors were developed in a 1985 study by Northern Kentucky Environmental Services (*NKES*) (Reed 1985). In the NKES study, measurements were made for the inlet and outlet ducts of 15 major uranium-emitting stacks. Particle sizes for the outlet ducts (emission stacks) represent emissions from the stacks with intact bag filters. Particle sizes for the inlet ducts represent emissions from the same stacks when the bag filters failed and allowed unfiltered air from the production process to escape to the atmosphere. Aerodynamic diameter distributions for the stack emissions measured during the 1985 NKES study are assumed to apply to other periods because of the similarity of the processes over time. Either uranium tetrafluoride ($UF_4$) or uranium oxide ($U_3O_8$) was emitted from all of the stacks in the NKES study, except for one stack that emitted a mixture of uranium dioxide ($UO_2$) and uranium trioxide ($UO_3$). For processes at the FMPC with no particle size data available from the NKES study, we made comparisons with similar operations at other facilities carrying out similar processes. The median ADs for inlet ducts of the dust collectors are in the range 0.5–15 μm. For the outlet ducts, the range is 5–9 μm.

Releases of uranium through the scrubbers of Plant 2/3 and Plant 8 present a different situation than for the dust collectors. Some small particles (mostly less than 2 μm in diameter) passed through the scrubbers without being washed out of the gas stream. However, a substantial fraction of the uranium released to the atmosphere from all FMPC processing plants came from these scrubbers as large droplets of reentrained scrub liquor, which rapidly dried down into large particulate solids, with ADs in the range of 30–150 μm. As mentioned previously, when calculating concentrations of radionuclides in air and soil, large particles have the following important properties:

- The large particles are rapidly deposited from the plume and, therefore, do not have time to travel far downwind.
- Particles of the AD range resulting from the reentrainment are not respirable and have negligible dosimetric consequences for inhalation. They are considered, however, in calculating contamination of locally produced garden and animal food products.

Figure 24 summarizes the different groupings of dry deposition velocities on the basis of particle AD. The calculated values shown are averages. In the dose calculations, dependence of dry deposition velocity on atmospheric stability, wind speed, and AD is taken into account.

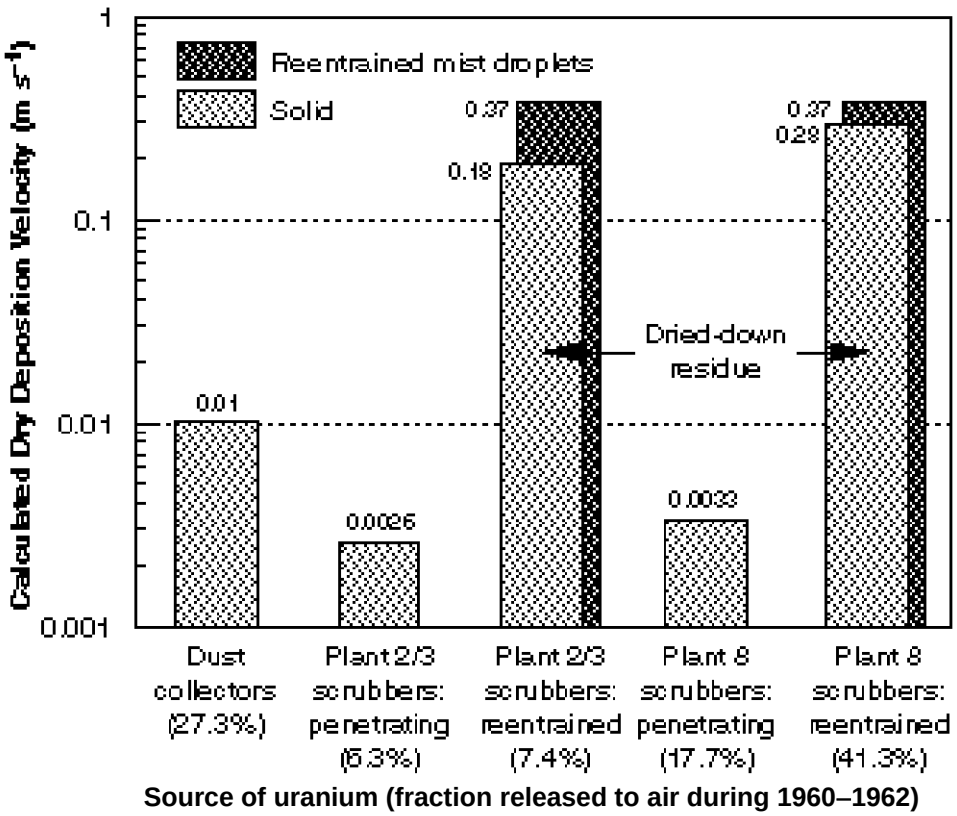



**Figure 24.** Dry deposition velocities (meters per second) calculated for FMPC releases of uranium. High rates of deposition are believed to be associated with uranium in large liquid droplets that escaped from the Plant 2/3 and Plant 8 scrubbers and quickly dry down to large-particle residues. Lower deposition velocities are associated with smaller particles that penetrated the scrubbers and from dust collector releases. The particles with low deposition velocities remain airborne longer and thus may be transported farther from the FMPC.

### Wet Deposition

The airborne plume moving from the FMPC is also affected by the process of wet deposition in which precipitation removes, or washes out, radioactive material from the air. The method we used to estimate this removal uses a *washout ratio*, which is experimentally obtained from the measured concentration of a tracer material in precipitation divided by the measured concentration of the material in the air. From this ratio and the precipitation rate,

a wet deposition velocity can be calculated (Killough et al. 1993, Appendix H). We were able to calculate a site-specific washout ratio for uranium in air at the FMPC based on uranium concentration measurements in precipitation and precipitation amounts. These measurements, taken during the years 1961–1967, are in good agreement with those reported in the literature. Mean annual precipitation recorded at the FMPC from 1966 through 1988 was 38.3 in (97.2 cm). Converting this value to m $s^{-1}$ and combining the result with the median washout ratio gives a wet deposition velocity 0.016 m $s^{-1}$ as an average value over time. The dose calculations use wet deposition velocities that are based on year-specific precipitation values; thus, the calculated wet deposition velocity changes from year to year.

**Building Wake Effects and Complex Source Region**

Calculations reported in the Task 4 report (Killough et al. 1993, Appendix J) strongly suggested that releases from a rooftop stack in the FMPC production area would be drawn down into a wake cavity at the downwind face of the building and that the release would subsequently behave much as if it had come from near the ground level, possibly with enhanced horizontal and vertical diffusion about the centerline of the plume. The success of the calibrated ground-level area-source model in representing the uranium air monitoring data (Figure 22) lends credibility to the wake cavity hypothesis. The empirical approach based on fitting the area source model to the 1986–1988 air monitoring data avoids some of the difficulties and added uncertainties of explicit modeling of complex airflows among the many buildings.

**Resuspension of Deposited Uranium Particles**

Airborne releases of uranium from the FMPC are eventually deposited on the ground surface through gravitational settling and wet and dry deposition. A subsequent transport mechanism for this deposited material is resuspension, which refers to the reentry of previously deposited particles into the air, where they are available for inhalation and redeposition.

Resuspension is of less concern in a climate like that of Ohio than in desert regions of the western United States, where much of the research on resuspension has been carried out. To estimate the air concentration of the contaminant, we used two parameters: (1) the *mass loading*, which is the concentration of dust in surface air and (2) the concentration of contaminant in the surface layer of soil.

To determine the mass loading, we averaged weekly measurements of the total suspended particulates measured in air at 14 air monitoring locations surrounding the FMPC for 1989 through 1991 to characterize the variations in particulate concentrations in air. These data were used to reconstruct the probable seasonal variation in dust loading. May through September are dustier than the annual average but are still within 30% of the annual average. The analysis also showed that the annual averages are quite consistent, and the 3-year average of 33.5 micrograms per cubic meter (μg $m^{-3}$) is in good agreement with the 1966 Air Surveillance Network average for nonurban locations of 38 μg $m^{-3}$ (Healy 1980). To apply the method described in Killough et al. (1993) to the entire time period of interest for the dose reconstruction, the particulate concentrations in air were compiled for 1971–

1988. We projected these data back to the 1950s and 1960s, when no measurements were available (Appendix D).

The second parameter in the mass loading method, the concentration of contaminant in the surface layer of soil, was based on simulations with the dynamic soil model, which is coupled to the air dispersion and deposition model.

**Dynamic Soil Model**

Levels of uranium and other radionuclides in the soil over time are simulated by a dynamic model of deposition and first-order removal of soluble and insoluble radionuclides (Appendix B). This model simulates a root-zone layer of 4-in (10-cm) depth and a thin surface layer of 0.2-in (0.5-cm) depth. The thin layer is assumed to be the source of resuspended radioactivity, and the resuspension model uses the concentration in this layer to determine the neighboring air concentration for the resuspended component. The root-zone layer furnishes concentrations of radionuclides to the GARDEN model to estimate root uptake by pasture grass and food crops. The time resolution of the soil model is annual. The soil model's parameters have been estimated from annual measurements of uranium in soil at six of the seven boundary stations since 1971.

**Atmospheric Transport of Radon-222 and Its Decay Products**

Predicting the air transport of radon and its decay products from the K-65 silos requires that the radon decay chain be simulated for the time of plume travel to each receptor location. Dosimetry for the radon decay chain is based on the model described in Report No. 78 of the NCRP (NCRP 1984). We refer to the model as the NCRP model. The NCRP model estimates the dose to the *tracheobronchial epithelium (TBE)* from the radon-222 decay products polonium-218 (radium A or RaA), lead-214 (RaB), and bismuth-214 (RaC). The shorter-lived decay product polonium-214 (RaC') is always present in the same radioactivity amount as RaC, and the NCRP model implicitly takes the RaC' dose into account. There are other decay products of radon-222, but their relative contribution to the tracheobronchial epithelium dose is considered negligible. The NCRP model further distinguishes between the unattached fraction of RaA that is airborne as free ions and the attached fraction that is attached to dust particles or other condensation nuclei. The unattached RaA contributes a higher dose to the tracheobronchial epithelium for each unit of inhaled radioactivity than does the attached fraction. Thus, the calculation of atmospheric transport must consider not only the decay chain kinetics, but also the migration of RaA from the unattached state to the attached state. This calculation depends on a rate constant for attachment of RaA that is estimated from the count of available condensation nuclei per unit volume of air. We give an example to illustrate these ideas.

Suppose the source of radon is releasing 1 picocurie of radon-222 per second (pCi $s^{-1}$) and that the wind speed is 2 meters per second (m $s^{-1}$) or about 4.5 miles per hour. For this example, we assume that no decay products are being released from the source. We may imagine a vertical slab of air 2 meters thick moving past the point of release in 1 second, and as it does, it would receive 1 picocurie of radon-222 from the source. We start the clock at the release point (time zero) and follow this slab of air as it continues to move downwind at 2 meters per second. The 1 picocurie of radon-222 undergoes radioactive decay, forming the decay products RaA, RaB, RaC, and RaC' described previously. Figure 25 shows the remaining

amount of the initial 1 picocurie of radon-222 and the net amount of each decay product in the moving slab of air for each subsequent time, up to 8,000 seconds (more than 2 hours). In 8,000 seconds, the slab of air would have moved about 10 miles (16 km) from the source if we assume that the wind speed remained constant. In this time, Figure 25 shows a small decrease in the amount of radon-222. The half-life of radon-222 is 3.8 days, which means that much time would be required for one-half of the initial 1 picocurie to decay. Each decay product will eventually reach a maximum level and begin to diminish. The unattached amount of RaA is plotted in Figure 25 as a dashed line, corresponding to a dust count of $3 \times 10^{10}$ particles per cubic meter of air ($m^{-3}$). This amount quickly comes into equilibrium with the available condensation nuclei (that is, the dashed line quickly reaches its maximum and becomes level).

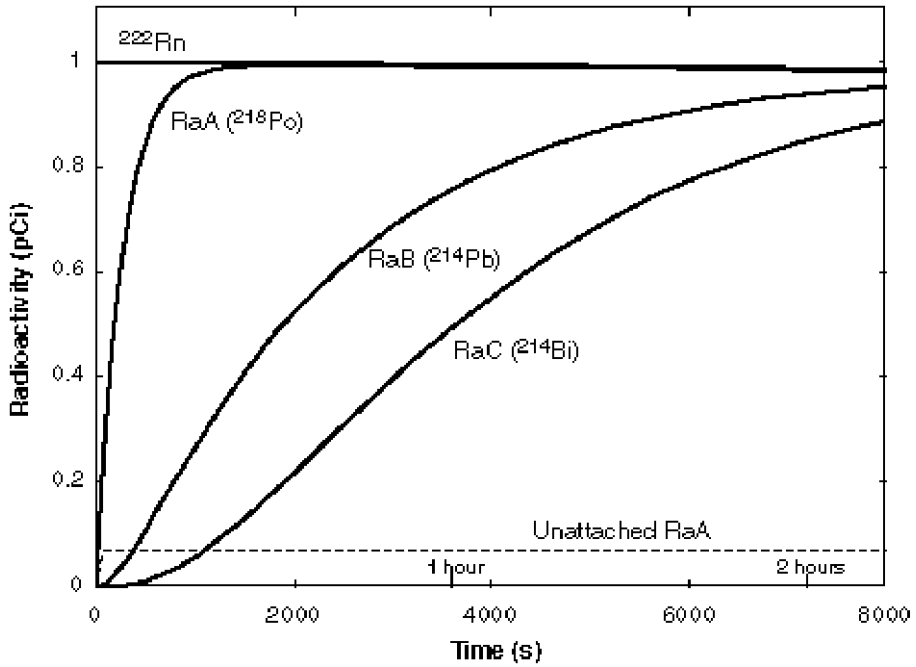



**Figure 25.** Decay of radon-222 and formation of decay products significant to dose with time. Initially present are 1 pCi of radon-222 and no decay products. Radium-A grows to equilibrium within one-half hour, and RaB and RaC form more slowly. Notice the very rapid equilibration of the unattached RaA, which for this illustration was based on a count of condensation nuclei of $3 \times 10^{10}$ per cubic meter ($m^{-3}$). To estimate air concentrations of the decay products, it is necessary to multiply each fractional level from the figure by $QfD/u$, where $Q$ is the release rate (picocurie per second), $f$ is the wind-directional frequency, $D$ is the diffusion (per square meter or $m^{-2}$; Figure 23), and $u$ is the wind speed (meter per second).

Within the moving slab of air, the radon-222 and each decay product diffuse vertically and laterally, reducing the ground-level concentration over time. The diffusion $D$ per square meter at any specified distance, $x$ meters, from the source comes from the curve in Figure 23. Multiplying the amount of radon-222 or any decay product in the slab when it is $x$ meters

from the source by the diffusion and dividing by the wind speed reduces the total amount in the slab to an air concentration at ground level. This concentration is further reduced when we multiply it by the fraction of the time the wind blows from the source toward the location we are interested in, such as a home or school. Finally, we must multiply by the actual measured or estimated release rate, $Q$ picocuries per second (we assumed 1 picocurie per second; the actual release rate may be more or less than this).

A distinction is made in the dose calculations between indoor and outdoor exposures to radon decay products. The calculation of indoor air concentration depends on the corresponding outdoor values and the rate of exchange of air between the building and the outdoors (usually measured in air changes per hour). Deposition of the decay products on indoor surfaces, called plateout, depletes the indoor air concentration of respirable radioactivity and reduces the dose. In buildings located near the release source, the reduction by plateout is offset by the relative stagnation of the indoor air, which gives the decay products more time to build up until they approach or possibly exceed outdoor levels, particularly RaB, RaC, and RaC'. Far from the source, however, outdoor concentration levels of decay products are closer to equilibrium with radon-222 (Figure 25), and plateout reduces the indoor concentrations below the levels of their outdoor counterparts. At most distances, the contrast between indoor and outdoor air concentrations is not great (Appendix I).

**Meteorological Data**

The atmospheric dispersion models used to reconstruct ground-level air concentrations at particular locations require knowledge of local meteorological data. At the FMPC, a meteorological tower has been in full operation only since August 1986. In 1992, when decisions on developing a meteorological database had to be made, the FMPC dataset included only about 5 years of recent data, with occasional gaps in that record. Thus, a problem existed for reconstructing air concentrations for earlier years of plant operation. Our principal options were to (a) use a composite of the recent FMPC tower data (1987–1991) and apply an appropriate compensating uncertainty to early air concentration estimates, or (b) substitute a surrogate dataset for the FMPC data.

We examined four sets of meteorological data from the region for their applicability to the dose reconstruction project. Information about these datasets is presented in Table 13. Only datasets taken at the Cincinnati and Dayton airports [19 mi (30 km) south-southeast and 37 mi (60 km) northeast of the FMPC, respectively] could be considered possible surrogates for the FMPC data because of the length of time these records were collected. The most useful comparison of the Cincinnati and Dayton datasets with the FMPC consisted of using data from the same period from each dataset with a standard Gaussian plume dispersion model to estimate air concentrations at a grid of receptor locations throughout the assessment domain. Pairs of results (for example, Cincinnati compared to FMPC) were plotted to indicate similarities and disparities of predicted air concentration between the two datasets. The distribution of the *predicted-to-observed ratios* (for example, Cincinnati/FMPC) measures the differences of the datasets. Figure 26 shows the logarithmic plot for the Cincinnati and FMPC comparison.

**Table 13. Meteorological Datasets Applicable to the Dose Reconstruction Proect**

| Meteorological data | Period | Location | Applicability to FMPC |
|---|---|---|---|
| FMPC tower | Aug. 1986–Dec. 1991 | Onsite | Judged best. Site-specific, of short duration, with some gaps |
| Cincinnati Airport | Jan. 1948–present | 19 mi (30 km) SE | Continuous record; probable underprediction of air concentrations |
| Dayton Airport | Jan. 1987–Dec. 1990 examined | 37 mi (60 km) NE | Long record available; probable underprediction of air concentrations |
| Oxford, Ohio | Jan. 1981–Dec. 1990 | 19 mi (30 km) N | Short duration, atypical terrain, difficult to determine stabilities |

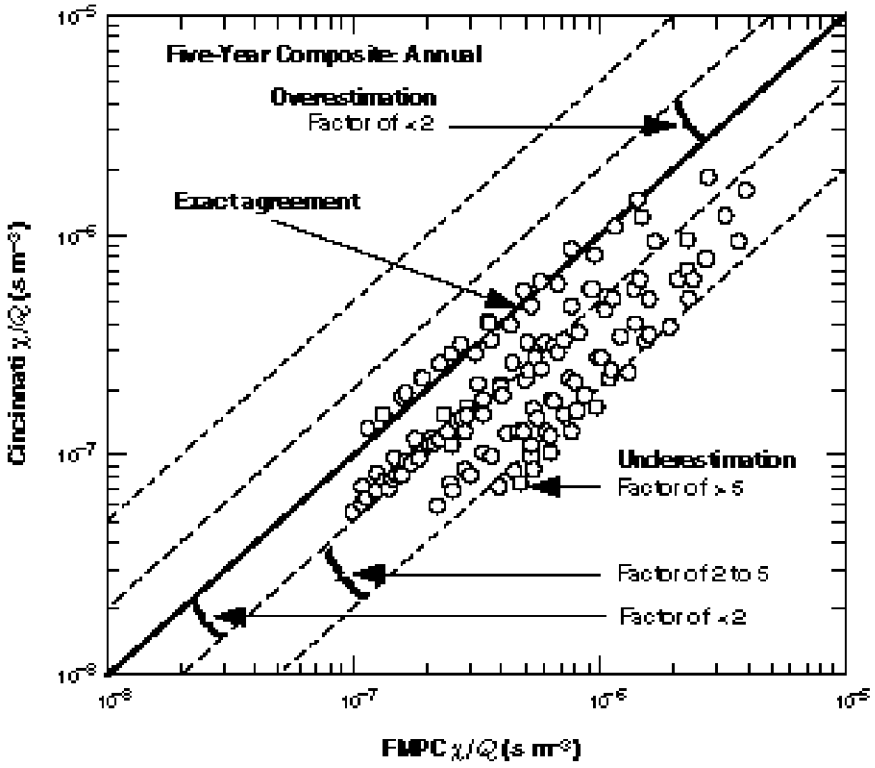



**Figure 26.** A comparison of 1987–1991 composite annual-average datasets for the Cincinnati- Northern Kentucky International Airport and the FMPC site. Normalized estimates of concentration ($\chi/Q$, s $m^{-3}$) predicted for each dataset with a Gaussian plume model are plotted against each other at each of 128 points of a grid that covers most of the assessment domain [out to 6.2 mi. (10 km)]. The regions between the lines indicate the extent of over- or underestimation of FMPC air concentrations by the Cincinnati data. In this comparison, most points correspond to underestimation by factors of 2 to 5.

The discrepancies between the FMPC and the Cincinnati and Dayton airport datasets involve differences in distributions of wind speed, wind direction, and atmospheric stability. In particular, airport wind speeds, on average, were about twice as high as wind speeds for the FMPC. Because of the reciprocal relationship between wind speed and predicted air concen-

tration, this disparity in wind speeds would correspond to estimates of higher air concentrations and generally higher doses if the FMPC site data were used. Three criticisms have been raised against the FMPC site data:

1. The brief duration of the record (5 years at the time the database for this project was compiled) makes this dataset inadequate to estimate air concentrations in the years before 1987.
2. There are numerous gaps in the 5-year record, which are the result of lightning, maintenance downtime, and other operational problems associated with the tower instrumentation and computer system that recorded the data.
3. The FMPC tower is poorly sited and may not represent local and regional wind patterns. In particular, it has been suggested that a pine tree plantation located on the FMPC property west of the tower would bias wind speed estimates. A 1987 survey by a National Oceanic and Atmospheric Administration (*NOAA*) team identified drainage flows through Paddy's Run Creek and other complex wind patterns that indicated a need for additional towers on the site to support real-time plume travel projections if there was an operational accident. However, plant operations ended in 1988, and presumably no action was taken on the recommendations of the NOAA team.

There is some validity in each of these criticisms of the FMPC dataset, and we have sought remedies to the extent we believe they are possible. The following paragraphs explain our responses to the criticisms.

The brief duration of the record is a severe limitation. An approach that has been suggested is to establish a correlation between the FMPC dataset and the same 5 years of data from an alternative site with a long record, such as Cincinnati. The correlation would be applied to earlier years of Cincinnati data to estimate the corresponding values for the FMPC. The wind speed, wind direction, and stability at the FMPC site would be predicted for any hour by using the same three variables for Cincinnati, whose values are known for the specified hour. We approached the correlation in a simpler fashion, which also makes use of the Cincinnati data. In our approach, the 5-year composite dataset for the FMPC is used to estimate meteorological frequencies for periods before 1987, but an uncertainty is applied to the estimates of air concentration. This uncertainty distribution is based on how well Cincinnati data for 1987–1991 estimate earlier air concentrations as calculated by Cincinnati data specific to the earlier times. It should be clearly understood that this method does not substitute Cincinnati data for FMPC data; rather, it relies on the assumption that the uncertainty of using 1987–1991 data to predict the past has the same distribution, whichever of the two datasets is used.

**We identified four sets of meteorological data that might be used to describe the local wind conditions. The FMPC dataset was the most representative of local conditions. For example, the mean wind speeds at the Cincinnati airport are about 2 times higher than those measured at the FMPC. Higher wind speeds generally result in lower downwind air concentrations of released materials.**

The gaps in the FMPC record are, of course, worrisome. After examining lists of their occurrences, we concluded that the gaps were sufficiently random to justify ignoring them in the analysis. We are not persuaded that attempts to fill in the gaps by interpolations or similar devices would add credibility to the dataset, and we chose not to do so.

The siting of the meteorological tower is an appropriate concern, but one for which no clear remedy presents itself. The NOAA survey team took 21 days of observations in November 1987 using 14 portable towers. The data indicated possibly significant differences between the FMPC site and each of the two nearby airports [Cincinnati and Dayton, about 19 and 37 mi (30 and 60 km) distant, respectively]. Doubts were raised about the suitability of the existing FMPC tower to support real-time forecasts of plume travel in emergency situations, but the very different question of suitability of the tower data for this dose reconstruction study was obviously not addressed. The NOAA survey identified drainage flows down Paddy's Run Creek and episodes of diverging wind directions between the valley floor and nearby ridgetops, but it could neither quantify frequencies of these phenomena nor associate them with longer-term estimates of air concentrations over a wide area. The survey report did not mention the pine tree plantation west of the tower, but this issue was raised in a review by the Fernald Environmental Management Corporation of the Task 4 report. There is, of course, the possibility that these trees affected the wind speed at the tower during the recording period (1987–1991). However, in examining land use maps of the region, we found that about one-third of the area of the assessment domain was forested, and several bands of forest cut across the northeast quadrant. The effectiveness of the windbreaks afforded by these natural growths may have been comparable to the pine trees west of the tower. We are led to conclude that any attenuation of predicted wind speed caused by the pine tree plantation may not represent a distortion for the assessment domain as a whole. In any case, we judge that the FMPC dataset is a better representation of the assessment domain than data from the Cincinnati airport, which is situated in a flat open space (as airports are) on a bluff south of the Ohio River. The authenticity of the FMPC tower dataset cannot be easily dismissed, despite its manifest shortcomings. Accepting its possible but unknown biases seems preferable to making adjustments that cannot be verified or substituting a surrogate dataset from a distant sampling station (for example, Cincinnati). Although they provide no absolute guarantees, our calibrations and comparisons of simulated environmental concentrations with measurements furnish a level of protection against gross errors in the meteorological database, the models, and other data.

## TRANSPORT OF RADIONUCLIDES IN WATER

Surface water transport of radionuclides represents an environmental exposure pathway for the dispersion of radionuclides from the FMPC. Radioactive materials, such as uranium, that are released from the FMPC to surface water, are dispersed and diluted in the water and transported by the current. They may be deposited onto sediments on the bottom of the Great Miami River, where they enter the aquatic food chain, or they may be taken up directly from the water by fish and humans. Exposure pathways are potential routes through which people may be exposed to radionuclides or radiation. The pathways are defined depending on the ways that people could be exposed at a certain time (Figure 27).

Initial screening calculations (Voillequé et al. 1995), along with site-specific irrigation and sediment sampling data, indicated that the drinking water and ingestion pathways were key surface water pathways for exposure to radionuclides from the FMPC. As a result, we chose a simple dilution and transport model to assess the transport and dispersion of ra-

dioactive materials in surface water. This method accounted for dilution and transport of the material in the Great Miami River and Paddy's Run Creek. The dilution factor is based upon the flow characteristics of the Great Miami River and FMPC discharge volumes for the site. These parameters have been studied extensively over the years (IT 1988, USGS 1991) and adjusted for the river in the vicinity of the FMPC by the Miami Conservancy District.

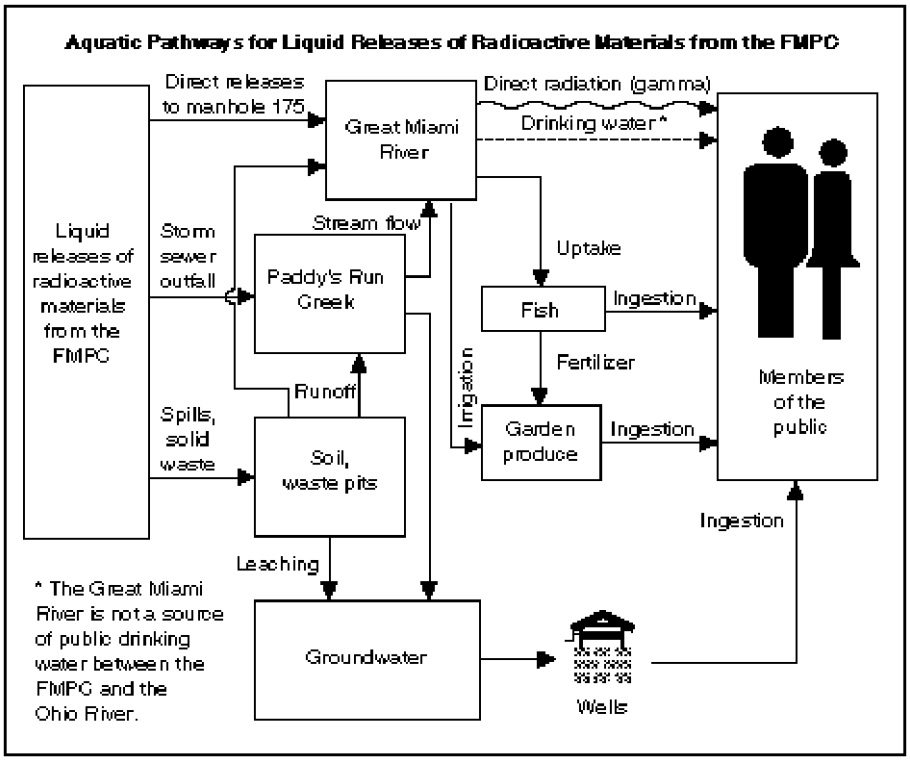



**Figure 27**. Possible aquatic pathways for releases of radionuclides in liquid effluents from the FMPC.

Figure 28 shows the relative magnitude of the river flow and the liquid effluent from the FMPC to the Great Miami River and to Paddy's Run Creek. Figure 28a compares the liquid effluent flow rate from the FMPC for a period in the 1960s with the flow that U.S. Geological Survey (USGS) measured in the Great Miami River. Although the river flow rate varies seasonally between 300 and 30,000 million gallons per day, the FMPC effluent discharge rate during this time was fairly constant month to month at about 1 million gallons per day. The final concentration of the radionuclides in the river varies inversely with flow rate. As the river flow rate increases, the dilution of the radioactive material increases, and concentration of the radionuclides decreases. The volume of liquid effluent discharged into the Great Miami River gradually declined from approximately 1 million gallons per day in the late 1950s and early 1960s to about 0.5 million gallons per day by the late 1970s (Figure 28b). The volume of effluent released to Paddy's Run Creek was much less, ordinarily 10 times lower, than to the river (Figure 28b).

The simple dilution model for our calculations incorporated the Crystal Ball™ uncertainty analysis program (Decisioneering 1993) to define the distribution of the annual average uranium concentration after dilution in the river, using the measured values of uranium in

the discharged effluent and the river and effluent flow rates. The annual average uranium concentration forecast is described by the median estimate with a 5th to 95th percentile range.

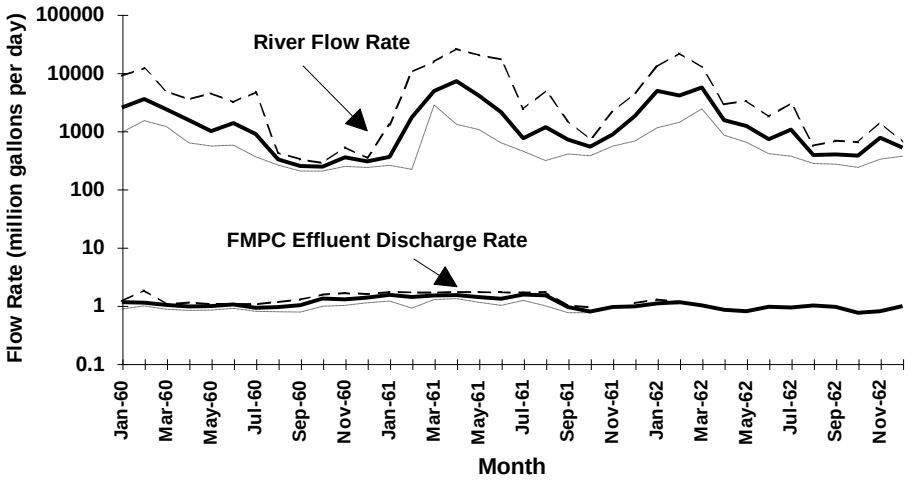



**Figure 28a**. Comparison of flow rate in the Great Miami River with the FMPC effluent flow to the river. The average flow is shown as the center black line, with the high and low flow shown above and below, respectively.

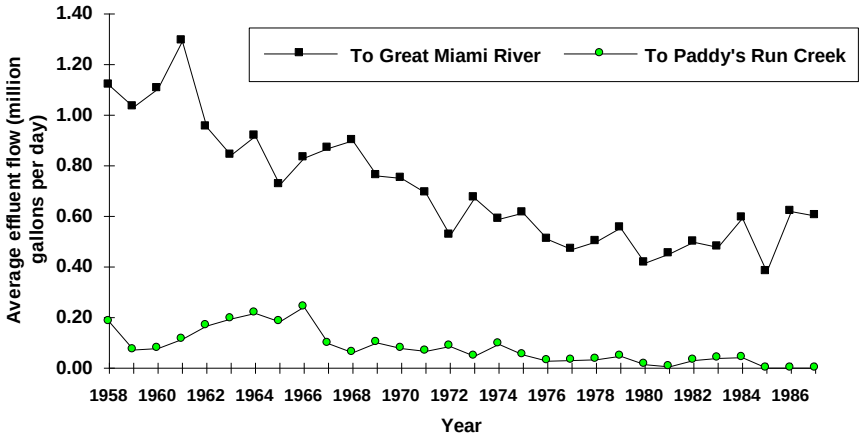



**Figure 28b.** Comparison of FMPC effluent flow to the river and the creek.

## COMPARING ENVIRONMENTAL MEASUREMENTS WITH PREDICTIONS

### Introduction and Concepts

Comparing measurements of a material like uranium in the environment with predicted concentrations of that material is sometimes called *validation* of the models or methods used. In contrast to prospective assessments for future or hypothetical situations, dose reconstruction involves historical events. Consequently, the methods developed for dose reconstruction studies can be checked against environmental monitoring measurements that were made during the time periods of interest.

In this study, predictions of environmental concentrations for various times and places in the past are obtained by reconstructing the releases from the FMPC facilities (source term) and the environmental transport of those releases (Figure 29). The source term is developed from effluent monitoring data, production information, and other calculations. Using our environmental transport model, the source term estimates (releases) are translated into predicted environmental concentrations around the facility. The dosimetry model relates these predicted environmental concentrations to the dose people may have received. Comparing environmental measurements with predicted environmental concentrations is one way we can gain confidence in the methods used to calculate dose. When the source term is well known, the validation is mainly checking how well the environmental model works. However, when the source term is not well known, a combination of the source term and the environmental transport model is being checked.

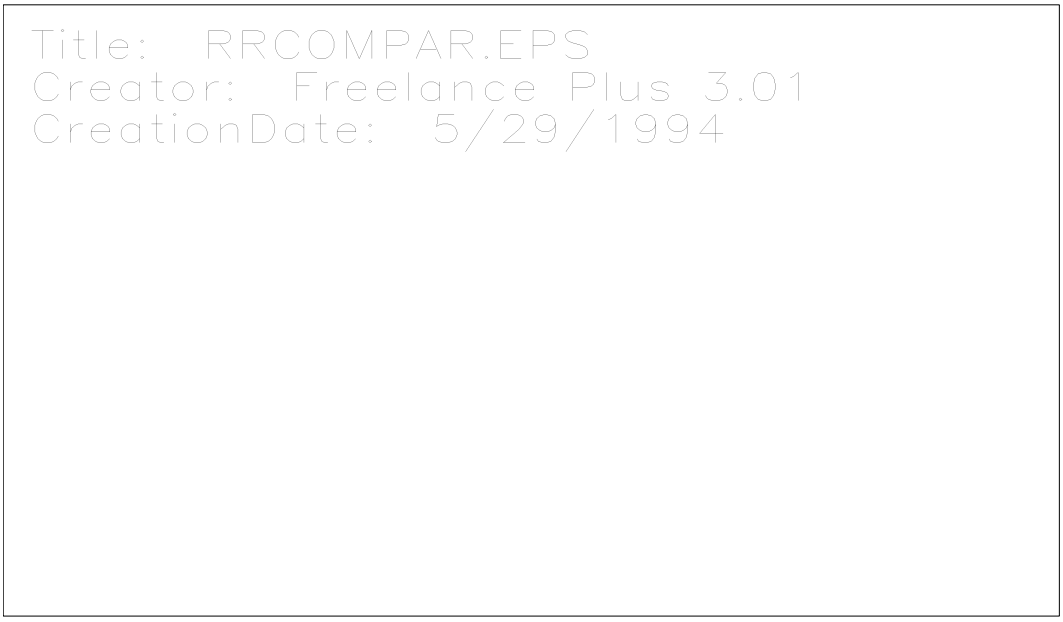



**Figure 29.** Validation is the comparison of measurements of environmental concentrations (observations) with predicted concentrations. Environmental measurements may also be used to calibrate or refine the transport model so it produces more accurate predictions. In this study, uranium air monitoring data from 1986–1988 were used for calibration and data from earlier years were used for validation.

There are several important reasons why models are needed in this study even when environmental measurements are available:

- Measurements are not available at all times or all places of interest
- Quality of data in early years (sensitivity, accuracy, and precision) is probably not as good as in later years (or it is difficult to assess)
- Particle-size and chemical form information are not available from routine monitoring, and these greatly influence the dose received by people
- At relatively distant locations, it is hard to distinguish site-released materials from other sources.

Shleien et al. (1995) reviewed the historical environmental measurements available for validation and other uses in this study. Unfortunately, there was very little information for validation of radon releases from the K–65 silos during the period of highest releases. Most of the environmental monitoring during the site's early history was focused on uranium.

There are several general ways that we compare predicted and observed concentrations. The first is a simple visual examination of the trends of both concentrations over time (a time trend) or space (a distance trend). It is useful, however, to quantify the difference between the model-predicted and measured concentrations (*model bias*). Bias can be expressed in terms of a predicted-to-observed ratio (P/O ratio), where:

| | |
|---|---|
| $P/O = 1$ | Exact agreement |
| $P/O > 1$ | Overprediction |
| $P/O < 1$ | Underprediction. |

When a number of years or locations are being examined, the *geometric bias* is used in this study, which is the geometric mean of the individual P/O ratios, given by

$$\text{Geometric bias} = \exp\left[\frac{\sum_i \ln(P_i / O_i)}{n}\right] \quad (1)$$

where:

$P_i$ = predicted concentration at location or time i
$O_i$ = observed concentrations at location or time i
$n$ = number of locations or times being compared.

There are numerous uncertainties both in the models and the measurements in any validation study. If predicted concentrations are within a factor of 2 of available measurements (geometric bias of 0.5 to 2), the agreement is generally considered good.

Another measure of the association between a series of observations and predictions is the *correlation coefficient*. This statistic ranges from –1 to +1. Relatively large values for the correlation coefficient are obtained when the concentrations are positively correlated, that is, locations or times having large predicted concentrations had large observed values. Large negative correlation coefficients would be obtained if large predicted values were associated with small measured values and vice versa. If the predicted and observed concentrations are unrelated, the correlation coefficient would be near zero.

**Summary of Treatment and Use of Air Monitoring Data (Uranium)**

The historic monitoring of uranium (U) in air around the Fernald site has proved very useful to the reconstruction project. Occasional monitoring began in the mid-1950s, but a routine program suitable for validation was not in place until 1958 (Table 14). Shleien et al. (1995) presents detailed information about the history of the FMPC monitoring program. The locations of the air monitoring stations on the FMPC perimeter and boundary are shown in Figure 30. The perimeter stations were dismantled and replaced with the boundary stations in 1972. The air monitoring program was expanded after 1986 to include stations both farther from and closer to the FMPC. Data from this expanded program in 1986–1988 were used to calibrate our air dispersion model for transport of uranium and to evaluate the uncertainty in predictions made with the model (Appendix M).

Model predictions of environmental concentrations and doses in the FDRP are based on a time resolution of 1 year. Therefore, the validation exercises, in which predicted and observed measurements are compared, are also performed using data averaged over a period of 1 year. The annual average air concentrations derived for this study are based on a thorough analysis of original data sheets of air monitoring results that were collected weekly. Appendix N contains the details concerning the production of average concentrations, data completeness, and a listing of those concentrations for each year and location. Plots of monthly average concentrations, presented in Shleien et al. (1995), illustrate the variation in the measured concentrations over shorter time intervals.

**Historical environmental monitoring data are not complete enough to rely on exclusively for assessing exposures to the public. The data can be used to check that the modeling methods are producing reasonable results for the times and locations for which they are available.**

**Table 14. Extent of Air Monitoring for Uranium at the FMPC Perimeter or Boundary During Different Time Periods**

| Time period (location) | Percentage of year covered by air monitoring | Months per year when no sampling occurred |
|---|---|---|
| 1953–1957 (perimeter) | <1 | 6–11 |
| 1958–1960 (perimeter) | 15-19 | 4–5 |
| 1961–1971 (perimeter) | 28–48 | 0 |
| 1972–1988[a] (boundary) | Continuous | 0 |

[a] Ending date for this study.

In Appendix L of Killough et al. (1993), we presented a method to adjust the measured uranium concentrations because the air samplers did not collect all particle sizes efficiently. A collection efficiency was determined for each air monitoring station and year; on average, efficiencies were about 75%. The details of the final determinations of air sampler collection efficiency are discussed in Appendix L of Volume II of this report.

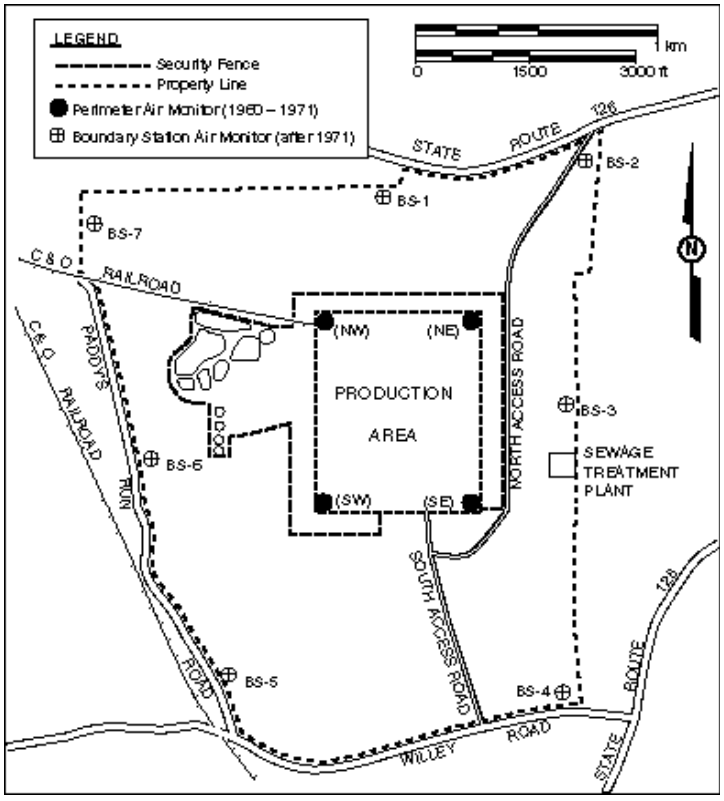



**Figure 30**. Ambient air sampling locations around the FMPC.

After correction for sampler collection efficiency, the historical measurements of uranium in air were compared to predicted concentrations at the same location and time. On the original laboratory data sheets, uranium measurements were recorded in both mass and radioactivity units. The activity concentrations presented in this report refer to total uranium. Concentrations of uranium–238 only would be one-half of these total uranium values. We chose to use the activity concentration unit of femtocuries per cubic meter (fCi $m^{-3}$) of air. This allows the concentrations to be plotted and visualized without a scientific exponent. One femtocurie (fCi) is $1 \times 10^{-15}$ Ci, or 0.001 picocurie (pCi).

### Validation Results — Uranium in Air

Detailed tables of predicted and observed measurements of uranium in air are included in Appendix N. Table 15 summarizes the results. The geometric bias at both the perimeter and boundary stations is 1.0, which is excellent overall agreement. The average correlation between the predicted and observed concentrations at perimeter stations is over 0.65, but at four of the boundary stations it is less (Table 15). The lower correlation at the boundary stations could be partially due to the smaller range of concentrations there. In general, concentrations are overpredicted to the east of the site and underpredicted to the north and west. An exception is boundary station BS–6, to the west, which has a geometric bias of 1.3.

**Table 15. Comparison of Predicted (P) and Observed (O) Uranium Concentrations in Air at Monitoring Stations**

| Monitoring Station [a] | Distance from FMPC center (km) | Correlation between ln(P) and ln(O) | Geometric bias | Long-term average predicted concentration (fCi $m^{-3}$) | Long-term average observed concentration (fCi $m^{-3}$) |
|---|---|---|---|---|---|
| **Perimeter (1958-1971)** | | | | | |
| SW | 0.5 | 0.65 | 0.57 | 150 | 220 |
| NW | 0.5 | 0.86 | 0.64 | 82 | 110 |
| NE | 0.5 | 0.85 | 1.86 | 440 | 200 |
| SE | 0.5 | 0.79 | 1.42 | 260 | 150 |
| **Boundary (1972-1988)** | | | | | |
| BS–1 | 0.9 | 0.62 | 0.64 | 13 | 17 |
| BS–2 | 1.3 | 0.72 | 1.57 | 26 | 13 |
| BS–3 | 0.7 | 0.50 | 1.46 | 37 | 20 |
| BS–4 | 1.4 | 0.30 | 1.28 | 11 | 6.4 |
| BS–5 | 1.3 | 0.18 | 0.81 | 8.6 | 8.2 |
| BS–6 | 1.1 | 0.67 | 1.30 | 23 | 12 |
| BS–7 | 1.6 | 0.90 | 0.52 | 4.2 [b] | 8.9 [b] |
| **Perimeter Group** | | | **1.0** | | |
| **Boundary Group** | | | **1.0** | | |

[a] See Figure 30.
[b] For years when observations were available (see Appendix N).

Subsequent charts in this section illustrate time trends. To permit a long-term summary view, the comparison of predictions with measurements of uranium in air in four directions are plotted in Figures 31 and 32. There was no routine monitoring of uranium in air in the mid-1950s when uranium releases from the FMPC were highest. Data from routine monitoring were available for comparison beginning in 1958 at the four perimeter locations. In 1972, air monitoring began at the seven boundary stations. Each plot in Figures 31 and 32 contains the monitoring data record from one perimeter and one boundary station, as shown in the figure titles. There is a clear decrease in uranium concentration over time, consistent with decreasing releases from FMPC facilities. Predicted and observed concentrations generally track each other quite well.

Natural background concentrations of uranium, mainly due to particles of soil that are suspended in the air, were much less than the uranium concentrations in air historically measured near the FMPC. For this reason, there was no need to subtract background concentrations from the observations before making comparisons to the predicted concentrations. The background concentration line in Figure 31 (and subsequent plots) is based on air monitoring data collected by the Environmental Protection Agency in Columbus, Ohio.

The approximate detection limit for the analysis of uranium in air at the FMPC is shown on the figures as a dashed line. For the earlier time period, we determined the detection limit by examining the analytical data sheets. Later, the detection limit was reported in the annual environmental reports. It is obvious that the method used in the 1960s was not adequate to measure typical background concentrations. However, the method was adequate to determine compliance with the applicable Atomic Energy Commission standard at the time of 2000 fCi $m^{-3}$ for uranium in uncontrolled areas. The monitored concentrations were almost always above detection limits. The main reason for the improvement in the detection limit in the early 1970s was a change from noncontinuous monitoring of air (about 56 hours per week) at the perimeter to continuous monitoring at the boundary. This resulted in a larger volume of air being sampled and consequently more uranium was collected.

**A typical annual average concentration of uranium in air at the perimeter of the FMPC in the 1960s was over 1000 times the natural background concentration of uranium in air.**

Predictions at the northeast and southeast perimeter stations tend to exceed measurements, especially before 1965. There is good agreement at the perimeter between 1965 and 1971. There is also overprediction at the northeast and southeast boundary stations in the 1970s and good agreement in the 1980s (Figure 31). To the west of the site, there is better agreement between predicted and observed concentrations at the perimeter before 1965, whereas observations exceed predictions in those directions between 1965 and 1971 (Figure 32).

Although they were not in place during the period of highest releases of uranium, the boundary air monitoring stations provide important data for validating our environmental transport methods at distances similar to those where the public were exposed. Appendix N contains detailed tables of the comparisons between predicted concentrations and observed measurements at the boundary air monitoring stations from 1972 through 1988. As indicated previously (Table 15), the agreement between predicted and observed concentrations of uranium in air at the boundary stations is good, with an overall geometric bias of 1.0 for all boundary stations. Figure 33 illustrates the time trend in predicted and observed uranium concentrations in air at two of the seven boundary stations for 1951 through 1988. The highest predicted concentrations of uranium in air are in 1955, when the releases were highest. The plots for all seven locations are included in Appendix N.

An average set of wind conditions, based on FMPC measurements for 1987–1991, was used for all years of the dose reconstruction (Appendix M). Therefore, actual weather data for a specific year are not incorporated into the predicted concentrations for that year. For the recent years in which wind data are available, our predications might be more accurate if the actual wind data for each specific year were used. However, the retrospective nature of the study emphasizes the years when releases were highest.

Boundary station BS–3 is the one showing the highest predicted and observed uranium concentrations in air, primarily due to its proximity to the production area and the predominant frequency of winds blowing from the west (Figure 29). It is also near the old solid waste incinerator, an area that became quite contaminated with uranium because of handling of ash residues (Appendix K, Voillequé et al. 1995). Although operations were discontinued at

the old solid waste incinerator at the end of 1979, BS–3 continued to show the highest uranium concentrations of the boundary stations. This suggests that either proximity to the production area or resuspension of contaminated soil around the incinerator is likely to be a more significant contributor to airborne uranium at BS–3 than stack releases from incinerator operations.

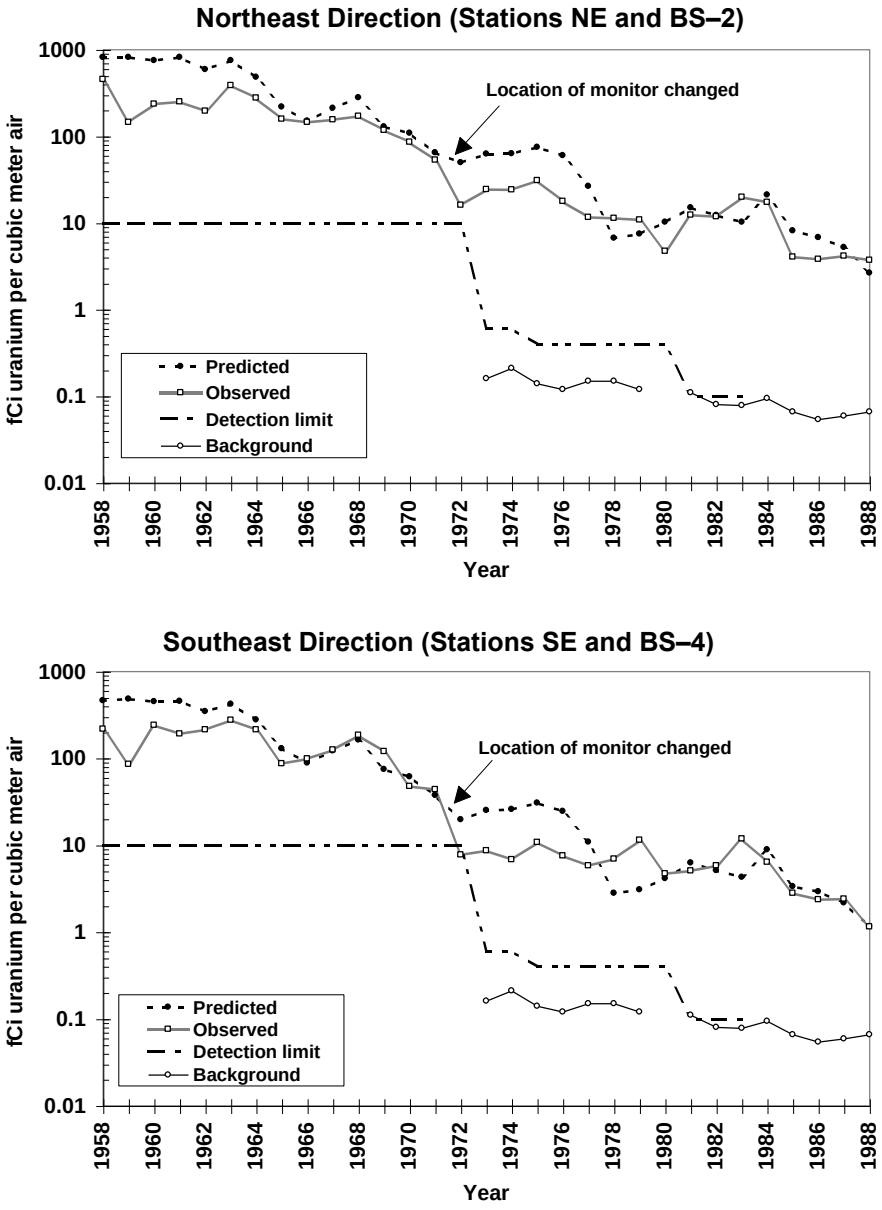



**Figure 31.** Time trend in observed and predicted concentrations of uranium in air at the perimeter and boundary stations northeast and southeast of the FMPC from 1958–1988. See Figure 30 for sampler locations. The observed measurements have been corrected for air sampler efficiency. The predicted concentrations result from our reconstructed source term and dispersion model.

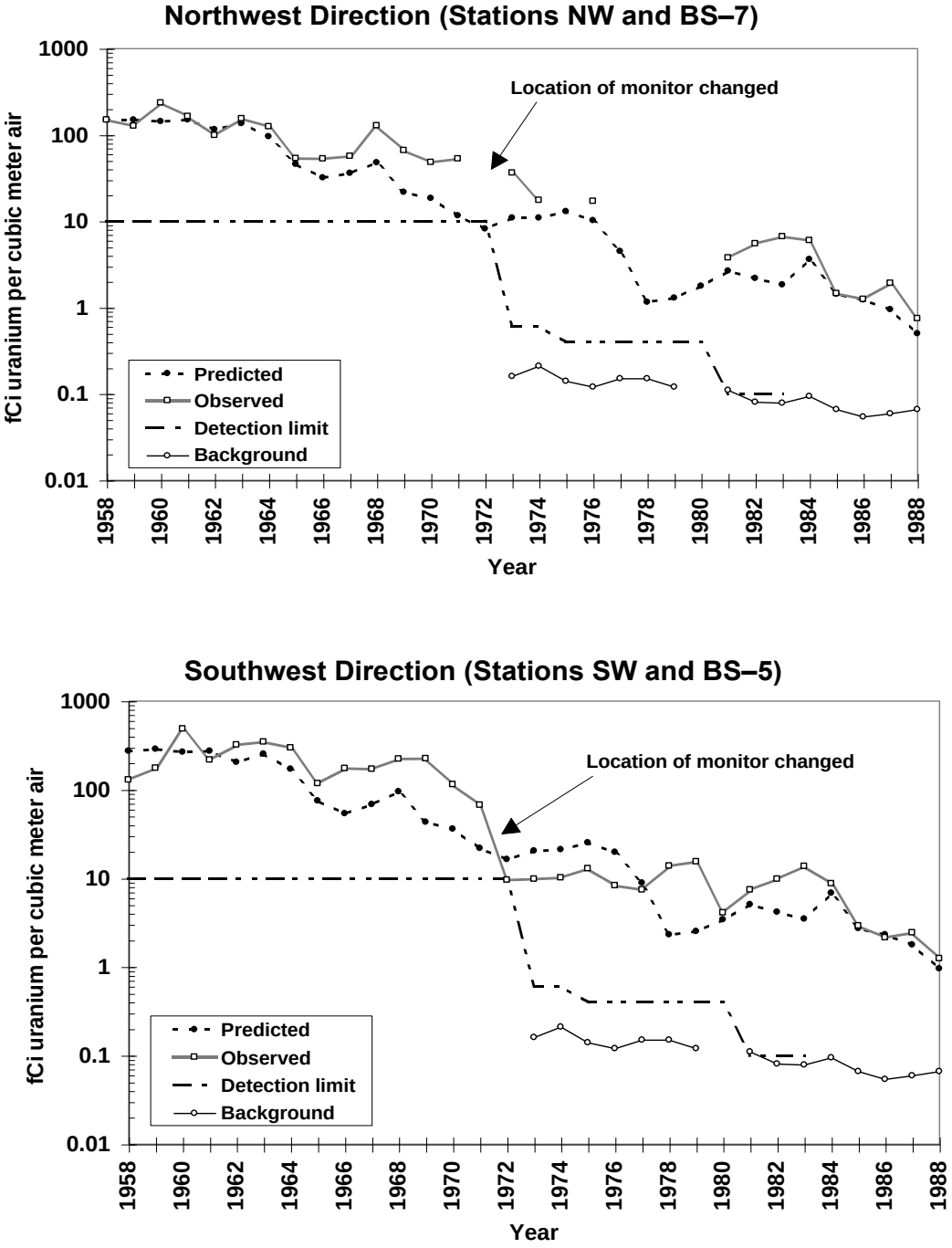



**Figure 32.** Time trend in observed and predicted concentrations of uranium in air at the perimeter and boundary stations northwest and southwest of the FMPC from 1958–1988. See Figure 30 for sampler locations. The observed measurements have been corrected for air sampler efficiency. The predicted concentrations result from our reconstructed source term and dispersion model.

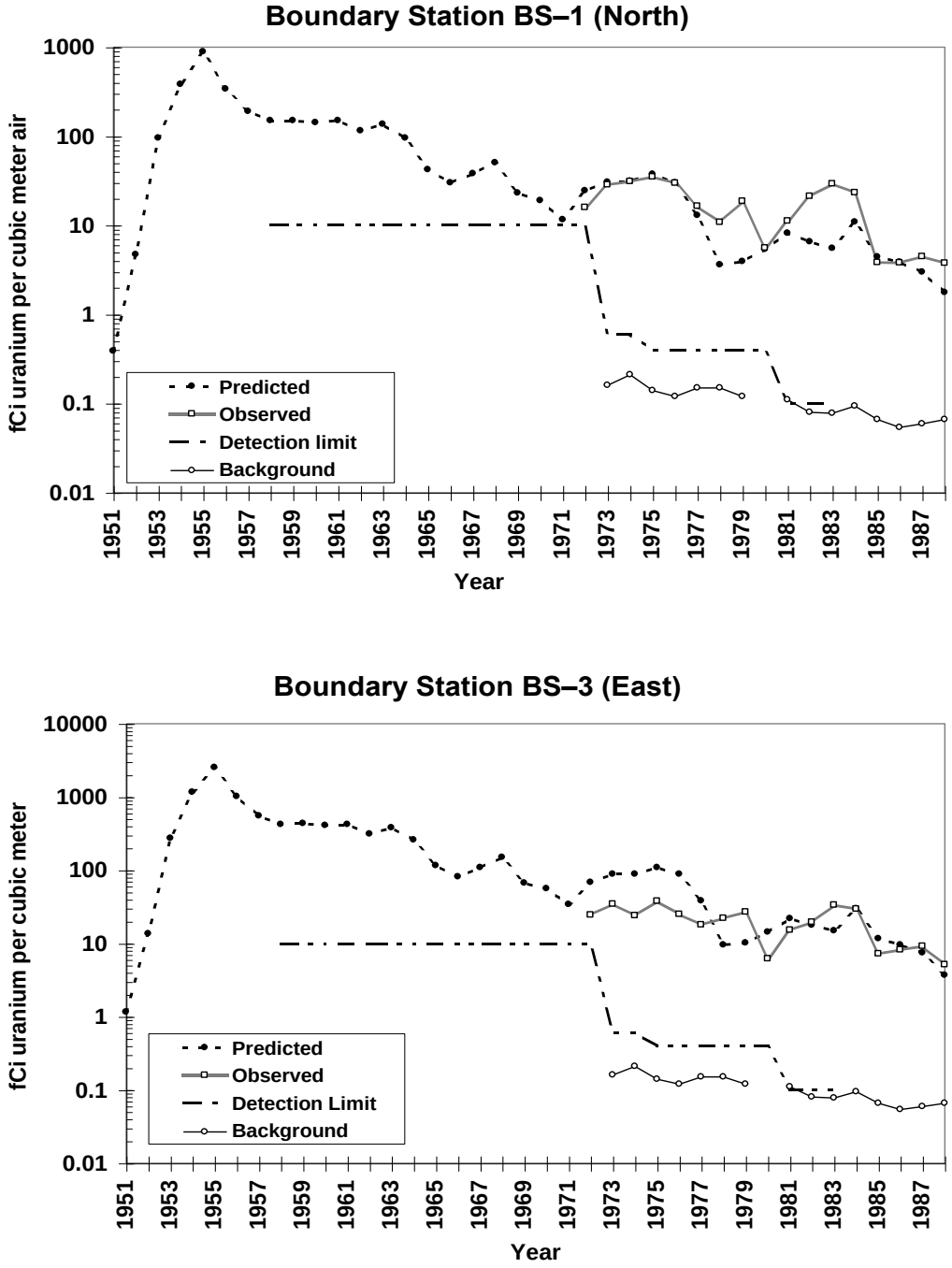



**Figure 33.** Observed and predicted concentrations of uranium in air at Boundary stations BS–1 (north) and BS–3 (east). Monitoring was conducted at boundary stations beginning in 1972. Predicted concentrations at the boundary were highest in 1955, when the uranium releases from the production area were highest.

**Validation with Deposition Measurements**

For uranium, deposition of particles released from the FMPC onto the ground surface eventually decreases the amount in the air transported downwind and results in a buildup of uranium in the soil. Deposition is a complicated process that depends on many factors including the characteristics of the released particles and the environment. Environmental measurements of deposition of uranium to gummed-film, a sticky monitoring surface that was collected periodically, are available over roughly a 10-year period (1954–1964). These data were used to validate the methods used to model deposition.

**The highest predicted annual average concentration of uranium in air at a boundary station was 2500 fCi uranium per cubic meter at the eastern station BS–3 in 1955. At that time, there was no air monitoring at the boundary which could confirm the predictions.**

A review of the gummed-film monitoring program and a tabulation of the data compiled from original analytical data sheets are included in Shleien et al. (1995). These original data were used to develop an estimate of annual deposition of uranium at a given place, which could be compared to the predicted annual deposition from our model. As with the air monitoring data, a correction for measurement bias (collection efficiency) was made before comparison of measured values to predictions. The collection efficiency correction was based on relatively recent reevaluations of this historical monitoring method and results of experiments done at the FMPC. The best estimate for the collection efficiency of the gummed-film was 14%.

**Deposition monitoring using gummed-film was conducted during the time period of highest releases of uranium from the FMPC. Monitoring was also conducted at various distances from the facility, giving us an opportunity to check our model performance with distance.**

Figure 34 locates the 24 gummed-film monitoring stations within the FMPC boundary. Another gummed-film monitoring station (NE–4), at 2.6 mi (4.2 km) northeast, is the only offsite location we used for validation. The other offsite stations, which were beyond 3 mi (5 km) from the FMPC, were excluded because of suspect data quality (discussed in Appendix O).

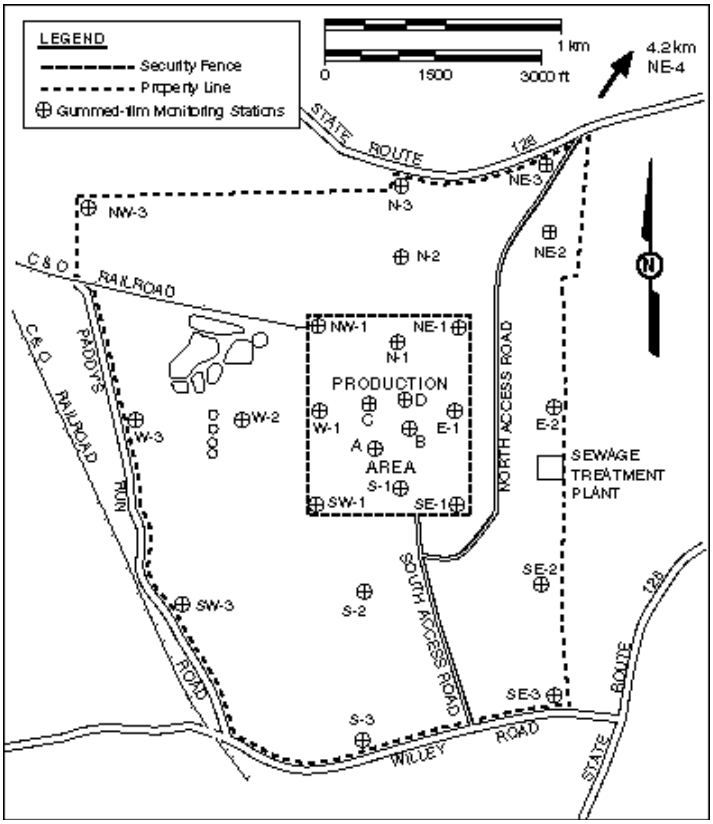



**Figure 34.** Gummed-film monitoring locations that provided data for validation of uranium deposition. This monitoring began at various times after 1954 and continued through 1964.

Figure 35 illustrates the trend in predicted and observed cumulative deposition of uranium in a northeasterly direction for the time interval 1957–1964. Cumulative deposition refers to the total amount deposited over the time interval that monitoring was conducted. As stated previously, the gummed-film monitoring data set was the only one available during this time period to check our model performance as a function of distance. We are generally pleased with the good agreement in this predominant wind direction.

The predicted and observed cumulative deposition (over the entire time interval of monitoring) was compared for all 25 locations (Figure 36). All predicted cumulative depositions were less than observations, after the observations were corrected for collection efficiency. Predicted and observed cumulative depositions were highly correlated (correlation coefficient = 0.92). The overall geometric bias of the predictions, compared to observations, was 0.41, indicating an underprediction of deposition (Figure 36). A detailed table illustrating this computation is included in Appendix N. The predicted-to-observed ratios show no trend with distance from the facility (Figure 36), and there are no other clear spatial patterns in the P/O ratios. As we found for air monitoring comparisons, the greatest underpredictions tend to be for locations west of the site. The best agreement between predicted and observed deposition is to the northeast.

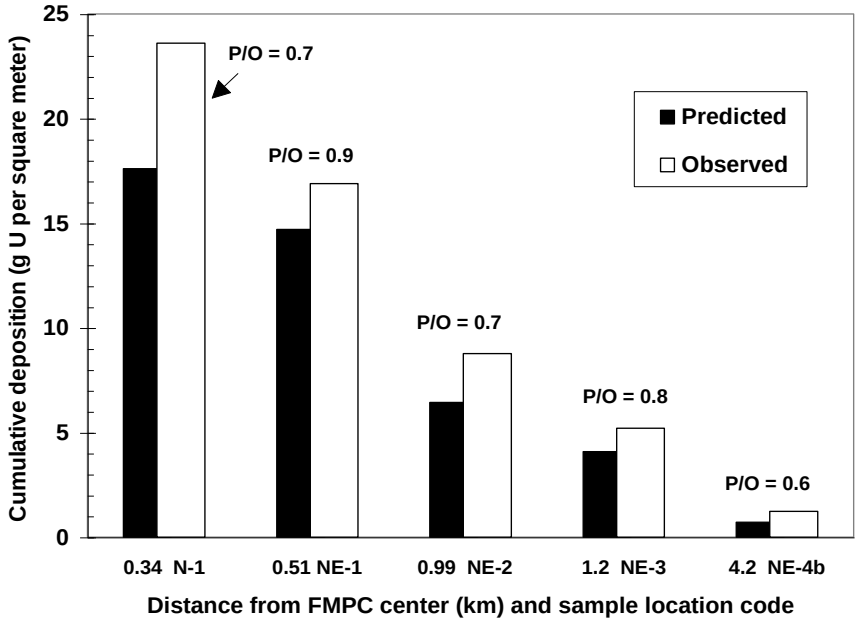



**Figure 35.** Comparison of predicted (P) and observed (O) deposition of uranium over an 8-year period (1957–1964) in a northeasterly direction from the FMPC. The period of comparison for the 2.6-mi (4.2-km) station is a 4-year period (1961–1964), because that monitoring station was not in place before 1961. Stations NE–1 and N–1 were operating as early as 1954. The predicted-to-observed (P/O) ratio for the longer period (1954–1964) was 0.8 for both N–1 and NE–1.

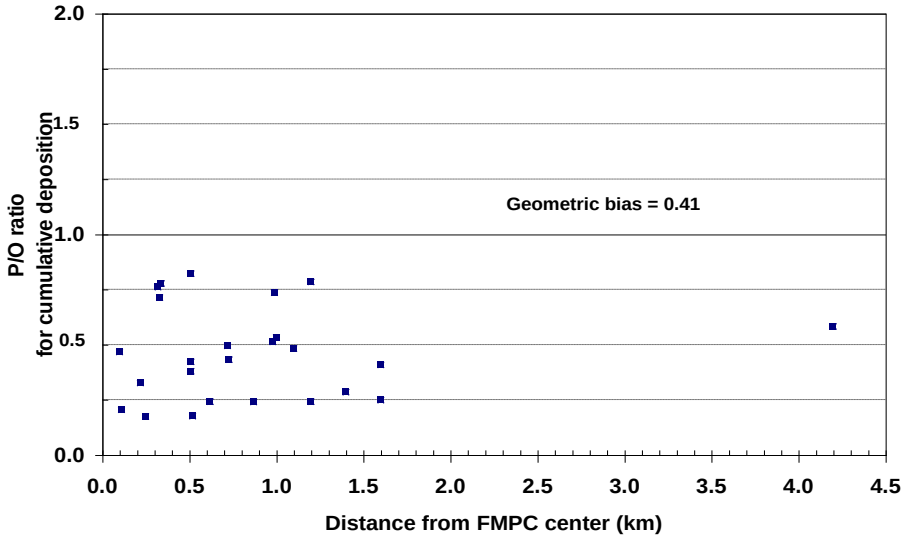



**Figure 36.** Distance trend in geometric bias (P/O ratio) for cumulative deposition as a function of distance from the FMPC. All 25 monitoring stations are included in this plot. The monitoring time interval varies (see Appendix N). The overall geometric bias was 0.41. Although there appears to be a systematic underprediction of deposition, there is no apparent trend with distance.

A time trend for annual observed and predicted deposition is illustrated in Figure 37. In general, our source term reconstruction and deposition models predict less deposition during peak years than was actually measured. However, only onsite monitoring locations were in place at that time. It is difficult to accurately assess deposition close to the sources and in an area with active operations. A waste oil burner, for example, increased onsite deposition of uranium in that immediate area (Voillequé et al. 1995, Appendix K). In addition, some fraction of the observed deposition could have been from resuspension and redeposition of uranium which had been spilled or released from stacks in a previous year, whereas the predicted deposition is based on the stack releases to air in the current year only.

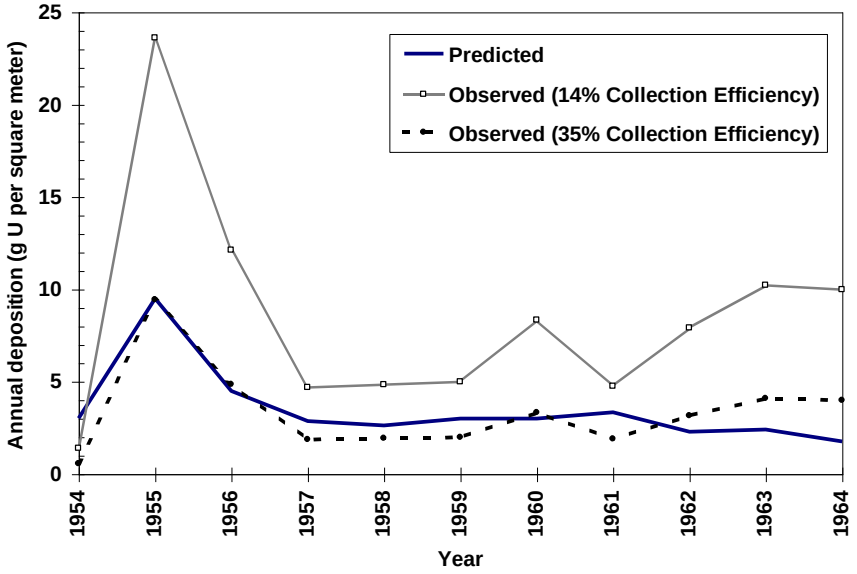



**Figure 37.** Time trend comparing predicted and observed depositions of uranium at onsite stations between 1954 and 1965. The observed data points shown are the annual averages of 10 or 11 stations located at the perimeter fence or within the production area. Our best estimate for collection efficiency of gummed-film is 14% (Shleien et al. 1995). The upturn in observed deposition at the end of this period is inconsistent with the trend in releases from the production area, which is gradually decreasing in 1961–1964. This discrepancy may be due to resuspension of previously deposited material by active operations occurring in the area. Although a collection efficiency of about 35% results in much better agreement with predicted depositions, we have no basis for that value. The time trend in observed depositions supports the reconstructed high annual release of uranium in 1955.

In summary, the most likely reasons for underprediction of uranium deposition include

1. Underestimation of airborne releases of uranium during this period (which is not supported by air monitoring data)
2. Underestimation of deposition of uranium by our model
3. Additional deposition from onsite operational activities and resuspension
4. Underestimation of collection efficiency of gummed-film (which results in higher adjusted observed deposition).

Deposition is a complicated process to model. A general tendency toward underprediction of uranium deposition is conservative because this results in more material remaining in the air for a downwind receptor to breathe. Inhalation of uranium in the primary release of airborne effluent is a more important pathway for dose to people than pathways following deposition of uranium on the ground. We are satisfied with the good agreement between predicted concentrations of uranium in air and historic air monitoring measurements and are not as concerned with the underprediction of uranium deposition. In addition, predicted deposition in the Ross area between 1957 and 1964 is in good agreement with observations made at that time (Figure 35).

**Validation Results — Uranium in Soil**

Another environmental medium which can be used to assess cumulative deposition of uranium is soil. Soil samples taken in 1959 indicated that large amounts of uranium had been deposited within the FMPC perimeter. Although soil samples were taken in conjunction with vegetation periodically throughout the 1960s, no information could be found on the soil sampling depth, which is needed to relate measured concentrations to predicted concentrations.

A routine monitoring program for uranium in soil at locations on the FMPC boundary was begun in 1971, providing an opportunity for comparison with predicted concentrations (Table 16). The geometric bias is quite good (1.1), although there is considerable year-to-year variation. Measured concentrations at boundary station BS–3 are believed to be affected by ash residues from the old solid waste incinerator near that location, resulting in a low geometric bias at that location. Several appendices in Volume II (B, C, and N) present detailed information about the measurements of uranium in soil, our models, and comparisons of predicted and observed concentrations.

**Table 16. Geometric Bias and Correlation for Predicted and Observed Concentrations of Uranium in Soil at Boundary Locations in 1959 and 1971–1988**

| Boundary Station | Geometric Bias | Correlation |
|---|---|---|
| BS–1 | 1.1 | 0.45 |
| BS–2 | 1.4 | 0.54 |
| BS–3 | 0.38 | 0.48 |
| BS–4 | 1.3 | 0.31 |
| BS–5 | 0.87 | 0.55 |
| BS–6 | 1.5 | 0.53 |
| BS–7 | 1.4 | 0.44 |
| **All-station bias** | **1.1** | |

**Summary of Validation Results for Uranium**

Three major types of environmental media (air, gummed-film, and soil) were used to validate our predicted environmental concentrations throughout the FMPC operating history. No one medium was monitored over the entire time period of FMPC operations, but in total,

these data cover a time interval from 1954 through 1988 (the end of this study's scope). Table 17 summarizes the validation results for these three media. There were significant discrepancies between predicted and observed concentrations for particular places and years. This is not unexpected, given the myriad of uncertainties involved in the source term reconstruction, the environmental transport calculations, and the environmental sampling and analysis. However, when viewed in the aggregate, the agreement between predicted and observed concentrations in the environment is excellent. This gives us confidence that our predicted doses from past releases of uranium from the FMPC are reasonable, particularly cumulative doses corresponding to long periods of exposure, and when the dose predictions are interpreted in the context of the uncertainties that accompany them.

**Table 17. Summary of Validation Results for Uranium Using Air, Soil, and Gummed-Film Monitoring Data**

| Medium and location | Number of stations (n) | Time interval | Geometric Bias (P/O) | Uncertainty in Bias [a] |
|---|---|---|---|---|
| Air (perimeter) | 4 | 1958–1971 | 1.0 | 0.6–1.8 |
| Air (boundary) | 7 | 1972–1988 | 1.0 | 0.6–1.6 |
| Gummed-film (perimeter to 4.2 km) | 20 | 1954–1964 | 0.4 | 0.3–0.7 |
| Soil (boundary) | 7 | 1959, 1971–1988 | 1.1 | 0.7–1.7 |

[a] All-station geometric bias (P/O) ×/÷ geometric standard deviation of geometric bias for n stations.

**Validation Results — Radon Measurements**

Routine monitoring of radon in air around the site boundary was initiated in 1980 by the FMPC and continues to the present. In the mid-1980s, Mound Laboratories performed additional radon in air monitoring at locations within the site boundary (including many locations closer to the K-65 silos). Because these are the most extensive sets of historical radon measurements around the FMPC, we used these two data sets to develop the radon dispersion model. One important criterion for validation data is that the data should not have been used to develop source terms or environmental transport models. Because these two data sets were used to calibrate the transport model, they were not used for validation exercises. There are a few other data sets that have been used for validation comparisons. Table 18 summarizes the important environmental radon measurements that were useful to the project. The data used for model development are discussed in Appendix M. Data used for validation comparisons are described here and in more detail in Appendix N.

Environmental measurements of radon around the FMPC have been much less extensive than those of uranium. Because the validation datasets for radon are relatively limited in scope, most of the validation exercises include fairly simple comparisons.

**Preliminary FMPC monitoring of radon in air.** The most important change to the K-65 silos that affected radon releases was sealing penetrations of the silo domes on June 25, 1979. Our predicted annual radon releases decrease by about 6.5 times after the silos were sealed. The routine monitoring program for radon in air began in 1980, so there is a large

data set for the time period after the silos were sealed. The small amount of data for radon in air before the sealing is discussed here.

**Table 18. Environmental Radon Measurements Around the FMPC Useful for Model Calibration orValidation**

| Data set | Years | Description | Use for the project |
|---|---|---|---|
| Preliminary FMPC | 1978–1980 | Radon concentrations in air, primarily at boundary air monitoring station BS-6. Measured before and after silo dome penetrations were sealed. | Validation: corroboration of decrease in radon releases after sealing of silo domes. |
| Radon decay products | 1978 | Radon decay product concentrations in air, primarily at BS-6. | Validation: qualitative support for radon decay product releases. |
| FMPC routine | 1980–present | Radon concentrations in air at seven (more in later years) boundary air monitoring stations and background locations. | Calibration of radon dispersion model. |
| Mound Laboratories, flux | 1984 | Radon flux measured on K-65 silo domes. | Validation: corroboration of general magnitude of 1980–1987 releases. |
| Mound Laboratories, concentration | 1984–1986 | Radon concentrations in air at 17 onsite locations, from less than 330 ft (100 m) to about 6560 ft (2000 m) from K-65 silos. | Calibration of radon dispersion model. |
| Hourly data | 1986–1991 | Hourly measurements of radon concentrations in air very close to the K-65 silos. | Validation: qualitative support for 1980–1987 releases occurring mainly during daylight hours. |

Time-integrated measurements of radon in air were made using passive radon monitors during 1978–1980 (although most measurements were made in 1979). Individual measurements encompassed periods of 1 day to 3 weeks. Because all but two of the measurements were taken at boundary air monitoring station BS-6, we only considered those measurements at BS-6. Figure 38 shows the location of the BS-6 sampling station, west of the silos. Results of the individual measurements at BS-6 are shown in Figure 39, along with average concentrations calculated for the periods before and after silo sealing.

The individual measurement results were presented in the Task 5 report (Shleien et al. 1995). From the individual measurements, the time-weighted average, gross concentrations (shown in Figure 39) were estimated for the periods before and after sealing. The results show a large decrease in measured radon concentration at BS-6 after the penetrations in the

K-65 silo domes were sealed, which is strong evidence that radon release rates were significantly higher before the sealing than after the sealing. This generally agrees with our predictions of significantly increased radon releases for the period before sealing of the silos.

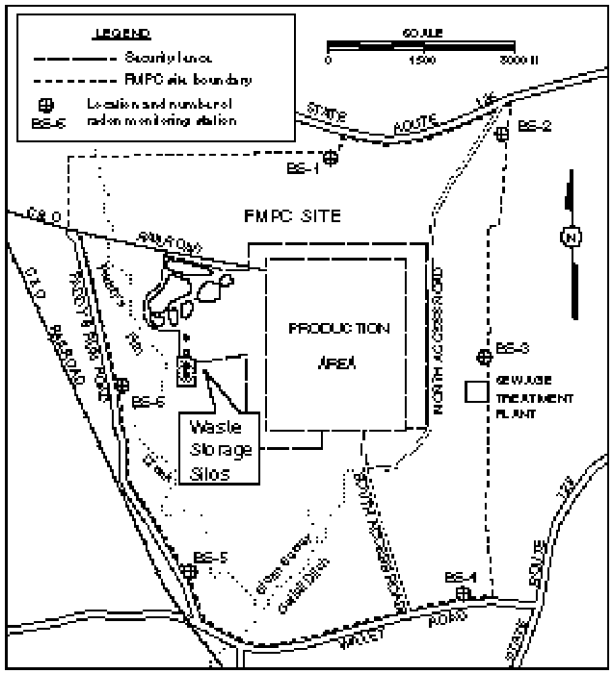



**Figure 38.** Preliminary FMPC radon monitoring was performed primarily at boundary air sampling station BS-6, west of the silos, during 1978–1980.

**Hourly measurements of radon in air close to silos.** Hourly measurements of radon in air can be useful in examining the diurnal variations (changes from day to night) in radon concentrations. Generally, winds are calmer at night, and less vertical mixing occurs in the stable air. These diurnal changes in meteorological conditions can produce related patterns in radon concentrations. In the presence of a constant radon source term, the meteorological conditions typically produce higher nighttime radon concentrations (less dispersion) and lower daytime concentrations (more dispersion).

**After sealing major openings for air exchange in 1979, radon concentrations in air decreased at a monitoring station west of the K-65 silos. This supports higher radon releases before the silos were sealed.**

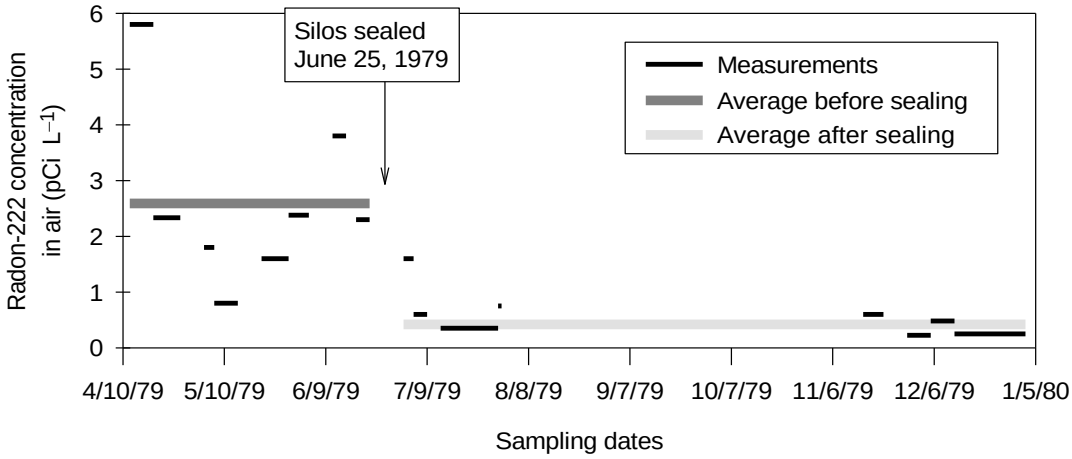



**Figure 39.** Measured concentrations of radon in air at boundary air monitoring station BS-6 before and after the penetrations in the domes of the K-65 silos were sealed. Station BS-6 is about 300 m west of the K-65 silos. Averages shown are time-weighted gross concentrations (gross means including natural background). The before-sealing average (2.6 pCi $L^{-1}$) includes results through June 22, 1979, and the after-sealing average (0.42 pCi $L^{-1}$) includes results of July 2, 1979, and later.

However, for our estimated radon releases from the K-65 silos for 1980–1987, we predicted release rates that are significantly greater during daylight hours. The hourly concentration measurements were useful to corroborate such a pattern.

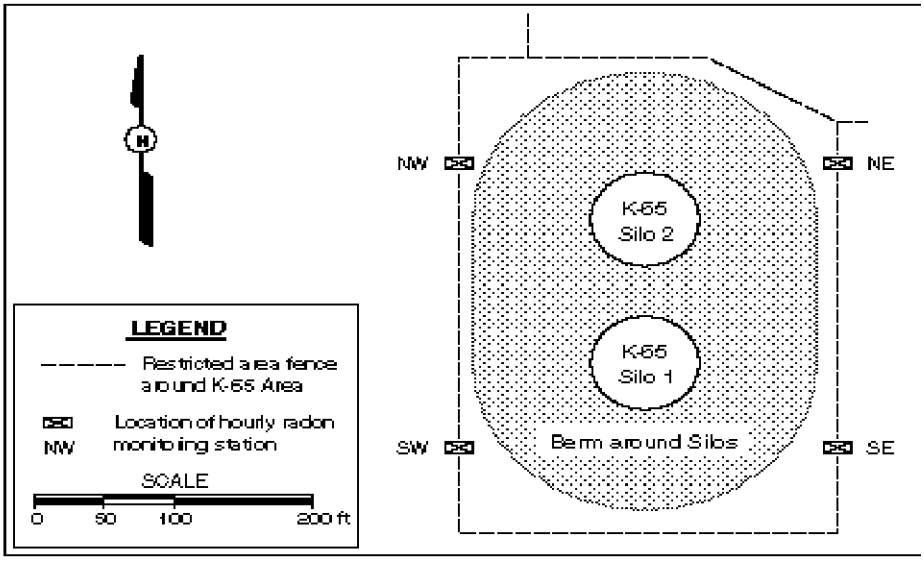



**Figure 40.** Locations of FMPC hourly radon monitoring on the fenceline of the K-65 area. The monitoring locations shown apply to the data from 1987 and 1991, although the locations are approximate and may have changed slightly (especially the northeast and southeast stations). For 1986, the precise location is not known, but the monitoring location was probably on the east side, outside the fenceline.

The hourly radon concentration measurements that can corroborate our estimates are those made nearest the K-65 silos, where radon concentrations can be many times higher than background concentrations. The hourly radon measurements are discussed in detail in Appendix N of this report. Very few days of hourly monitoring were available for 1980–1987. Hourly measurements became routine in 1988, and thus more data are available for 1988–present. Figure 40 shows the locations monitored near the K-65 silos. Table 19 summarizes the data evaluated (see Appendix N for details).

In evaluating the hourly data, we assumed data from 1986 and 1987 are representative of conditions for 1980–1987, and data from 1991 are representative of conditions for 1988–1991. For both groups of data, the peak radon concentrations are significantly greater than background concentrations, indicating the influence of the radon releases from the K-65 silos. The 1986 and 1987 data show peak concentrations in daylight hours. This is consistent with higher radon release rates during daylight hours. Peak radon concentrations in 1991 occurred at night and in the morning. This is consistent with a relatively constant radon release rate.

**Table 19. Summary of Hourly Radon Monitoring Data Evaluated**

| Dates | Days of data | Locations [a] | Comments |
|---|---|---|---|
| April 1986 | 3 | One location, not precisely known [b] | Two days before and day of an accidental radon release |
| Early November 1987 | 3 | NE, NW, SW | Before installation of foam layer on silo domes |
| Mid-November 1987 | 3 | NE, NW, SW, SE | Before installation of foam layer on silo domes |
| October 1991 | 9+ [c] | NE, NW, SW | After installation of foam layer on silo domes |

[a] NE = northeast, NW = northwest, SW = southwest, and SE = southeast. Locations are shown in Figure 40.
[b] The precise location is not known to us, but it was probably on the east side, outside the fenceline. See Appendix N for more information.
[c] Data were evaluated for the first 9 days of October 1991, although more data are available for 1988–1991.

Table 20 summarizes the conclusions that we drew from these data. Most importantly, we believe that the diurnal patterns seen for the two periods provide strong evidence that for 1980–1987 the radon release rate was much higher in daylight hours. This evidence is consistent with our predicted radon releases for 1980–1987.

**Table 20. Conclusions Drawn from Hourly Radon Data**

| Findings | 1980–1987 | 1988–1991 |
|---|---|---|
| Peak concentrations occur | Daylight hours | Night and morning |
| Source term predictions | Larger releases during daylight hours | Releases relatively constant over 24-hour period |
| Conclusions | Timing of peak concentrations supports predicted larger releases during daylight hours | Timing of peak concentrations appears consistent with predicted constant release rate |

**Mound measurements of radon flux on K-65 silo domes.** In 1984, personnel from the Mound Laboratories (a DOE facility in Miamisburg, Ohio) measured the radon flux emitted from the domes of the K-65 silos. Measurements were made with accumulating samplers, which used activated charcoal to adsorb the radon emitted from the domes. The measurements were made on October 18 and 24, 1984, at 24 locations on each of the K-65 silos (see Figure 41). On each silo, samplers were placed at 12 locations on what appeared to be intact concrete of the domes, along north-south and east-west lines. The other 12 sampling locations on each dome were on concrete showing obvious cracks. Details about the measurements are given in Appendix N of this report.

**Flux refers to an emission rate per unit area from a surface. For radon flux, results are often reported in units of picocuries per square meter per second.**

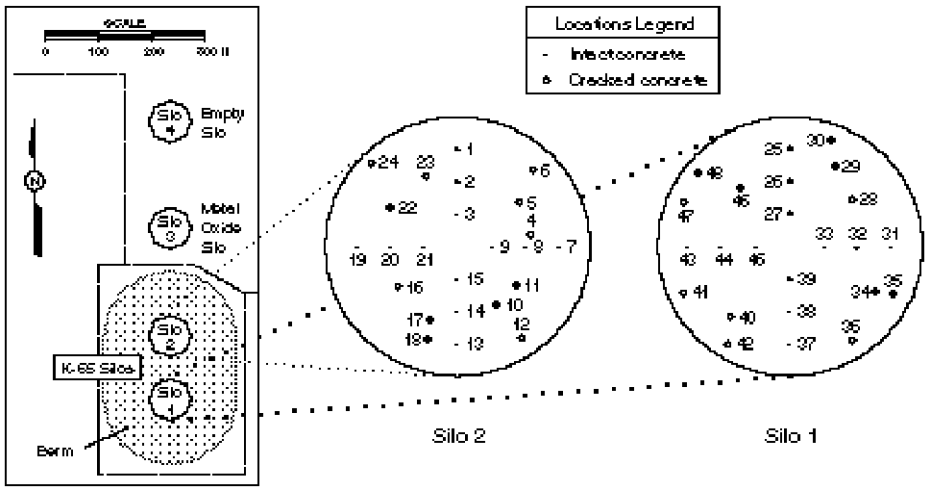



**Figure 41.** Locations of the Mound Laboratories radon flux measurements on the K-65 silo domes in October 1984 (based on figures in Hagee et al. 1985). Cracked concrete contained obvious cracks and intact concrete was without significant, visible cracks.

Compared to the average flux emitted from the silo domes, the measurements on cracked concrete are considered biased toward a higher flux and the measurements on intact concrete

are biased toward a lower flux. Because of these biases and the lack of knowledge about how much of the concrete was intact or cracked, a reasonable estimate of the average flux could not be determined. Instead, we used the intact concrete measurements to determine a lower bound on total radon flux through the silo domes. This lower bound flux was then converted to a lower bound on the total radon release rate for comparison with our predicted radon release rates from the K-65 silos. Figure 42 shows the 90% probability intervals for predicted releases by diffusion and air exchange and for the lower bound release rate based on the flux measurements. Details about individual measurement results and the calculated lower bound are provided in Appendix N.

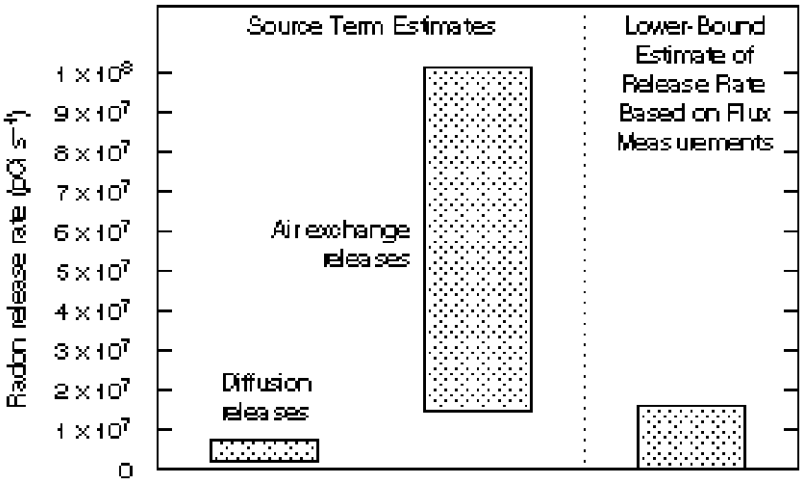



**Figure 42.** Comparison of predicted radon release rates from the K-65 silos for daylight hours during 1980–1987 with a lower-bound estimate of release rate based on flux measurements made by Mound Laboratories over intact concrete of the silo domes in 1984. Each box represents a 90% probability range.

The lower-bound estimate of radon releases, based on the flux measurements, is less than our current estimate of total radon releases from the silos for 1980–1987, but ranges to greater than our current estimate of diffusion releases. The general magnitude of the lower-bound estimate from the Mound Laboratories flux monitoring in 1984 is consistent with our current estimate of total radon releases for 1980–1987. We acknowledge that the radon flux measurements represent a limited data set, but the general consistency in results is reassuring.

**Radon decay product concentrations in air.** Preliminary radon monitoring also included some short-term (30-minute) air filter samples analyzed for radon decay products. These radon decay product samples were taken during September and October 1978, primarily at BS-6 (see Figure 38). Because of the short sampling times and an extremely high result for one measurement, the uncertainty in the radon decay product measurements is large, and we have made only a simple comparison of the data to predictions. Some additional information is provided in Appendix N of this report, and details about these measurements can be found in the Task 5 report (Shleien et al. 1995).

**Equilibrium of radon and radon decay products occurs when the concentrations of radon decay products equal the radon concentration.**

In the source term report (Voillequé et al. 1995), we estimated that for 1959–1979, radon decay products were released from the K-65 silos in quantities almost in equilibrium with radon releases. The average measured radon decay product concentration at BS-6 in 1978 was significantly above background and roughly comparable to the average gross radon concentration at BS-6 measured in 1979 before sealing the silos. This rough equivalence of radon and radon decay product concentrations indicates that radon decay products were probably released from the silos at levels close to equilibrium. This evidence provides some qualitative support for our estimates of the radon decay product releases.

## ESTIMATED DOSES RECEIVED BY THE PUBLIC FROM FMPC RELEASES

The final steps along the pathway of exposure are calculating the radiation *dose*s and health risks to the residents in the FMPC area (Figure 2). This final stage follows steps previously described in this report:

- Identifying release points of radioactive material from the FMPC
- Determining how much and what kinds of radioactive materials were released from the facility (source term)
- Understanding the way materials were dispersed from the facility to the environment (air and water transport)
- Identifying the material in various environmental media (air, soil, water, vegetation, milk, and food).

The radiation dose that a person receives depends upon a number of factors, such as:

- Where the person lived, went to school, and worked in relation to the facility
- When and how long that person lived near the facility (for example, during the 1950s when releases from the site were high or in the 1970s when releases were less)
- Lifestyle (that is, did the person spend a great deal of time exercising outdoors or doing heavy work on a farm)
- The types and amounts of food produced in the assessment domain that the person ate.

To consider these features of a person's life, we developed profiles, or exposure scenarios, of nine hypothetical, but realistic residents of the FMPC area, for whom radiation dose estimates could be made. In the following sections, we describe these scenarios in detail and present the dose estimates for these reference individuals.

**Description of Scenarios**

Dose calculations are based on these scenarios, which incorporate some typical lifestyles and food habits of residents in the area and specify the applicable home, school, and work locations. The number of years that the person resided in the area during the period of FMPC operation is related to the total dose. In some cases, we have assumed individuals spent their entire lives in the area (scenarios 1, 2, 3, 5, 6 and 7), whereas in others, we have assumed individuals left after completing high school (scenarios 4 and 9). We have assumed that individuals for most scenarios lived in the area during the highest releases of the 1950s (1, 2, 3, 5, 6, 7, 9) whereas for individuals in scenarios 4 and 8, we have assumed they were born or came to the area after the period of peak releases. These scenarios can help residents determine dose ranges for themselves by finding a lifestyle profile that most closely matches their background. The scenarios are not designed to include all conceivable lifestyles of residents of this region during the time of the FMPC operations. Rather, they serve as guides to a range of potential exposures to FMPC radionuclides and radiation of people in the area.

**An exposure scenario is a profile of a fictional, but realistic, individual in the Fernald area for whom we have made specific assumptions about lifestyle, diet, and locations of residence, school, and work. Many people who have lived near the FMPC may be able to relate their own experiences to features in the nine scenarios developed in this study.**

Each of the scenarios is identified by a number and a phrase that highlights the main feature of the scenario. Figure 43 locates each of the nine scenarios by the applicable home, school, and work locations. Table 21 provides a key to the home, school, and work locations. All scenarios consider only exposure to FMPC releases that occurred within the 6.2 mi (10-km) assessment domain during the years 1951–1988. These scenarios are described in detail in Appendix J; their distinctive features are summarized here.

**SCENARIO 1:** ***Realistic maximum inhalation exposure (northeast of site)***
**Home location:** Within 1 mi (1.7 km) of site center on Route 126.
**Direction:** Northeast sector
**Date of birth and sex:** 1-1-46 Female
**Years exposed to FMPC releases:** 38
**Brief summary:** This woman was born in 1946 and has lived her entire life at the same location (family farm) to the northeast of the site. She attended Ross schools, married, and continues to live in the area. She carried a pregnancy to term during 1964-1965. She obtained 50% of her vegetables from a local garden, 50% of her annual consumption of fish from the Great Miami River, 50% of her beef and poultry from local sources, and 100% of her eggs from a local source, and 100% of her milk from a local source. She used no contaminated water for drinking or irrigation.

**SCENARIO 2:** ***Resident close to K-65 radium storage silos (west of the site)***
**Home location:** Within 1.2 mi (2 km) of site center to west of Paddy's Run Road.

**Direction:** West sector
**Date of birth and sex:** 1-1-51 Male
**Years exposed to FMPC releases:** 38
**Brief summary:** This man was born in 1951 and has lived his entire life at the same location (family farm). He attended Ross schools; worked in Hamilton as an adult; consumed 50% of his vegetables from a local garden, 50% of his poultry from a local source, 100% of his eggs from a local source, and 100% of his milk from a local source; and irrigated his garden with water from Paddy's Run Creek (this last assumption is highly unlikely).

**SCENARIO 3:** ***Ingestion of uranium-contaminated well water (south of the site*** **)**
**Home location:** Within 1.2 mi (2 km) of site center to the south.
**Direction:** South sector
**Date of birth and sex:** 1-1-51 Male
**Years exposed to FMPC releases:** 38
**Brief summary:** This man was also born in 1951 and lived his entire life at the same location (family farm) to the south of the site. He attended Ross schools; worked at a local dairy as an adult; consumed 50% of his vegetables from a local garden, 50% of his poultry from a local source, 100% of his eggs from a local source, and 100% of his milk from a local source; and obtained his drinking and irrigation water from a well with elevated levels of uranium from the site. Chemical toxicity of uranium in the kidneys is discussed in Appendix R for the subject of this scenario

**SCENARIO 4:** ***Realistic average inhalation exposure from*** **1960 to 1988 (i*n Ross*)**
**Home location:** 2.4 mi (4 km) from site center in Ross.
**Direction:** East northeast sector
**Date of birth and sex:** 7-15-60 Female
**Years exposed to FMPC releases:** 18
**Brief summary:** This woman was born in 1960 and was raised and attended schools in Ross. She left the area after high school in 1978. While living in the Fernald area, she consumed 10% of her vegetables from a local garden, 10% of her eggs from a local source, and 10% of her milk from the local dairy.

**SCENARIO 5:** ***Realistic low exposure who worked outside area (north of site)***
**Home location:** Near Layhigh, 5 mi (8 km) north of site center.
**Direction:** North sector
**Date of birth and sex:** 1-1-51 Male
**Years exposed to FMPC releases:** 38
**Brief summary:** This man was born in 1951 and lived 5 mi (8 km) north of the site near Layhigh. He attended Morgan Elementary School on Chapel Road West of Shandon and Ross Middle and High Schools. As an adult, he worked in Hamilton. He consumed no food produced or grown locally.

**SCENARIO 6:** ***Garden irrigated with Great Miami River water (southeast of site)***
**Home location:** 1.9 mi (3 km) east southeast of site center on the river.
**Direction:** East southeast sector
**Date of birth and sex:** 1-1-46 Female

**Years exposed to FMPC releases:** 38
**Brief summary:** This woman was born in 1946 and grew up about 2 mi (3 km) to the southeast of the FMPC near the Great Miami River. She attended Ross schools and lived her entire life at same location (family farm). She carried a pregnancy to term during 1964-1965. She consumed 50% of her vegetables from a local garden, 50% of her annual consumption of fish from the river, 50% of her beef and poultry from local sources, 100% of her eggs from a local source, and 100% of her milk from the local dairy. She used water from the Great Miami River to irrigate his garden.

**SCENARIO 7:** ***Garden irrigated with Great Miami River water further from site (south of site)***
**Home location:** 6.2 mi (10 km) south of site center in Miamitown on Great Miami River.
**Direction:** South sector
**Date of birth and sex:** 1-1-51 Male
**Years exposed to FMPC releases:** 38
**Brief summary:** This man was born in 1951 and grew up about 6.2 mi (10 km) south of site in Miamitown near the Great Miami River. He attended schools in Ross and worked in Miamitown after graduation from high school. He consumed 50% of his vegetables from a local garden, 50% of his fish from the river, 50% of his poultry from local sources, 10% of his eggs from a local source, and 10% of his milk from a local dairy. He used water from the Great Miami River to irrigate.

**SCENARIO 8:** ***Exposure from 1975 to 1988 as a child (in Ross)***
**Home location:** 2.5 mi (4 km) from site center in Ross.
**Direction:** East northeast sector
**Date of birth and sex:** 1/1/70 Male
**Years exposed to FMPC releases:** 13
**Brief summary:** This boy was born in 1970 and moved with his family from out of state to Ross in 1975. He attended Ross schools and consumed 10% of his vegetables from a local garden, 10% of his eggs from a local source, and 10% of his milk from the local dairy.

**SCENARIO 9:** ***Attended school in Ross; lived 6.2 mi (10 km) away (northeast of site)***
**Home location:** Near Rte 128 to the NE, 6.2 mi (10 km) north of site center.
**Direction:** Northeast sector
**Date of birth and sex:** 1-1-51 Male
**Years exposed to FMPC releases:** 18
**Brief summary:** This man was born in 1951 and lived 6.2 mi (10 km) northeast of the Fernald site. He attended Ross Schools and left the area following high school graduation. He consumed no food produced or grown locally.

**Dose Estimates for the Nine Scenarios**

A key variable in all scenarios is the primary location of the person relative to the site where the releases occurred. Both the distance and the direction of the subject's location from the center of the FMPC production area are important in estimating exposure, dose and, ultimately, risk to the person. In cases where the residence is closer to the site (scenario 1), the

dose for individuals would be expected to be higher than for individuals living farther away (scenario 4) if all other factors were comparable. Because the predominant winds in the area blow from the southwest toward the northeast, an individual northeast of the FMPC may have received a higher dose from airborne radioactivity than an individual at the same distance from the site but in another direction.

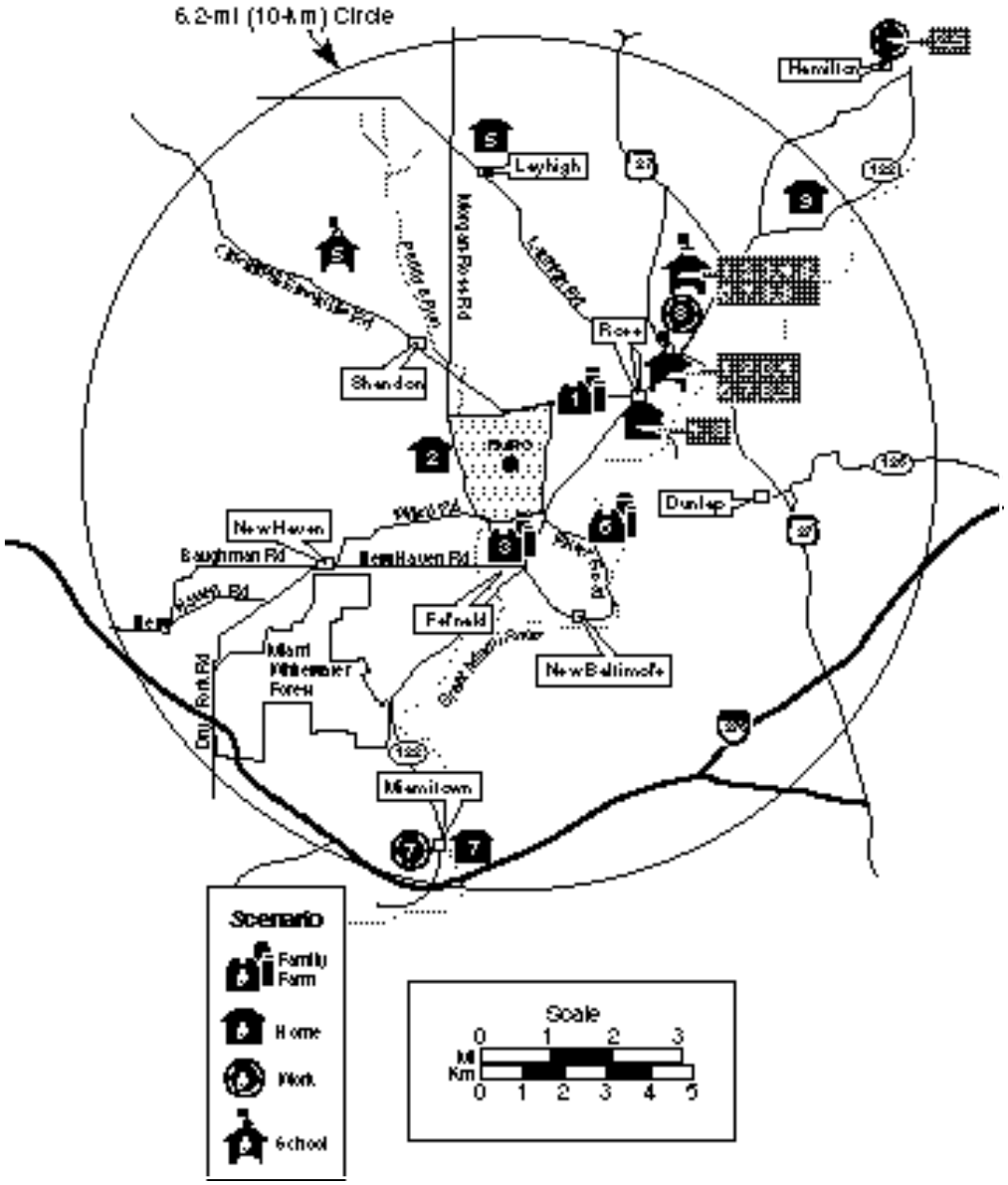



**Figure 43.** A map of the FMPC region showing the locations of the nine scenarios used in the dose calculations. For each of the nine scenarios, home, school, and work locations are marked (Table 21). Family farm locations are identified when a person represented by a scenario lives and works on a family farm. In some cases, the three locations are the same or quite close together (for example, scenarios 4 and 8).

**Table 21. Key to Locations of Home, School and Work for the Nine Exposure Scenarios[a]**

| Scenario number | Years of exposure | Key feature | Home | School | Work |
|---|---|---|---|---|---|
| 1 | 38 | Received inhalation exposure close to site | Family farm | Elda Elementary, Ross Middle and High Schools | Family farm |
| 2 | 38 | Lived close to K-65 silos | Family farm | Elda Elementary, Ross Middle and High Schools | Hamilton |
| 3 | 38 | Drank well water | Family farm | Elda Elementary, Ross Middle and High Schools | Family dairy farm |
| 4 | 18 | Received typical inhalation exposure | Ross | Elda Elementary, Ross Middle and High Schools | Moved away |
| 5 | 38 | Worked outside the area | Near Layhigh | Morgan Elementary Ross Middle and High Schools | Hamilton |
| 6 | 38 | Irrigated using water near the site | Family farm | Elda Elementary, Ross Middle and High Schools | Family farm |
| 7 | 38 | Irrigated using water farther from the site | Miamitown | Elda Elementary, Ross Middle and High Schools | Miamitown |
| 8 | 13 | Received exposure as a child in Ross | Ross | Elda Elementary, Ross Middle and High Schools | Family farm |
| 9 | 18 | Attended school in Ross and then left area | Near Route 128 | Elda Elementary, Ross Middle and High Schools | Moved away |

[a] Each of the nine exposure scenarios is identified by number and symbol on the map shown in Figure 43.

Furthermore, for similar locations, the doses tend to be higher for individuals who lived at a location for a longer period of time (scenario 1 compared to scenario 4) or who lived there during a period of higher releases (scenario 4 compared to scenario 8). It is important to keep in mind that all scenarios consider a number of key pathways of exposure. For example, in scenario 3, where a distinguishing characteristic is the consumption of uranium-contaminated well water in the diet, the inhalation of uranium and radon were also considered in the dose calculations.

The final dose calculations presented in this section consider all those characteristics described in each scenario. We calculated doses to the residents in the area of the FMPC represented by each of the nine exposure scenarios from the two main types of radioactive materials released and to seven target organs from uranium and external gamma radiation:

1. lungs
2. bone surface
3. kidney
4. liver
5. red bone marrow
6. testes
7. ovaries.

The residents in the area of the FMPC were exposed to two main types of radioactive materials:

1. Uranium isotopes (uranium-234, uranium-235, and uranium-238); thorium isotopes from thorium production; and decay products of these radionuclides. The exposure pathways from these materials are inhalation of airborne material; consumption of contaminated drinking water and food from a family garden or locally produced animal products; and external exposure to gamma-emitting radionuclides by immersion in contaminated air, standing on contaminated ground, or swimming in contaminated water.
2. Radon-222 and its decay products. Radon-222 is produced by the decay of radium-226 contained in the K-65 silos west of the FMPC production area (Figure 38). The exposure pathways from radon are inhalation of radon decay products and external exposure to gamma rays from the K-65 silos and radon decay products in the air.

**Effective dose provides a measure of the dose to the whole body and considers the dose absorbed by each of the target organs and the sensitivity of those organs to radiation. Equivalent dose for a particular type of radiation is the absorbed dose multiplied by a quality factor that adjusts approximately for the probability of health effects.**

The general term, dose, is used in this report to describe the amount of radiation that a person received. There are more specific terms that describe a particular dose quantity, such as, *absorbed dose, equivalent dose* or *effective dose.* The absorbed dose is the energy imparted to a unit mass of tissue in the body by radiation. The unit of absorbed dose is the rad (traditional system) or *gray* (SI system); 1 gray (Gy) = 100 rad. The probability of a health effect for 1 gray of absorbed dose to an organ depends on the type of radiation that is absorbed (for example, alpha, beta, or gamma radiation). The equivalent dose for a particular type of radiation is the absorbed dose multiplied by a quality factor that adjusts approximately for the probability of health effects. The effective dose provides a measure of the dose to the whole body, taking into account the dose absorbed by each of the target organs and the sensitivity of those organs to radiation effects. The unit of equivalent dose and effective dose is the rem (traditional system) or sievert (SI system); 1 sievert (Sv) = 100 rem.

For each exposure scenario, the absorbed dose to each relevant organ from each radio-

nuclide was estimated for each of the exposure pathways using dose conversion factors (Appendix I and K). The radiation components are categorized according to two distinct groups: (1) high linear energy transfer LET radiations such as alpha particles and (2) low LET radiations such as beta particles and gamma rays. High LET radiations are approximately 20 times more effective biologically for inducing cancer than low LET radiations.

**Ionizing radiation can be divided into low-LET and high-LET components. Gamma and x-rays are low LET; alpha-emitting radionuclides like some radon decay products are high LET. The breakdown into these two components is important in analyzing the risk from FMPC releases.**

The equivalent dose for each organ and each pathway is determined by multiplying the high LET component of the absorbed dose by 20 and adding the low LET component of the absorbed dose to it. The total equivalent dose for each organ is the sum of the equivalent doses for all of the pathways for that organ. For radon and radon decay products, the equivalent dose to the lung was estimated by a dosimetric model that takes into account the concentration in the air the person breathes of each radon decay product. The model is outlined in Report 78 of the National Council on Radiation Protection and Measurements (NCRP 1984) and in Appendix I (Volume II) of this report. The specific region of the lung that is most affected by radon decay products is the *tracheobronchial epithelium* (TBE). No organ or tissue other than the lungs (tracheobronchial epithelium) receives a significant internal exposure from inhalation of radon decay products, but external dose from airborne gamma-emitting radon decay products is estimated and is the largest componenet of external dose considered.

All estimates of dose with uncertainty bounds for the nine scenarios are described and tabulated in detail in Appendix K, Tables K-2, K-4, and K-5. Determining the uncertainties associated with the dose and risk estimates is a fundamental part of the project. We used a Monte Carlo procedure to draw random samples of all possible combinations of input values and calculate the dose or risk for each combination. This sampling and calculation is repeated hundreds (sometimes thousands) of times to develop a distribution of possible results. Figure 44 shows an example of a frequency distribution of dose results, with the 5th, 50th and 95th percentiles marked (frequency is the number of times a result was obtained; in the figure, the total of all frequencies would be 1000, because 1000 samples were drawn). The figure shows a nearly complete picture of the way probabilities (which are related to the plotted frequencies) are distributed. Half of the probability (frequency) lies below 240 rem, making 240 the 50th percentile (also called the median of the distribution). Ninety percent of the probability lies between 90 rem and 636 rem, so that we would conclude that the dose has 90 chances in 100 of lying in this interval. Notice in the figure that the 50th percentile is not at the midpoint of this 90% interval but nearer the lower end of it, indicating that the distribution is skewed. If we could begin at zero and add up frequency from the plot as we moved to the right, when we reached 90 rem, we would have added 5% of the total; when we reached 636, we would have accounted for 95%. When we present a dose estimate by giving the 5th, 50th, and 90th percentiles, we do not give complete information about the distribution, but rather we account for 90% of the probability, and the location of the 50th percentile gives a

sense of how skewed the distribution would look if we plotted it. The tails of the distribution are the parts that lie outside the 90% interval; there is a 5% probability that the dose lies in the upper tail (above 636 rem) and a 5% probability that it lies in the lower tail (between 0 and 90 rem). One should be careful not to focus all attention on the 50th percentile while forgetting the other information about the distribution, such as the 5th and 95th percentiles. The 50th percentile, by itself, is not a dose (or risk) estimate; the entire distribution is the estimate.

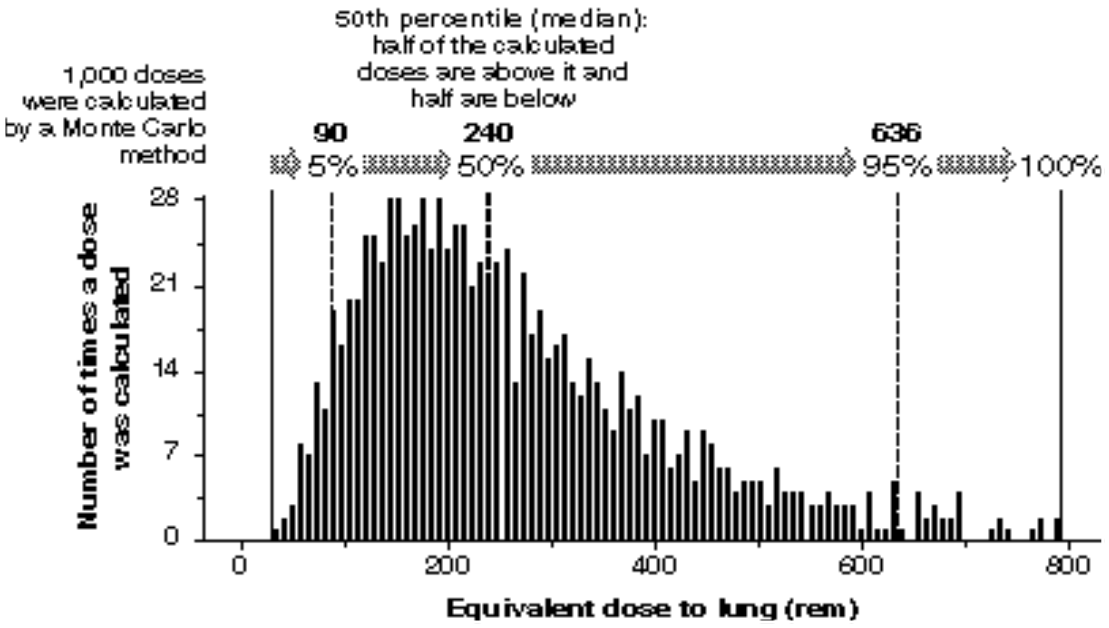



**Figure 44**. An example of a distribution of dose calculations. A computer randomly samples all possible combinations of input values (parameters) and calculates the dose for each combination. The sampling and calculation are repeated (in this case) 1000 times. The vertical scale represents the number of times a result was obtained. The horizontal scale represents the dose that was calculated. The value of 240 rem divides the frequency of calculated values in half—50% of the frequency is above 240 rem and 50% is below. This value (240 rem) is called the median or 50th percentile. In this example, 90% of the frequency lies between 90 rem (the 5th percentile) and 640 rem (the 95th percentile). Five percent of the frequency lies above the 95th percentile and 5% lies below the 5th percentile.

This section presents the radiation doses in the following order:

- Effective radiation dose from uranium compared to doses from radon decay products
- Cumulative equivalent dose, that is, the annual radiation dose to specific organs in each scenario summed over the years of exposure
- Comparison of these doses with those from natural background radiation
- Annual doses to the person described in scenario 1.

First of all, we can compare the relative contribution of the two general categories of radioactive materials (uranium, thorium and associated products, and radon decay products*)* to

the effective dose. The effective dose gives a measure of the dose to the whole body, taking into account the dose absorbed by each of the target organs and the sensitivity of those organs to radiation effects. Our calculations show that the decay products of radon are the most important radionuclides, contributing roughly 84 to 98% of the effective dose, depending on the particular scenario (Figure 45). Even the person who consumed water from contaminated wells (scenario 3) received a much higher radiation dose from inhalation of radon decay products than from uranium in the well water. The person in sceneario 3 received a cumulative effective dose from all pathways and from both uranium and radon decay products of about 0.20 Sv (20 rem). Of that total, radon contributed 87% or 0.17 Sv (17 rem), and the ingestion of uranium in contaminated water from wells contributed about 0.015 Sv (1.5 rem). (The chemical effects of uranium on the kidney are considered in a later section of the report.)

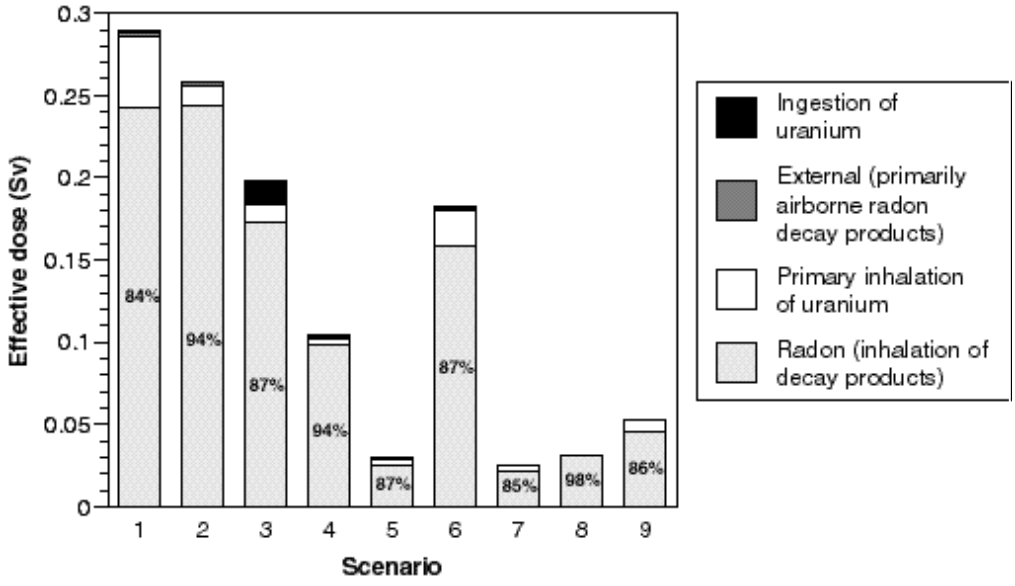



**Figure 45.** Comparison of the cumulative effective dose contribution from exposure to uranium and to radon decay products. The numbers in the black area refer to the percentage of the total effective dose, which results from radon and radon decay products. From 84% (scenario 1) to 98% (scenario 8) of the dose for all scenarios is contributed by radon decay products. The individuals in scenarios 1, 2, 3 and 6 received median radiation doses greater than the natural background effective dose of 0.11 Sv (11 rem).

Table 22 is the principal summary of effective dose from uranium and associated radionuclides to the individuals of scenarios 1 to 9 with propagated uncertainties. Each dose estimate represents the median (50th percentile) of the total (such as inhalation, ingestion, and direct external dose) and cumulative over all years of plant operation (1951–1988).

**Table 22. Cumulative Effective Dose (Sv) from Uranium Releases at the FMPC[a]**

| Scenario | Years of exposure[b] | Median, or 50th percentile | 5th and 95th percentiles | Background effective dose [c] |
|---|---|---|---|---|
| 1 | 38 | 0.061 | 0.021–0.18 | 0.11 |
| 2 | 38 | 0.021 | 0.0074–0.057 | 0.11 |
| 3 | 38 | 0.031 | 0.019–0.061 | 0.11 |
| 4 | 18 | 0.0093 | 0.004 –0.022 | 0.054 |
| 5 | 38 | 0.0053 | 0.0017–0.014 | 0.11 |
| 6 | 38 | 0.0.033 | 0.011–0.088 | 0.11 |
| 7 | 13 | 0.0055 | 0.0021–0.012 | 0.039 |
| 8 | 38 | 0.00096 | 0.0003–0.0025 | 0.11 |
| 9 | 18 | 0.011 | 0.0035–0.03 | 0.054 |

[a] See Appendix K for details.
[b] To FMPC releases.
[c] Values are computed for the number of years of exposure based on NCRP Report No. 93 (NCRP 1987), Table 2.4. The median average annual effective dose from various sources of natural background radiation is 0.003 Sv (0.3 rem); thus, for 38 years of exposure, the background dose would be (0.003 Sv per yr) × (38 yr)= 0.11 Sv (11 rem). For scenario 7, the background dose from natural background radiation during the 13 years of exposure to FMPC releases would be 0.039 Sv (3.9 rem). The lifetime (70 year) background effective dose would be 0.21 Sv (21 rem).

A second important finding of the study was that the organs receiving the highest equivalent radiation doses, in order, were lung, bone surfaces, red marrow, kidney, and liver. Figure 46 compares the median doses to the lungs, bone surfaces and kidney from uranium, to the dose to the TBE portion of the lung from radon releases at the FMPC. The doses to the testes and the ovaries were less than those to the other organs. For all scenarios, the dose to the lung from releases of radon was significantly higher than the dose to other organs from uranium. The person in scenario 1 had the highest median dose to the TBE portion of the lung from radon releases of 3.6 Sv (360 rem), with an uncertainty distribution of 0.98 to 14 Sv (98 to 1400 rem). The subject of scenario 3, whose source of drinking water was a uranium-contaminated well, had the highest median dose to bone surface of 0.26 Sv (26 rem), with an uncertainty distribution of 0.22 to 0.34 Sv (22 to 34 rem). However, his bone surface dose was significantly less than his dose from radon releases to the TBE of 2.6 Sv (260 rem), with an uncertainty distribution of 0.89 to 10 Sv (89 to 1000 rem).

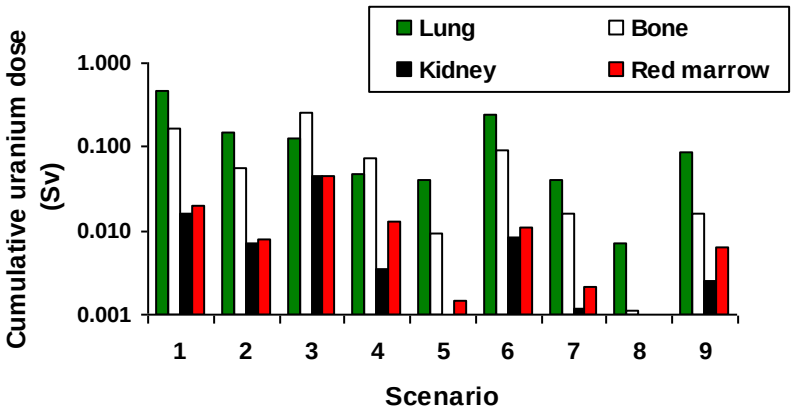



**Figure 46**. Median estimates of cumulative equivalent doses (Sv) to the lungs, bone surfaces, kidneys and red bone marrow of individuals in the nine scenarios from releases of uranium from the FMPC. Radiation doses to the liver, testes and ovaries were less than doses to the organs shown in the figure.

Table 23 summarizes the cumulative equivalent doses from radon decay products to the TBE of the individuals of scenarios 1 to 9 with propagated uncertainties. For scenario 1, the natural background radiation dose to the TBE would have been about 0.95 Sv (95 rem). The background values in Table 23 are computed for the number of years of exposure to FMPC releases and the annual background dose, based on the NCRP Report No. 93 (NCRP 1987). The NCRP estimates the average annual equivalent dose to the TBE of the lung from various sources of natural background radiation to be 0.025 Sv (2.5 rem). Thus, for a lifetime exposure (70 yr), ), which some would consider a more appropriate comparison, the background dose would be (0.025 Sv per yr) × (70 yr) = 1.8 Sv (180 rem).

**Table 23. Cumulative Dose (Sv) to Tracheobronchial Epithelium from Radon**

| Scenario | Years of exposure[a] | Median, or 50th percentile | 5th and 95th percentiles | Background dose to TBE of lung[b] |
|---|---|---|---|---|
| 1 | 38 | 3.6 | 0.98–14 | 0.95 |
| 2 | 38 | 3.6 | 0.98–13 | 0.95 |
| 3 | 38 | 2.6 | 0.89–10 | 0.95 |
| 4 | 18 | 1.5 | 0.40–7.2 | 0.45 |
| 5 | 38 | 0.42 | 0.10–1.9 | 0.95 |
| 6 | 38 | 2.2 | 0.53–9.2 | 0.95 |
| 7 | 13 | 0.389 | 0.12–1.5 | 0.33 |
| 8 | 38 | 0.44 | 0.10–2.2 | 0.95 |
| 9 | 18 | 0.84 | 0.17–4.8 | 0.45 |

[a] To FMPC releases.

[b] Values are computed for the number of years of exposure based on the NCRP Report No. 93 (NCRP 1987), Table 2.3. The median average annual equivalent dose to the tracheobronchial epithelium is 0.025 Sv; thus, for 38 years of exposure, the background dose is (0.025 Sv per yr) × (38 yr)= 0.95 Sv (95 rem). The lifetime (70 years) background dose to the TBE of the lung from radon in the environment is 1.8 Sv (180 rem). The NCRP report points out that more than 1% of the population is exposed to at least 5 times the average concentrations of radon decay products, and thus presumably at least 5 times the annual 0.025 Sv to the TBE.

The results demonstrate clearly that inhalation is the most important pathway by which the public would have been exposed to radioactive materials released from the FMPC from 1951 through 1988, and radon is the largest contributor to total dose. When considering just uranium, however, the ingestion pathway can be as important as inhalation, depending upon the scenario. The radiation dose from uranium by the ingestion pathway is comparable to the inhalation pathway for scenario 3 (well water consumption) (Figure 45).

**Inhalation is the most important pathway of exposure and radon is the source of most of the dose. Calculations show that the decay products of radon contribute from 70% to 95% of the effective dose, depending on the scenario.**

Figure 47 shows the effect that distance from the center of the FMPC has on the cumulative dose to the individuals who were assumed to have had 38 years of exposure (1, 2, 3, 5, 6, 7). (We assumed that the years of exposure were less for those in scenarios 4, 8, and 9). As the figure confirms, the cumulative dose tends to decrease with increasing distance from the FMPC. Other factors contribute to the dose, as well, and the direction of the residence from the site is also a factor directly related to the dose that a person receives.

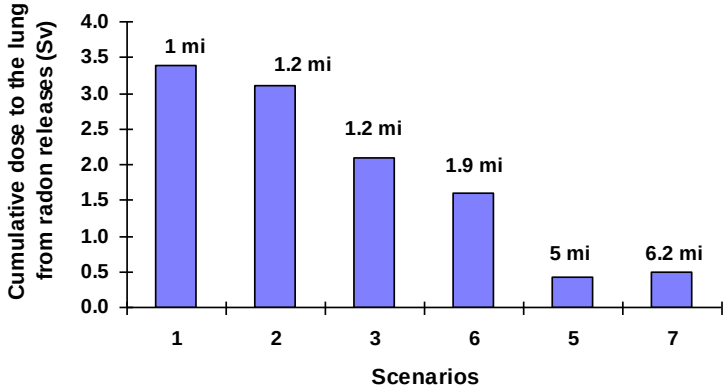



**Figure 47.** Effect of distance from the center of the FMPC on the cumulative dose to the representative individuals in the scenarios, who had 38 years of exposure to FMPC releases. The scenarios are listed in order of distance from the site; as a result, scenario 6 (1.9 mi) precedes scenario 5 (5 mi) on the graph.

Another way to look at the effect of distance and direction from the site on the radiation dose a person receives is by constructing a dose contour plot (Figure 48). In the figure, the dose contour curves are based on the subject of scenario 1, who was exposed to the releases for 38 years. The curves surrounding the site at increasing distances represent the cumulative dose from radon releases at the FMPC that the person in scenario 1 would receive if she lived at different distances and directions from the site. Each curve (contour line) is labeled with the cumulative radon dose that she would have received had she lived at any location on that curve, assuming all other characteristics were unchanged (for example, the locations of the schools she attended are unchanged).

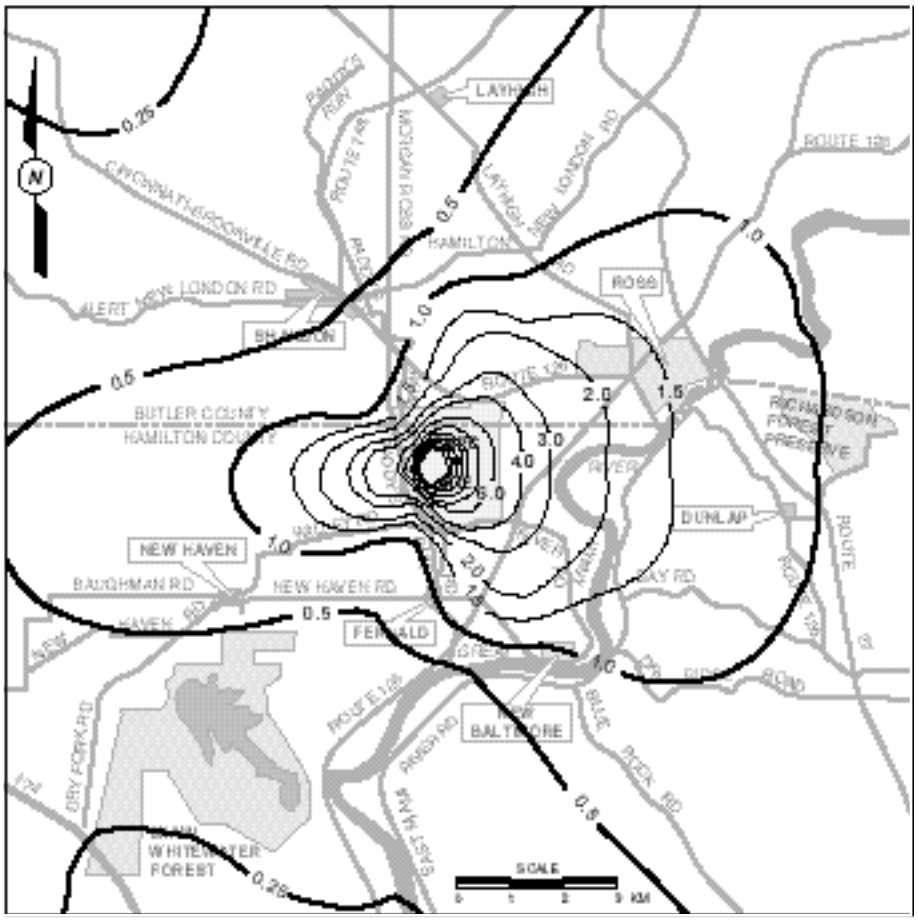



**Figure 48.** Cumulative radon dose (Sv) from the K-65 silo releases as a function of an exposed individual's dwelling location. The dose contour curves are based on the subject of scenario 1, who was exposed to the releases for 38 years. She was born in 1946, attended Elda Elementary School, Hamilton-Cleaves Middle School, and Ross High School in the 1950s and early 1960s. Otherwise, she spent most of her time at home. Her home was on Route 126, just northeast of the FMPC site, and the estimate of her cumulative radon dose for 38 years was 3.0 Sv (Appendix K). The contour plot allows us to see how this cumulative dose would have varied by moving her home about the region, but leaving fixed all other assumptions (including the locations of the schools she attended). For example, her cumulative dose would have remained the same if her home had been located anywhere along the 3.0-Sv contour line (except for segments of it that cut across the southwestern and northwestern corners of the site). The information in this plot is not applicable to workers at the FMPC site, because occupancy times and other assumptions would be different.

These scenarios include different lifestyles, diets, and residences. The results suggest that the most important characteristics that contribute to the dose of a person historically exposed to FMPC releases are the length of time a person lived near the site, the distance and directions from the facility, and whether the source of a person's drinking water was a contaminated well.

Residency time and distance from the site are the most important variables in calculating the dose to an offsite resident. Next in importance to dose are the direction from the site and whether well water containing uranium was drunk.

In addition to looking at the cumulative radiation doses from all years of exposure (Tables 22 and 23), we can look at the dose received year by year. Figure 49 shows the annual releases of uranium (Voillequé et al. 1995) and scenario 1 annual uranium doses to the lungs, bone surfaces, and kidneys from 1951 through 1988. Scenario 1 resulted in the highest doses of all the scenarios considered. The person of scenario 1 lived about 1 mi (1.7 km) northeast of the center of the site throughout the entire period of plant operation. The figure shows that her exposure to uranium was greatest during the 1950s and early 1960s when releases peaked. The dose curves gradually decline after the 1960s because releases decreased. There is a longer buildup and much more gradual decrease of annual dose to the bone surface compared to the other organs. This occurs because uranium is removed more slowly from bone than from soft tissues.

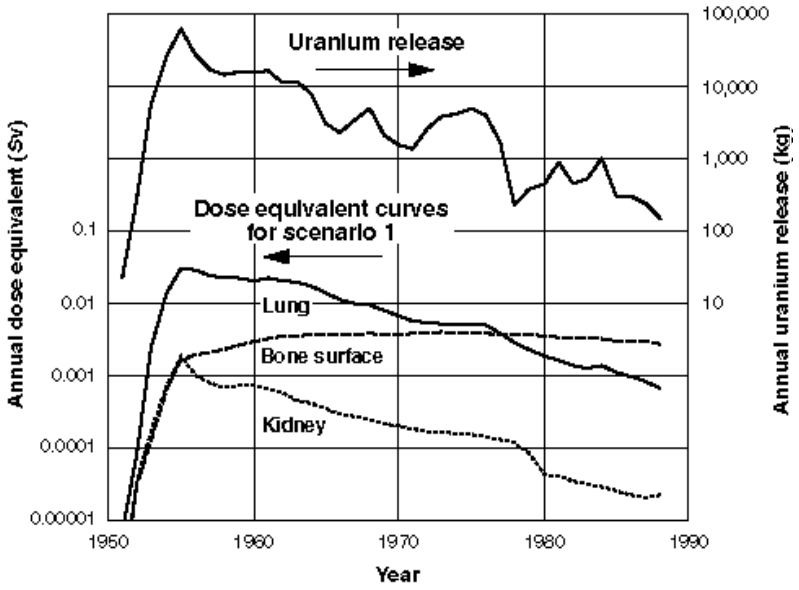



**Figure 49.** Uranium organ doses for scenario 1, shown with uranium releases from the FMPC. The dose response curves correlate generally with the uranium releases over time but are smoothed by the accumulation of residual dose resulting from delayed removal of the radionuclides from the organs. Bone surfaces in particular have a longer retention time for uranium than softer tissues and consequently show a response with a longer buildup and more gradual decrease.

In a similar fashion, Figure 50 compares radon released from the K-65 silos over time with the dose from radon decay products to the lung (TBE) for the person in scenario 1. This figure shows a much clearer correlation of annual radon dose with annual radon release because of the short half-lives of the radon decay products.

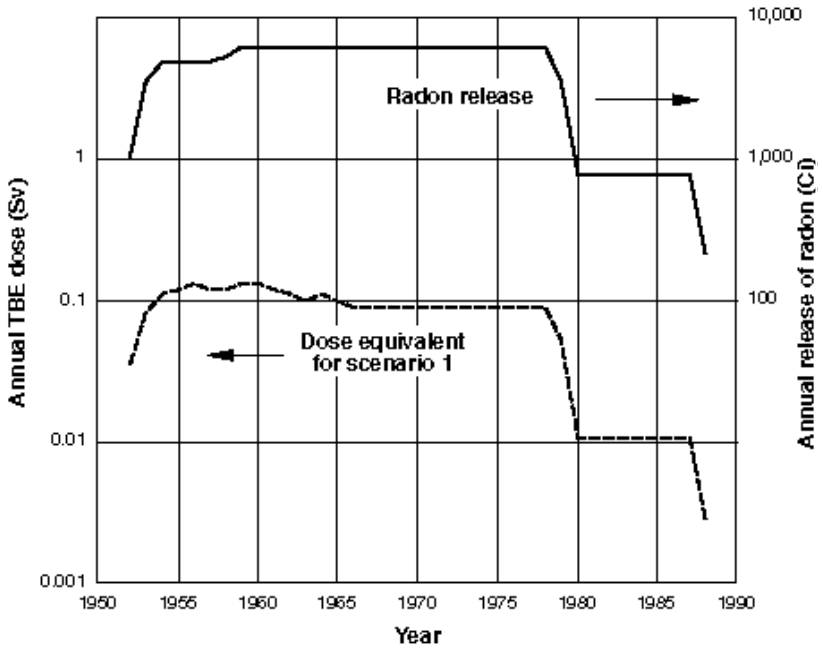



**Figure 50.** Radon dose to the tracheobronchial epithelium (TBE) for the person in scenario 1, shown with radon release from the K-65 silos from 1952 through 1988. The period before 1966 shows the effect of age dependence of the dose.

**Dose to the Embryo/Fetus**

The individuals in two scenarios (1 and 6) carried pregnancies to term during 1964-1965, at the age of 18-19 years. The dose to the embryo/fetus from radionuclides taken into the body of these pregnant women was estimated as the *absorbed dose* to the uterus of the woman during the nine-month gestation period. This is not a committed dose. After birth, the individual would continue to accumulate dose from radioactivity incorporated in tissues from placental transfer during gestation. Except for the time limitation to nine months, the calculation of this dose does not differ from that for other systemic organs. Table 24 lists the absorbed doses in units of gray (Gy) for the individuals in scenarios 1 and 6 who were assumed to be pregnant during their exposure to radioactivity released from the FMPC. Both the internal and external components are considered in the calculation. No dose to the embryo-fetus is expected from exposure of the mother to radon.

**Table 24. Median, or 50th Percentile Values, for Absorbed Dose to the Embryo/Fetus from FMPC Releases**

| | Internal dose(Gy) [a] | | External dose (Gy) [a] |
|---|---|---|---|
| Scenario | Low-LET | High-LET | Low-LET [b] |
| 1 | $1.1 \times 10^{-8}$ | $4.8 \times 10^{-8}$ | $1.2 \times 10^{-7}$ |
| 6 | $9.4 \times 10^{-9}$ | $3.9 \times 10^{-8}$ | $5.5 \times 10^{-8}$ |

[a] Absorbed dose for a 9-month period; 1 Gy = 100 rad; LET = linear energy transfer.
[b] For comparison, the population-weighted average background absorbed dose rate in air from terrestrial sources is $4.4 \times 10^{-4}$ Gy per year (12 months) (NCRP 1987).

## HEALTH EFFECTS FROM ESTIMATED DOSES

The release of radionuclides to the environment can expose human beings to ionizing radiation and cause harmful health effects. People may be exposed to radiation from external sources or from ingested or inhaled radionuclides. Incidence of health effects depends on the amount of dose received. This section will examine the potential health effects that may result from the doses due to uranium and thorium, and to radon described in the previous section. There are two main classes of health effects: *deterministic* and *stochastic* effects.

Deterministic effects cause direct damage to tissues and include effects that most often occur within days to weeks after exposure. For example, deterministic effects can cause reddening of the skin, cataracts, hair loss, sterility, and bone marrow depression. Threshold doses for acute exposure must be exceeded for deterministic effects to show. Because the threshold doses for most of these effects are quite high (a minimum of 0.1 to 0.5 Sv and many are much higher, delivered in a short time) (NCRP 1991), deterministic effects generally would not occur from environmental releases, except after accidents involving high doses. Thus, deterministic effects are not usually a factor in cases where relatively low levels of contamination or releases of small quantities of radionuclides to the environment may have occurred.

Stochastic effects are assumed to occur randomly at all dose levels, including the lowest doses. The frequency of stochastic effects is dependent on the dose, and effects usually occur at long intervals after exposure. In a large population exposed to low doses only a few of the exposed individuals will be affected. The two principal types of stochastic effects are induced cancer and genetic effects. People exposed to radiation are several times more likely to be affected by an induced cancer than to transmit genetic effects to their children. Therefore, induced cancer is usually given greater attention. However, the possibility of genetic effects should be considered.

Some cancers are more likely to be induced than others, and the latency period (the period between the exposure and the expression of a cancer) also varies with the cancer type. For example, leukemia may occur from 2 to 30 years after exposure. Solid tumors, such as those of the lung, colon, stomach, esophagus, liver, bladder, breast, skin, and thyroid, can begin to occur only 5 to 10 years after exposure and can still arise after 40 years or more.

**The health effects of most interest to this study are stochastic effects: induced cancer and genetic effects. Cancer induction in those exposed to radiation is much more likely to occur than genetic effects in their children and, consequently, is given greater attention.**

Most of our detailed information about the

induction of cancer in irradiated populations, including the quantitative estimates of risk in organs and tissues, the age and sex dependence, the time relationships, and latencies, comes from the experience of the atomic bomb survivors in Japan who have been studied for over 40 years. The results of these studies are supported with generally similar results from other studies, mainly of medically exposed populations and some occupationally exposed populations.

To estimate the risks after exposures occur in a population, we first estimate the dose (to relevant organs and tissues) and then apply appropriate risk factors. These risk factors may be derived from study of the atomic bomb survivors, or, as in the case of radon, from direct epidemiological studies of populations exposed to radon.

Later in this report, we consider whether any other health effects of the radionuclides such as chemical toxicity can be a significant factor at the levels of exposure identified. This is an important consideration in the case of uranium.

**Radiation Exposure and the Risk of Cancer**

The equivalent dose to an organ is the main factor determining the probability of an induced cancer. After an initial latent period, such a cancer can occur at any time throughout the life of the individual, and the probability that one will occur is called the lifetime risk of a radiation-induced cancer.

The lifetime risk of getting a cancer in *any* organ (that is, the total risk of cancer after whole body exposure) is about 0.05 per Sv (that is, 5 in 100 or 5% per sievert) for an average member of a population of all ages (Table 25). Adults have a smaller risk, 0.04 per Sv (4% per sievert) and adults older than 65 a smaller risk still, 0.01 per Sv (1% per sievert). Children have about twice the risk of an average member of the population, 0.1 per Sv (10% per sievert). The risk of cancer being induced in a specific organ is less than that of cancer being induced in *any* organ and varies with the organ. For the principal organs in which cancer is likely to be induced by ionizing radiation, the risks are given in Table 25.

The numbers in Table 25 are averages for both sexes. Individual organ and tissue risks for the whole population (and for workers) are based mainly on the atomic bomb survivor data and are specified for use in radiation protection internationally by the International Commission on Radiological Protection (*ICRP*) (ICRP 1991) and in the U.S. by the NCRP (NCRP 1993). Information on children is specified by NCRP (NCRP 1993) and for older adults is available from Sinclair (1992) based on Land and Sinclair (1991).

Estimates of cancer risks involve many uncertainties, and in the case of the total risk of fatal cancer (0.05 per Sv), these uncertainties lead to estimates of risk per unit dose that can be higher or lower by a factor of about two to three (NCRP 1997). That is, the 5th and 95th percentiles of the risk may be about 0.02 to about 0.10 per Sv. Uncertainties in individual organ risks are generally larger. In addition, it is necessary to combine these uncertainties with the uncertainties in the dose estimates.

**Table 25. Probability of Fatal Cancer in Organ and Tissue Sites and for Whole Body Exposure**

| Tissue or Organ | Average probabilities of fatal cancer in a population of all ages [a] (% per sievert) |
|---|---|

| | |
|---|---|
| Bladder | 0.30 |
| Bone marrow (Leukemia) | 0.50 |
| Bone surface | 0.05 |
| Breast | 0.20 |
| Colon | 0.85 |
| Liver | 0.15 |
| Lung | 0.85 |
| Esophagus | 0.30 |
| Ovary | 0.10 |
| Skin | 0.02 |
| Stomach | 1.10 |
| Thyroid | 0.08 |
| Remainder | 0.50 |
| Total (Whole body) | 5.00 |

[a] (ICRP 1991, NCRP 1993); for adult workers multiply by 0.8; for children multiply by 2 (NCRP 1993, Sinclair 1992); for adults older than 65 multiply by 0.2 (Sinclair 1992); numbers are averages for both sexes.

**Most Important Organs Exposed to the Releases at Fernald**

When the whole body is uniformly exposed to ionizing radiation, all the organs and tissues listed in Table 25 are exposed to approximately the same dose. However, it is more usual with radionuclide releases that some organs and tissues are exposed to higher doses than others. In the case of the releases at the FMPC, which are due mainly to airborne releases of radon and isotopes of uranium and thorium (Figure 45), doses to a few organs and tissues were expected to dominate. The seven exposed organs considered are the lungs (particularly the TBE region), bone surface, kidney, liver, red marrow, testes, and ovaries. If exposure by ingestion had been primary, the list of organs would have been different.

**The increased risk of lung cancer dominates all other radiation-related health effects from FMPC releases.**

The most important organs to consider depend on a number of factors: (a) the organs most likely to get cancer per unit dose (Table 25), (b) the organs receiving the highest dose from inhalation, (c) the organs receiving the highest dose from ingestion (ICRP 1995), and (d) possible chemical toxicity.

Among the seven organs selected on this basis are the testes and ovaries. However, the doses to these organs are only about 0.001 Sv or less. Furthermore, the risk of cancer in the ovaries is 0.001 per Sv. At a dose of 0.001 Sv, the risk of cancer to the ovaries is less than 1 in a million, and the risk of cancer in the testes is much less. The risk of genetic effects is 0.024 per Sv (ICRP 1991), and for a dose of 0.001 Sv it is 0.000024 or about 2½ in 100,000, a much smaller risk than those to be considered later. Consequently, risks of exposure to the testes and ovaries are not considered further.

**Lifetime Risks**

The lifetime risks associated with a given equivalent dose to an organ resulting from ex-

posure to uranium and radionuclides other than radon are taken from Table 25 for four of the seven selected organs, namely, lungs, bone surface, liver and bone marrow. For kidneys, the risk is not given in Table 25 because it is not stated separately by ICRP and NCRP. Instead, the risk for kidneys was determined separately from the atomic bomb survivors data as 0.0008 per Sv (0.08% $Sv^{-1}$) (Appendix S). Testes and ovaries are not considered further because of the small risk of cancer or genetic effects from exposure of these organs to radiation, as noted above. The embryo-fetus is not considered further either because of the very small dose to be expected (Table 24).

The risk of lung cancer for radon is assumed to be 0.0027 $Sv^{-1}$ (0.27% $Sv^{-1}$) based on direct epidemiological data as assessed by the Biological Effects of Ionizing Radiation (*BEIR*) IV Committee (NAS/NRC 1988) and supported by other evaluations, such as ICRP (ICRP 1993) (Appendix S).

### Epidemiological Experience with Uranium

No previous environmental exposure situations involving uranium, have been studied adequately to quantify the health effects from exposure of the public to uranium. However, nuclear energy workers in certain phases of the nuclear energy cycle have been exposed to uranium in various forms. Recent reviews of epidemiological experience with uranium appear in the BEIR IV report (NAS/NRC 1988) and in the United Nations Scientific Committee on the Effects of Atomic Radiation report for 1994, Annex A (UNSCEAR 1994). Neither assessment finds a clear association between uranium exposure and cancer risk. A later paper on the evaluation of these workers (Dupree et al. 1995) also finds no clear association between uranium exposure and excess cancer. Nevertheless, approximate estimates of the risk of lung cancer from the few data available are broadly similar to those derived from the atomic bomb survivor data (see Appendix S).

## LIFETIME RISKS OF CANCER FOR EACH SCENARIO

### Individual Exposure Scenarios

This report considers nine exposure scenarios involving representative residents at different locations in the FMPC area. For each scenario, the equivalent doses (from Table K–2, Appendix K) for the five principal organs identified above (lungs, bone surface, kidney, liver, and red marrow) for each pathway (inhalation, ingestion and external exposure) have been estimated and added to estimate the total equivalent dose to that organ. Doses less than 0.0001 Sv cannot contribute a risk greater than 0.000001 per Sv and are, therefore, not included. Some doses from external exposure are in this category and are not included. Radon exposure to the lungs from the K-65 silos was estimated separately from uranium as indicated above. Scenarios 1, 2, and 3 feature primary locations within about 1.2 mi (2 km) from the center of the FMPC production area; scenarios 5, 7, and 9 are based on primary locations that are 5 to 6.2 mi (8–10 km) from the site; and scenarios 4, 6, and 8 are approximately 2 to 2.5 mi (3–4 km) from the site (Table 21 and Figure 43).

### Lifetime Risks of Cancer Derived for Each Scenario

The procedure outlined above was applied in the case of each of the nine scenarios using dose data for uranium and other radionuclides and for radon from Appendix K, Table K–2. Appendix S provides detailed calculations of risk for each of the organs in each of the scenarios. We assumed that the risk is proportional to the dose in the low dose region. Because the individuals in the scenarios were young during a portion of their exposure, we considered whether an adjustment factor for age and sex should be included. The issue was considered separately for each organ but is only important for lung cancer. In this case, in contrast to some other organs, the evidence does not suggest that young individuals are at greater risk for lung cancer induction than older individuals. Nor is there a significant sex difference. Consequently, no adjustment for age and sex was made except in the case of bone where younger males are at greater risk than older males or females. The risk of bone cancer is very small, however. The 50th percentile results of the calculations are summarized in Table 26 for lung cancer and total cancer for all nine scenarios. These point estimates show that radon is always the dominant component and that the risk of lung cancer is virtually all of the cancer risk. Risks of induced cancer in other organs are small.

**Adjustment factors for age and sex were considered. They were found not to apply in the important case of the lung. No adjustments for age and sex were made except to bone cancer risk, which is small.**

### Uncertainties in Risk Estimates

Risk estimates made for the individuals in the nine scenarios are subject to many uncertainties. Principal among them are two: first, the uncertainty in the dose which is described by a distribution such as that in Figure 44 from which 5th, 50th, and 95th percentiles are available, and second, the uncertainty in the risk per unit dose. This latter uncertainty is different for uranium and thorium radionuclides on the one hand and for radon on the other. The various components of the uncertainties have been analyzed and combined separately in the two cases, in Appendix S. The results of the combined uncertainties in the risk per unit dose and in the dose are further combined for each source, and then for both sources. The parameters of the final distribution of the risk of lung cancer, which includes all identified uncertainties, is shown in Table 27. These risks are presented for lung cancer only, which is 98% of the total risk (see Table 26), and uncertainties in the remaining 2% would not affect the result in any meaningful way. The 50th percentile values of the risk of fatal lung cancer range from $0.11 \times 10^{-2}$ (scenario 8) to $1.3 \times 10^{-2}$ (scenario 1) or from 0.11% to 1.3%. It is clear, however, that there is a small chance of a risk as high as 9.6% (scenario 1, 95th percentile) and also a small chance of risks as small as 0.02% (scenario 8, 5th percentile).

**Table 26. Lifetime Risk of Fatal Cancer Estimated for Each Scenario Based on 50th Percentile Doses**

| | Risk of lung cancer (%) | | Total risk of | Risk of cancer in | Total cancer |
|---|---|---|---|---|---|
| | Sources other | | lung cancer | other organs | risk |
| Scenario | than radon[a] | Radon | (%) | (%) | (%) |

| | | | | | |
|---|---|---|---|---|---|
| 1 | 0.349 | 0.972 | 1.321 | 0.019 | 1.340 |
| 2 | 0.119 | 0.972 | 1.091 | 0.009 | 1.100 |
| 3 | 0.111 | 0.702 | 0.813 | 0.047 | 0.860 |
| 4 | 0.043 | 0.405 | 0.448 | 0.011 | 0.459 |
| 5 | 0.035 | 0.113 | 0.148 | - | 0.148 |
| 6 | 0.187 | 0.594 | 0.781 | 0.011 | 0.792 |
| 7 | 0.033 | 0.103 | 0.136 | 0.003 | 0.139 |
| 8 | 0.007 | 0.113 | 0.120 | - | 0.120 |
| 9 | 0.065 | 0.227 | 0.292 | 0.003 | 0.295 |

[a] Radionuclides, especially uranium and thorium, see Appendix K, Figure K-1.

**Table 27. Risks of Lung Cancer and Uncertainties in Risks from All Radiation Sources at Fernald**

| | Risks (%) | | | | | |
|---|---|---|---|---|---|---|
| | Nominal val- | Percentiles | | | | |
| Scenarios | ues of risk | 5th | 25th | 50th | 75th | 95th |
| 1 | 1.1 | 0.24 | 0.7 | 1.3 | 2.5 | 9.6 |
| 2 | 0.9 | 0.18 | 0.51 | 1 | 2.2 | 6.8 |
| 3 | 0.65 | 0.15 | 0.41 | 0.8 | 1.4 | 4.9 |
| 4 | 0.35 | 0.08 | 0.23 | 0.41 | 1.1 | 2.9 |
| 5 | 0.11 | 0.025 | 0.075 | 0.16 | 0.32 | 0.9 |
| 6 | 0.69 | 0.15 | 0.38 | 0.75 | 1.6 | 4.2 |
| 7 | 0.096 | 0.026 | 0.072 | 0.15 | 0.28 | 1 |
| 8 | 0.1 | 0.02 | 0.047 | 0.11 | 0.26 | 1.1 |
| 9 | 0.2 | 0.046 | 0.14 | 0.31 | 0.59 | 2.5 |

These risks are for individuals not specified as smoker or nonsmoker. For a smoker the risks would be 1.41 times higher and for nonsmokers 0.71 times lower, i.e., 50th percentile values for scenario 1 of 1.8% and 0.9%, respectively. The 95th percentile values could reach 13.5% for a smoker and 6.8% for a nonsmoker, whereas the 5th percentile values would be 0.34% and 0.17%, respectively.

### The Significance of the Estimates of Fatal Lung Cancer Risk in Each Scenario

For four scenarios 1, 2, 3 and 6, the 50th percentile value of lung cancer risk is about 1%, the 95th percentile risk is close to 10% and the 5th percentile risk about 0.1 to 0.2%. These risks are not negligible. The 50th percentile risks are about the same order as natural radiation background risk in a lifetime (1.25% vs 1.3% for scenario 1).

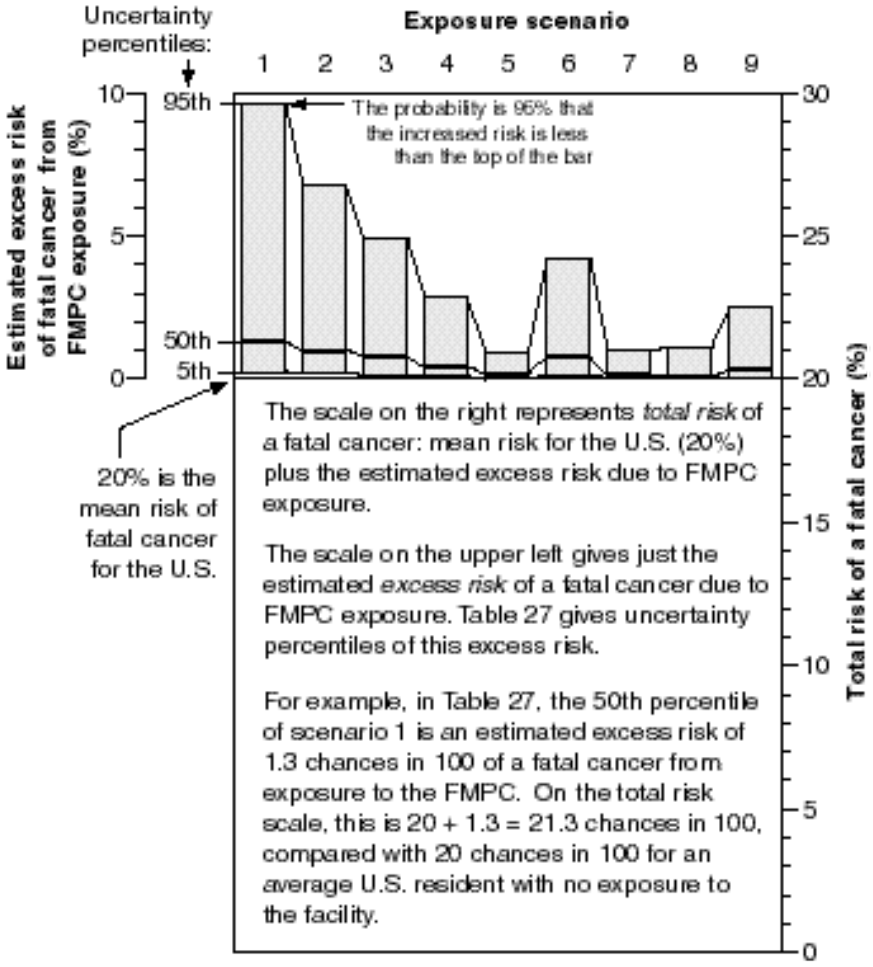



**Figure 51**. Comparison of the risk of a fatal cancer for each scenario to the natural background risk of fatal cancer of approximately 20%. The median or 50th percentile estimate is represented by the heavy line and the 95th percentile value is the top of the open bar. For scenario 1, the highest exposure scenario, the median estimate of the increased risk for a fatal cancer is about 1.3 chances in 100 or 1.3%. The uncertainty range for the increased risk is from 1 chance in 100 (1%) to 9 chances in 100 (9%). Adding the 1.3% increased risk to the 20% background risk, the person in scenario 1 now has a median risk of 21.3%.

The risk of lung cancer in scenario 1 is about twice the natural risk of a nonsmoker and about one-sixth of that of a smoker. This fatal lung cancer risk can also be compared with the risk of cancer from the average natural background radiation (including radon and average exposure to man made sources) in the U.S. in a lifetime. The lifetime equivalent dose from natural background radiation is about 0.25 Sv (NCRP 1987), with a risk of about 0.0125 (1.25%) in a lifetime, or 1 in 80 (Appendix S). Natural background radiation varies somewhat with location in the U.S. depending on altitude and radon level. The highest-exposure scenario 1 evaluated for the FMPC entails a fatal cancer risk (1.3% at the 50th percentile) that is about the same as the risk due to the average natural background radiation in the U.S (1.25%). The least important scenario (8) evaluated for the FMPC, has risks 10 times less than the scenario (1) highest risks. Table 28 compares these risks.

**Table 28. Comparison of Lifetime Fatal Cancer Risks**

| Cancer | Natural background risk | Natural Background Radiation risk-U.S. average | Highest FMPC exposure scenario (50th percentile) | Lowest FMPC exposure scenario (50th percentile) |
|---|---|---|---|---|
| Lung cancer | | | 1.3% | 0.11% |
| Smoker [a] | 9 % | 0.8% | | |
| Nonsmoker [b] | 0.9% | | | |
| All cancers [b] | ~20% | 1.25% | 1.3% | 0.11% |

[a] Male ~12% female ~6%, NAS/NRC 1988.
[b] Male 1.1% female 0.6%, NAS/NRC 1988.

For scenario 1, at the 5th percentile of the risk estimate (0.24%, Table 27), the risk is equivalent to about 1.2% (0.24%/20%) of the risk of dying of cancer naturally. At the 95th percentile of the risk estimate (9.6%, Table 27), the risk is equivalent to about 48% (9.6%/20%) of the risk of dying of cancer naturally. Again, the 95th percentile risk estimate for scenario 1 is approximately equal to the natural risk (9%) of dying of cancer for a smoker (Table 28).

Another useful comparison is with lung cancer resulting from domestic radon exposure. The average concentration of radon in homes in the U.S. is about 1 pCi $L^{-1}$ with a risk of about 0.5% lifetime. The EPA has an action level of 4 pCi $L^{-1}$ for the radon concentration in homes, with a corresponding risk of about 2% lifetime. The 50th percentile risk of lung cancer for scenario 1 is an additional risk greater than the average risk of domestic radon concentration but not much more than half of the risk at the EPA action level (1.3% vs 2.0%). On the other hand, the 95th percentile is 5 times the risk at the EPA action level and the 5th percentile is about 1/16th of the risk at EPA action level.

## DETERMINISTIC EFFECTS

In the scenarios considered, the main hazard from releases was an increased risk of lung cancer from radon. The highest exposures to the lungs are in scenarios 1 and 2, where the doses were received over a very long period of time. Deterministic effects in the lungs have thresholds, the lowest of which is 4 Sv (acute exposure) (NCRP 1991) for pneumonitis. Pneumonitis is an acute reaction. Chronic exposures spread over many years, even if total doses were several times the threshold, would not be expected to cause pneumonitis. In this case, for scenario 2, the central estimate for the chronic dose from FMPC effluents is below the acute dose threshold, and for scenario 1, it is almost equal to it (see Table 22). The 95th percentile chronic dose estimates for scenarios 1, 2, 3, 4, and 6 all exceed the threshold for acute exposures (Table 22), with scenario 1 having a 95th percentile dose about three times the threshold. However, because the dose would have been received over many years, no acute effects are expected, and at these dose levels no late deterministic effects such as fibrosis are expected either.

## THE TOXICITY OF NATURAL URANIUM

Uranium is widely distributed in soil and in some minerals in the earth's crust. It is taken up in crops and is transferred to humans mainly through the food chain. Normal levels in the human body are in the range 2–62 micrograms (μg) uranium per gram (g) of tissue. At much higher levels uranium is chemically toxic to humans and the principal target organ is the kidney.

**This study focuses on past releases of radioactive materials from the FMPC, and not specifically on releases of toxic chemicals. However, uranium is also chemically toxic to the kidney at high enough levels. Therefore, the chemical toxicity of uranium has been considered.**

The toxicity of natural uranium to the kidney depends upon the concentration in the kidney (NAS/NRC 1988, Spoor and Hursh 1973, Alexander 1988). At concentrations of up to 0.5 μg $g^{-1}$, the effects appear to be mild and possibly reversible. Others describe a threshold ranging from 0.1 μg $g^{-1}$ to 1 μg $g^{-1}$, with the severity of effect increasing in this range (Morris and Meinhold 1995). This means no effects are expected below 0.1 to 0.2 μg $g^{-1}$, mild effects might be seen at about 0.5 μg $g^{-1}$, and more severe effects could appear beginning at 1 μg $g^{-1}$.

Calculations were made to estimate maximum levels of uranium concentration (μg of uranium per gram of kidney tissue) in the kidneys of the subjects of nine exposure scenarios, defined in Appendix J, during and subsequent to their periods of exposure. The calculation applies the ICRP Publication 69 (ICRP 1995) retention function for uranium in the kidneys to simulate a dynamic level (μg $g^{-1}$) over time. The simulation is driven by the estimated annual intake by ingestion and inhalation as functions of time for the period of the subject's exposure. The calculation takes into account the variation of kidney metabolism, mass, breathing rate, water consumption, and dietary intakes with age and, where the data support the distinction, sex.

**The concentration of uranium in the kidney determines its effect.**

**Scientific studies show:**
- **No toxic effects .....<0.1 to 0.2 μg $g^{-1}$**
- **Mild effects.....0.5 μg $g^{-1}$**
- **More serious effects ..... 1 μg $g^{-1}$**
- **Widely used toxic threshold..... 3 μg $g^{-1}$**

**For this study, the median estimate of the maximum level of uranium in the kidney (scenario 3).....0.75 μg $g^{-1}$**

The 50th and 95th percentiles of the concentrations of uranium in the kidney are given for each scenario in Table 29. Figure 52 compares the median estimate of the maximum uranium concentrations in the kidneys of the people represented by the nine scenarios and the toxicity expected at various concentrations of uranium.

An episodic (accidental) release of uranium hexafluoride on February 14, 1966 could have produced a maximum concentration in the kidney which was above the no-effect level. This episodic release is reviewed in Appendix T. The release time was short (1 hour). We estimated the possible effects for a person who was directly downwind during the accident.

It is apparent from the final two columns of Table 29 that in four of the exposure scenarios (5, 7, 8, and 9) neither the 50th nor 95th percentile concentrations of uranium ever exceeds 0.1 μg $g^{-1}$, and therefore there are no nephrotoxic effects of any kind to be expected in representative individuals in these scenarios in any circumstance. Two additional scenarios (4 and 2) have 50th percentile values at 0.038 μg $g^{-1}$ and 0.092 μg $g^{-1}$ respectively and 95th percentile values at 0.12 μg $g^{-1}$ and 0.26 μg $g^{-1}$ and there is virtually no chance of even mild effects in the kidney in these two scenarios.

**Table 29. 50th and 95th Percentiles of Uranium Concentration (μg $g^{-1}$) in the Kidneys of Subjects of the Nine Scenarios**

| | Percentile | |
|---|---|---|
| Scenario | 50th percentile | 95th percentile |
| 1 | 0.28 | 1.0 |
| 2 | 0.092 | 0.26 |
| 3 | 0.75 | 1.4 |
| 4 | 0.038 | 0.12 |
| 5 | 0.0025 | 0.010 |
| 6 | 0.14 | 0.52 |
| 7 | 0.012 | 0.033 |
| 8 | 0.001 | 0.0029 |
| 9 | 0.0055 | 0.024 |

Scenarios 6, 1, and 3 are more problematic. Scenario 6 has a 50th percentile of 0.14 μg $g^{-1}$ and a 95th percentile of 0.52 μg $g^{-1}$. The latter is just into the range where mild effects are possible but with a low likelihood. Scenario 1 has a 50th percentile value of 0.28 μg $g^{-1}$ and a 95th percentile value of 1.0 μg $g^{-1}$. In this scenario mild effects are quite likely and there is a small chance of more severe effects. Scenario 3 has the greatest uncertainties with 0.75 μg $g^{-1}$ at the 50th percentile and 1.4 μg $g^{-1}$ at the 95th percentile. Thus, mild effects are likely and there is a small chance of more severe effects. However, it should be pointed out that severe effects in humans beginning at 1 μg $g^{-1}$ are inferred rather than directly known from experience.

In conclusion, it would seem that no chemical toxicity effects are to be expected for scenario 2, 4, 5, 7, 8, and 9. There is a small chance of mild effects in scenario 6. In scenarios 1 and 3 mild effects are likely and there is a small chance of more severe effects in the kidney.

The chemical forms of the uranium in the 1966 accident were quite soluble in body tissues, resulting in a low radiation dose but a relatively large amount reaching the kidney for a short time. The estimated maximum concentration in the kidney for each of the scenarios is shown in Figure 52 along with the estimated concentration for the episodic exposure scenarios.

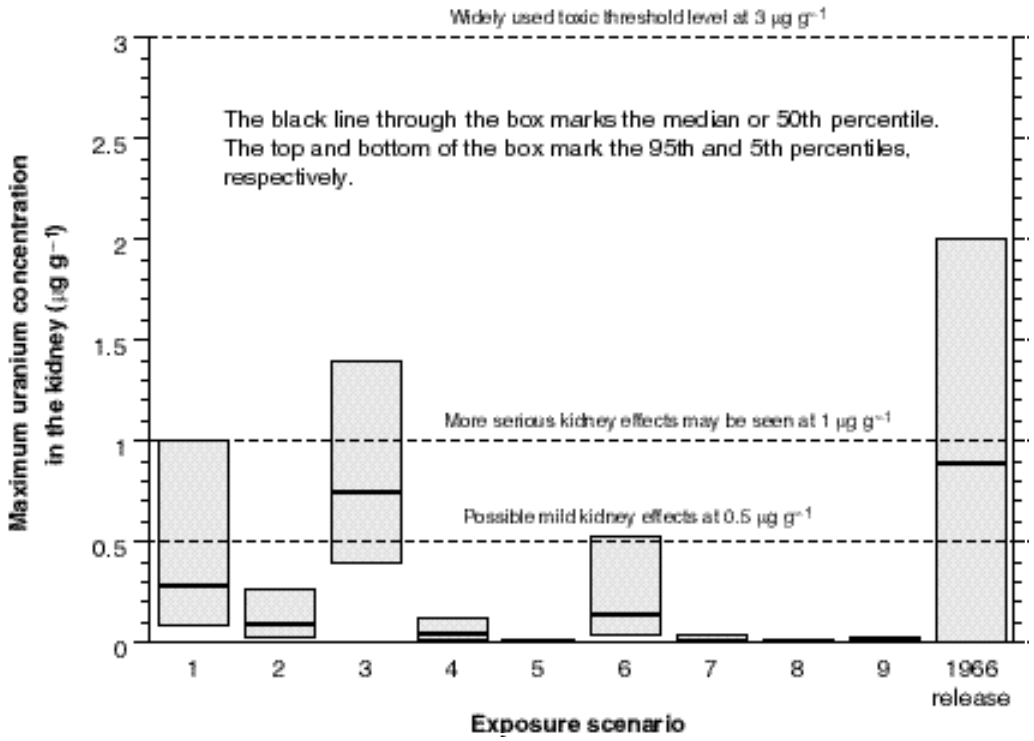



**Figure 52.** Estimates of the maximum uranium concentration in the kidney for each of the nine exposure scenarios and for the accidental release in 1966. The uranium concentration in the kidney, shown here for each scenario, is the highest concentration expected in the kidney from the intake of uranium over the time period associated with each scenario (Table 29). The figure compares the uranium levels in the kidney with the toxicity of natural uranium. The scientific evidence suggests no effects measured in the kidneys when the uranium concentration is below 0.1–0.2 μg $g^{-1}$; possible mild effects at about 0.5 μg $g^{-1}$, more serious effects in animals at 1 μg $g^{-1}$, and some renal damage observed in humans beginning at 3 μg $g^{-1}$ .

## CONCLUSIONS ON THE RISKS FROM FMPC RELEASES

This review of the health impacts of the exposure of people represented by the nine scenarios to releases in the vicinity of FMPC has resulted in a number of conclusions.

1. There is virtually no likelihood of direct tissue damage to the lung or any other organ of individuals in any of the scenarios due to radiation exposure.
2. Chemical toxicity from uranium in the kidney is unlikely in scenarios 2, 4, 5, 7, 8, and 9. If the 95th percentiles are considered, there is a small chance of mild effects in scenario 6. In scenarios 1 and 3 mild effects in the kidney are possible and there is a small chance (at the 95th percentile) of more severe effects in the kidney. The estimated uranium concentration in the kidney following the acute episodic release in 1966 was higher than for any of the chronic scenarios.
3. The doses to the gonads are very low, and the resulting genetic risk is very small.
4. There is a risk of cancer, almost entirely lung cancer, for all scenarios. For the scenarios with the greatest exposures, these risks are not negligible.
5. The median, or 50th percentile, estimate of the lifetime fatal cancer risk identified for the

scenario providing the largest dose (scenario 1) is about 1.3%. This would increase the natural risk of fatal cancer in a lifetime (20%) to 21.3%, i.e., by about 7%. For lung cancer alone, the risk is about twice that of a nonsmoker but less than one-sixth that of a smoker. The risk for the individual in scenario 1 is about the same risk attributed to the average natural background exposure over a lifetime (that is, for this person, this amounts to doubling the natural background risk). Some of the scenarios show risks about 10 times lower.

6. When the 5th to the 95th percentile of the uncertainty distribution is considered, the estimated risk for scenario 1 is 0.24% to 9.6%, respectively. The 95th percentile risk is 48% of the natural risk of fatal cancer or about equal to the risk of lung cancer for a smoker.

## PUTTING RISKS INTO PERSPECTIVE

For some individuals, risk comparisons may help them put the risks from the Fernald releases into perspective. Some people may be able to better evaluate their potential risks from historic releases from Fernald within a framework of risks from similar contaminants. However, the challenge is to find appropriate risk comparisons. Risk estimates for any activity or disease process are for populations of people, not for individuals with the populations but the comparisons may give direction to some individuals to use in making decisions about their potential risks from Fernald releases. Most importantly, comparing risks from several events or contaminants can be difficult and should never be used to trivialize or to exaggerate the risks that have been calculated in this study from FMPC releases. Risk estimates or comparisons can never measure the pain and suffering or emotional impact of those injured or of family members left behind.

Comparing risks directly can be problematic because the bases for two risk estimates that are compared may be different. Some risk estimates are based on numbers of known deaths associated with a particular activity or disease. For example, one way for determining risks from automobile travel over a certain distance and time is based on the number of individuals killed in a certain population. These values are combined into a risk estimate for death from automobile driving.

The cancer risks calculated for the Fernald releases are a different type of risk estimate. These risks are projected lifetime risks based on animal studies and on information about the effects of high doses of radiation on people. The risks from high doses of radiation are then *projected* or extrapolated to what is expected to be seen with low doses of radiation, assuming that the dose response relationship is the same. As is apparent, it is not appropriate to directly compare the risks of death from driving to lifetime risks of fatal cancer from the past Fernald releases. In comparing risks, or putting them into perspective, it is important to compare similar activities, contaminants, and risk values. Because the highest risks of cancer from Fernald releases were almost entirely lung cancer, looking at risks from contaminants that affect the lung may be one way of putting the Fernald risks into some context.

When a person is exposed to smoke in the environment involuntarily, the term passive smoking is used. The passive or secondhand smoke can come from the smoke released from

the burning tobacco or from smoke exhaled by the person who smokes. In a report released in 1993, the EPA concluded that in adults, passive smoke is a class A (known human) carcinogen responsible for about 3,000 lung cancer death annually in U.S. nonsmokers (EPA 1993). About 60% of the U.S. population (265 million) are nonsmokers (160 million nonsmokers). Therefore, the total lung cancer risk from secondhand, or passive smoking is about $2\times10^{-5}$ (3,000/160 million) among nonsmokers.

The risk from passive smoke is shown in Figure 53 (modified from Voillequé 1996), along with the lifetime fatal cancer risks for all cancers in the U.S. (0.20 or 20%), and for fatal cancers caused by natural background radiation (0.0125 or 1.25%). Natural background radiation does vary with location in the U.S. depending on altitude and radon level. EPA regulatory risks are also provided in the figure. The 50th percentiles, or median estimates of the risk of fatal cancer from the Fernald releases are shown for scenarios 1 and 8 as the shaded lines on the graphic. For these two scenarios, the 5th to 95th percentile distributions are shown at the left. The highest doses were calculated for the individual represented by scenario 1 and the lowest median dose was calculated for scenario 8. The fatal cancer risk for the highest-exposure scenario 1 evaluated for the FMPC (0.013, or 1.3% at the 50th percentile) is about the same as the fatal cancer risk attributed to the average natural background radiation in the U.S. (0.0125, or 1.25%). The 95th percentile risk estimate for scenario 1 is approximately equal to the risk (0.09 or 9%) of dying of lung cancer for a smoker.

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**Addendum to Volume I**

# ISSUES RAISED BY THE NATIONAL RESEARCH COUNCIL REVIEW OF THE TASK 6 DRAFT REPORT

## INTRODUCTION

A draft version of this task 6 report was reviewed by the National Research Council Committee on an Assessment of CDC Radiation Studies. The review was published by the National Academy Press (NAS/NRC 1997). Radiological Assessments Corporation (*RAC*) published a detailed response (Killough et al. 1997), expressing disagreement with many of the NRC committee's criticisms but accepting some of the committee's points. At a meeting in Washington, D.C., on March 31, 1997, members of the *RAC* research team and CDC staff members presented additional material and answered questions posed by NRC committee members. Subsequently, the committee issued a letter to Dr. James M. Smith, Chief of the CDC's Radiation Studies Branch, stating the committee's sense of that meeting. The committee acknowledged mistakes in its review of the draft task 6 report and requested that some additional material be incorporated into the final report. The letter is reproduced at the end of this section.

This final report contains responses to all points raised in the NRC committee's letter. Details of some responses occur elsewhere within the text of Volume I and in the appendices of Volume II, at places where the substance of the responses can be placed in proper context. This section summarizes the responses and indicates where additional information can be found.

## SPECIFIC RESPONSES

### Summary Presentation of the Method for Estimating the Radon Source Term

The primary discussions of the method developed for estimating radon releases from the K-65 silos (and to a relatively minor extent, from drums on a pad in the production area during the 1950s) were given in considerable detail in the Task 2/3 report (Voillequé et al. 1995), and that report was cited in the draft task 6 report. At the meeting, *RAC* provided the NRC committee with a two-page summary and diagram of the method, which the committee members found helpful. The committee has requested that a similar presentation be incorporated into the final report, and we have provided such a discussion near the end of Appendix Q (page Q-30).

### Expansion of the Uncertainty for Radon Releases During 1959–1979

The NRC committee expressed a belief that the *RAC*-derived uncertainty distribution for radon releases from the K-65 silos during 1959–1979 was too narrow (i.e., implied too little uncertainty). This propagated distribution depends primarily on the uncertainty in the average radon concentration in the silo headspaces after the silos were sealed in mid-1979 and before 1988, when further work was done. The estimate of this concentration was based on measurements made during a single day in 1987. We acknowledge that temporal and (possibly) spatial limitations in the sampling could have introduced some bias into the estimated

concentration, although we were never persuaded that such a bias could have affected the release estimates to the extent that the committee seems to fear.

Even so, we have substantially increased the variance in this parameter by replacing the distribution used in the draft report (which was based on sample statistics) by a lognormal uncertainty factor of two for this concentration (geometric standard deviation 1.52). The temporal variability in the concentration would depend on the expansion and contraction of gases in the silo headspace. To provide some perspective for this change, we simulated this diurnal variation using hourly temperature records for 1987 from the Cincinnati airport and a correlation between measurements of headspace gas expansion in the K-65 silos and corresponding records of temperature change at the Cincinnati airport (Voillequé et al. 1995). Our simulation calculated daily changes in concentration in the headspace based on the correlation equation, the temperature data for 1987, and other parameters reported in the task 2/3 report (Voillequé et al. 1995). The simulation produced estimates of the maximum and minimum 1987 radon concentrations of $2.82 \times 10^7$ and $2.38 \times 10^7$ pCi $L^{-1}$, respectively (about 9% above and below the average for the simulation). Thus, the factor-of-two assumption that we have adopted for the uncertainty in this parameter seems more than sufficient, even when one allows for some variation of concentration within the headspace volume.

The radon dose percentiles in Table K-5 (Appendix K) reflect the increased variance attributed to the use of the headspace radon concentration and other parameters measured after 1979 for estimating releases of radon before the K-65 silos were sealed. Even with this substantial increase of the parameter variance, the 95th/50th-percentile ratio of the dose distribution for 1959–1979 increases only by about 17%. Other components of uncertainty, predominantly the air transport model calibration and the back-extrapolation of 1987–1991 meteorological data, dominate the composite uncertainty distribution. Additional information is given in Appendix Q (page Q-37).

**Effect of Rapid Turnover During 1959–1979 on Decay-Product Equilibrium and Gamma-Field Measurements on the Silo Domes**

Lacking adequate sampling of radon in either the silo headspaces or ambient outside air during the period before the K-65 silos were sealed, the *RAC* methodology relied partly on measurements of the gamma field near the silo domes taken before and after sealing to calibrate the release model. The NRC committee raised a question concerning "the effect of rapid releases during the early period on radon progeny equilibrium and hence on the gamma-ray measurements made at the silos" in the third bullet item on the second page of its letter to the CDC (the letter is reproduced at the end of this section). We interpret this bullet item as a request for a demonstration of consistency among the simulated pre-1980 quantities: net radon production into the silo headspace, release rate, decay-product equilibrium, and gamma field strengths resulting primarily from the $^{214}$Pb and $^{214}$Bi daughters in the headspace and from radioactivity from $^{226}$Ra and gamma-emitting decay products in the K-65 material.

Of these quantities, only the gamma field was not explicitly calculated, because the calibration depended only on the ratio of the gamma-field components due to the headspace radioactivity before and after the silos were sealed. The ratio was estimated from gamma exposure rate measurements taken on the silo domes at various times before and after the silos were sealed. In order to remove the component of the gamma field that resulted from radioactivity in the K-65 material, we used gamma exposure rate measurements that were

made in November 1987, when radioactivity had been evacuated from the silo headspaces during the operation of the Radon Treatment System (RTS). These measurements presumably would provide direct estimates of the field component due to the radioactivity in the K-65 material.

We emphasize that the calibration of the radon release model depended on the *ratio* of the gamma exposure rates from headspace radioactivity measured before and after the sealing of the silos — we emphasize the ratio rather than the absolute magnitudes. It is conceivable that the inferred equilibrium state of radon decay products in the silo headspaces, together with radioactivity in the K-65 material, might predict a gamma field with absolute magnitude that is inconsistent with the measurements. If such were the case, it would cast doubt on the model's calibration and thus on its predictions of release. We considered this possibility unlikely, but to respond to the NRC committee's continuing concern, we have performed gamma exposure calculations based on the geometry and material composition of the silos and the K-65 material, using the levels of $^{222}$Rn and the equilibrium ratios of decay products in the silo headspace that were calculated by the release model before and after the silos were sealed. Gamma-emitting radionuclides in the K-65 material are discussed in Appendix Q. The gamma exposure calculations were carried out by integrating a gamma point-kernel model over each source region (headspace and K-65 material) and using mass attenuation coefficients and exposure buildup factors appropriate to emitted energy spectra and elemental compositions of the shielding media (K-65 material, air, and concrete dome). The results are summarized by Figure A1.

The results of these calculations closely approximate the primary ranges of the measurements, including those taken during the RTS operation in 1987. The close agreement with the RTS data tends to indicate that the calculation accurately represents the component of the gamma field due to the radioactivity in the K-65 material. Figure A1 shows the computed ranges and the measurements.

Two measurements taken in May 1982 exceed the calculated range. We think it likely that these two measurements may have been taken with detectors unintentionally placed near points of leakage. It is also possible that temporal variations in the thermal pumping mechanism that partially drives the release of headspace gases might have permitted a temporary buildup in the headspace concentration of $^{222}$Rn and decay products, and that this buildup coincided with the measurements, although we consider this explanation unlikely. It is also possible that instrument miscalibration or localized uranium contamination on the dome could have led to the elevated readings. In any case, these two data points are not typical of other measurements made since the silos were sealed.

We consider the results of these calculations a further validation of *RAC*'s release model. The calculations and their results are discussed further in Appendix Q (page Q-40).

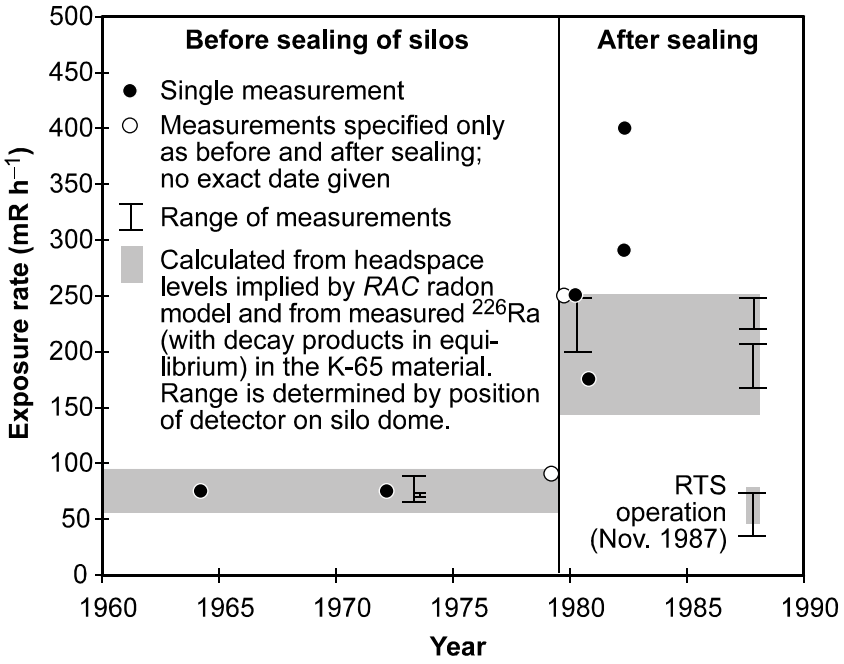


**Figure A1.** Gamma exposure field above the K-65 silo domes. Shaded areas indicate *RAC* calculations of the gamma exposure field based on headspace concentrations of $^{222}$Rn gamma-emitting decay products (primarily $^{214}$Pb and $^{214}$Bi) and on gamma-emitting radioactivity in the K-65 material. The ranges of the calculations show variations resulting from the placement of the detector on the dome. Most measured values lie within the calculated ranges. The two outlier points might have resulted from placement of the detector near a point of leakage, from a temporary buildup of radioactivity in the silo headspace, instrument miscalibration, local uranium contamination, or other cause.

### The Mechanism of Air Exchange Inside and Outside the Silo Headspaces During 1959–1979

The NRC committee requested that *RAC* develop additional information about mechanisms of air exchange between the silo headspaces and the outside before the silos were sealed. This request seems to express a desire to introduce into the assessment a process-level model that would afford some independent confirmation of the radon turnover times that have been estimated primarily from radiological measurements and mass-balance models calibrated to the measurements. The type of process-level model that comes to mind would be aerodynamic in character, simulating the velocity field of the interior air as it responds to pressure gradients driven by wind moving across the mouth of the pipe and over cracks and other penetrations in the dome. It would also depend on the physics associated with the heat budget of the air mass and on diurnal changes that would set up convection currents in addition to changes in volume.

An undertaking of this kind might be intrinsically interesting, but we feel obliged to point out some practical difficulties. First, a process-level model such as we have described would require advice from experts in fluid-mechanical turbulence. Simulations would require the solution of augmented versions of the Navier-Stokes partial differential equations, discretized on a spatial network of thousands (if not tens of thousands) of nodes. The effort could

involve many runs on a supercomputer. Moreover, the results would depend on numerous scenarios of assumptions (e.g., geometry of the cracks, heat capacity and conductivities of the concrete walls, domes, and the K-65 mixture, interior heat budgets under various assumptions about outdoor temperature, wind, and solar loading). And there is ample reason to doubt that the resulting range of estimates would be narrow enough either to exclude or to restrict the uncertainty distribution we have already computed from measurements of radioactivity and radiation fields. We cannot recommend extending the scope of the present project to include such an exercise, whatever its supposed merits.

If doubts about the radon source term persist, we suggest that consideration be given to developing an experimental approach, such as a scaled-down physical model of a silo that can be studied in a wind tunnel, possibly with a tracer gas. It is also possible that the empty silo 4 on the former FMPC site could be rigged for experimental studies with a tracer gas. Such studies, which are beyond the scope of the present project, would be expensive, and their cost would have to be weighed against their prospect of providing information that would confirm or negate the results predicted by the radon release model described in this report. The reasonableness of the remaining doubts about the model should also be carefully evaluated.

**Matters Related to Risk**

We note some of the points where the NRC committee concedes that it had misconceptions. These misconceptions included a belief that the risk-per-unit-exposure for radon used by *RAC* was too high (in fact, it was a central value), and that *RAC* had used a risk factor greater than 1 for females exposed to radon. The revised text of Volume I and Appendix S takes into account where these and other misconceptions occurred, and the new text attempts to present some of the complex matters of calibration and risk estimation more clearly. The new text also has a section on the combined effects of smoking and radon with respect to lung cancer, which addresses a concern expressed by the NRC committee. This section presents a derivation of a risk of radon-induced lung cancers twice as great for smokers as for nonsmokers. Thus, the nonspecific risks presented for the various scenario members (whose smoking status was not specified) should be multiplied by 1.41 for a nominal smoker and by 0.71 for a nominal nonsmoker. This has been indicated in the text (page S-32).

The NRC committee had objected to our use of a risk factor of 2 for children, maintaining that existing data on lung cancer fail to support such a factor. Careful review of the most recent data on induced lung cancer from the Lifespan Study (LSS) of the A-bomb survivors (Thompson et al. 1994, Pierce et al. 1996) reveals that, with respect to the dependence on age at exposure, lung is an exception among the organs and does not show a greater risk at young rather than old ages. The very limited information on lung cancer after radon exposure in children vs. adults (e.g., the young Chinese miners) also shows no greater effect in the young. This means that for both uranium and thorium and for radon, it is inappropriate to apply a correction factor for the young ages at which most of the scenario members were exposed. Again, while females generally have a higher risk of lung cancer in the LSS than males (Pierce et al. 1996), the difference is not large and is within the statistical uncertainties. Therefore, no correction for age and sex dependence has been applied to the lung cancer risk estimates in the final report. Furthermore, as the NRC committee recommended, further consideration has been given to *all* the uncertainties involved in risk estimation. The net re-

sult is that the median values of the risk estimates have dropped by a factor of approximately 2 from those presented in the draft report. However, because of the broader uncertainties, the 95th percentile values are almost identical with those presented earlier, and discussions relating to these will still apply. The 5th percentile values have been further reduced because of the expanded uncertainty distribution. The risk distributions are shown in Appendix S, Table S-27.

**Uncertainties in Dosimetric Parameters and Scenario-Related Quantities**

Scenarios 1–9 (Appendix J) are carefully constructed sets of assumptions about hypothetical individuals living near the FMPC during 1951–1988. Such scenarios make it possible to examine the sensitivity of dose estimates to assumptions affecting individual exposure to radioactivity from the FMPC. They might also be helpful to some individuals whose exposure histories appear similar to a scenario. But it has been emphasized throughout this report and its appendices (Volume II) that the scenarios may not correspond to known individuals and their exposure histories. Accordingly, the components of uncertainty that appear in the dose estimates express only uncertainties in release rates and environmental transport of the radionuclides considered in this study. Such components of uncertainty are real, deriving as they do from historical records and inferences from those records. By contrast, the dietary or other habits of the subject of scenario 1 (for example) are not considered uncertain, because this subject is not a real person, nor is she a surrogate for one or more real individuals with presumably similar behavior patterns (we do not yet know whether such individuals exist). Rather, the calculations for this fictitious individual are intended to show the results of possible patterns of exposure, but *not* to approximate dose that is known to have been delivered or risk that is known to have been incurred. Such approximation cannot be undertaken until information is developed about the FMPC-related exposure of real individuals and groups, which is a separate task that the CDC staff has in progress.

It is reasonable and instructive to use the scenario subjects for testing the sensitivity of calculated dose (or risk) to variations in the assumptions that define them (sensitivity analysis). But such an analysis emphasizes the relative importance of parameters and assumptions, and it differs, in both concept and execution, from an uncertainty analysis. In order to respond to the NRC committee's questions, however, we provisionally relax the distinction somewhat and consider some factors that might be deemed to fall into a gray area.

Specifically, the NRC committee suggests consideration of "physiologic factors, breathing rates, time spent indoors, turnover rates of indoor air, and so on, in the assessment of overall risk uncertainties." We assume that "physiologic factors," in the committee's usage, consist primarily of internal radiation dose coefficients (or dose conversion factors) and the myriad of assumptions and parameters that go into their estimation. For radon dosimetry, we take the range of predictions of accepted models as an indication of uncertainty, with estimates based on Figures 2B-2 through 2B-4 of the BEIR IV report (NRC 1988). These figures indicate data and model curves for three factors: minute volume (breathing rate), unattached fraction of decay products in the ambient atmosphere, and the particle size distribution for available condensation nuclei. We treat the factors as being stochastically independent and assume that the indicated ranges are 5th and 95th percentiles of lognormal uncertainty factors. The geometric standard deviation (GSD) of each factor will determine the factor's contribution of a component of uncertainty. Then we may examine the extent to which including such com-

ponents would perturb the distribution of dose for, say, scenario 1 as given in Table K-5 of Appendix K. We will return to this uncertainty component below, after we consider other parts of the NRC committee's list.

The NRC committee includes "time spent indoors" and "turnover rates of indoor air" in its series of concerns for uncertainty analysis. These factors are of primary concern for radon dose, because of its dominance for the FMPC site. A sensitivity analysis of the ratios of dose rates for indoors over outdoors is shown, for various distances from the source and other factors, in Table I-10 (Appendix I). The ranges of variation in this table give a strong sense of the relative importance of these two modes of exposure. When all combinations in the table that are based on the estimated pre-1979 headspace equilibrium are considered (1 : 0.97 : 0.86 : 0.78, right-hand half of the table), the ratios range from 0.61 to 1.01 (indoors/outdoors). If we assume an arbitrary baseline dose rate of 1 mSv $day^{-1}$ for outdoor exposure, the maximum "uncertainty" would obtain for someone who remains indoors all of the time, and the interval in that case would be 0.61 to 1.01 mSv $day^{-1}$ (geometric mean 0.78 mSv $day^{-1}$ and GSD 1.17, corresponding to a 90% uncertainty interval of about 29% if we interpret 0.61 and 1.01 as 5th and 95th percentiles, respectively, of a lognormal distribution). One of the factors considered in Table I-10 and included in the given uncertainty interval is the turnover rate of indoor air, based on the distribution of annual average values for a climatic region that includes southern Ohio, taken from the tabulation of Murray and Burmaster (1995).

Now let us illustrate the potential significance of indicated distributions for dose coefficients combined with breathing rates, and for indoor/outdoor exposure to radon decay products. The scenario 1 distribution of radon dose has 50th and 95th percentiles 3.6 and 14 Sv, respectively (Table K-5, Appendix K). This distribution includes the augmented uncertainty that we discussed above for the releases of radon before mid-1979. For the present purpose, this dose distribution is reasonably represented by a lognormal form with geometric mean (GM) 3.6 Sv and GSD 2.28. When this distribution and the uncertainty factors described above are combined by forming the product of their respective random variables, the result, or composite, is a lognormal distribution with geometric mean 3.6 Sv and GSD 2.51, with 95th percentile increased from 14 to 16.4 Sv. The analysis is summarized in Table A1. The introduction of these components of uncertainty results in an increase of 17% in the 95th percentile of the Scenario 1 radon dose (a corresponding decrease in the 5th percentile would occur). Nearly all of the increase comes from the uncertainty in the radon dose conversion factor; the indoor/outdoor component, including air exchange, is practically negligible in its effect on the overall uncertainty (by itself, it would account for an increase of less than 3% in the 95th percentile).

As Table A1 shows, the explicit inclusion of these components of uncertainty would not constitute a dramatic change in the uncertainty distributions in Table K-5, and we have not included them in the calculations for reasons discussed previously. However, the numbers given in the example for scenario 1 should convey a sense of the likely modest expansion of the uncertainty distributions for dose when these models and data are applied to real individuals and populations for whom valid parametric uncertainties can be introduced. But as a matter of perspective, one needs to keep in mind that the uncertainty components related to the atmospheric transport and the temporally limited meteorological database tend to overwhelm lesser components such as the ones that interested the NRC committee.

**Table A1. Uncertainty Analysis for Radon Dose for Scenario 1**

| Lognormal distribution | GM | GSD | 95th percentile |
|---|---|---|---|
| Scenario 1 radon dose (Sv) | 3.6 | 2.28 | 14.0 |
| Uncertainty components (dimensionless): | | | |
| Dose conversion factor | | | |
| Minute volume | 1 | 1.22 | |
| Unattached fraction | 1 | 1.16 | |
| Particle size | 1 | 1.33 | |
| Indoor/outdoor and air exchange[b] | 1 | 1.17 | |
| Composite distribution (Sv) | 3.6 | 2.51 | 16.4 |

[b] Maximum uncertainty for this component, because it is based on an individual who always remains indoors.

**Questions of Interpretation and Presentation**

The NRC committee raised objections to the use of graphs with broken ordinate axes for presenting dose and risk in the summary brochure, expressing fear that such graphs might be easily misunderstood. The graphs in question have been redrawn with the committee's comment in mind. The committee also felt that the report and brochure tended to overemphasize the upper percentiles of uncertainty distributions of dose and risk and asked that more attention be given to mentioning lower (e.g., 5th) percentiles. We have included such mention at appropriate places, and we grant that such balance is desirable. But we must point out that it is the high side of the possible risks that concerns the public the most and that will always receive the greatest attention, regardless of any competing emphasis on the low side that we might provide.

Our comparison in the draft report of the (essentially) 27-year radon exposure of a long-term resident of the Fernald area to 27 years of exposure to naturally occurring radon that infiltrates homes caused the NRC committee some concern (27 years corresponds to the period before the K-65 silos were sealed). The committee considered a comparison with a lifetime exposure to naturally occurring radon more appropriate. Volume I of this report includes that comparison. Similarly, the report includes comparisons of the Fernald radon exposures for scenarios 1–9 to the EPA action level.

The NRC committee acknowledges that no one yet knows the best way to communicate health risks to the public, and it expresses confidence that *RAC* has striven for a balanced approach to its presentation. We confirm the high priority of that goal in this work, and we assure the reader that we have frequently reminded ourselves of it as we prepared this report.

**The Assessment Domain for the Study**

As a result of litigation brought against the Department of Energy (DOE) in the 1980s, a region within 5 miles (8 km) from the FMPC site boundary was prescribed by the court for purposes of the settlement. The DOE undertook an extensive soil and vegetation sampling within a circular region, centered on the site, with a 5-mile radius. These samples provided a

significant fraction of the database that was available for dose reconstruction, although they were taken at essentially a point in time (a serious limitation for some of our purposes). That same circular region was adopted as the domain of a dose and risk assessment commissioned by DOE and carried out by IT Corporation (IT 1989).

The same circular domain was initially adopted for the present study and was subsequently extended to a 10-km radius at the request of the CDC. The reasons for not using an even larger domain at that stage were primarily related to capabilities of the models and the spatial extent of existing data with which to calibrate (or validate) them (in particular, the DOE soil and vegetation data all lay within the 8-km circle previously mentioned). Beyond the 10-km boundary, it was our judgment that predictions would become increasingly speculative and of less utility for the specific purposes anticipated by the CDC staff for epidemiological power analysis. Other dose reconstructions, such as the Hanford Environmental Dose Reconstruction (HEDR), have been implemented for much larger domains, but the potential dose and risk suggested by the magnitude of the source term provided at least partial justification for such extended designs, and the initial choice of methodology for those studies was made with the large domain in mind.

For the Fernald study, most of the attention was initially focused on the reconstruction of the uranium source terms (releases to air, surface water, and ground water), and preliminary estimates of uranium doses at 8 and 10 km from the site did not seem to support any argument for extension at that time. Although the IT Corporation's assessment had identified radon as having risk potential comparable to that of uranium at the location of maximum risk, we expected our reconstruction of the uranium source term to increase the estimated doses from uranium relative to radon (the IT Corporation's study had used a DOE source term for airborne releases that assumed zero releases for some periods for which records had not been located). No one anticipated what our investigation of radon emissions from the K-65 silos would reveal.

It would be unproductive at this time to consider extending and redesigning the present study for a larger domain. In the best tradition of successful engineering practice, we froze the design (i.e., models, domain, dosimetric methods, scenarios) early enough to enable us to bring the work to a conclusion. We recommend that the CDC staff and the NRC committee use the information in this and our previous reports and the computer programs that have been transmitted to them, together with the demographic information that the CDC staff has developed, and in consultation with interested parties, decide where further resources would be best invested. Such decisions might well inform themselves from a cost-benefit analysis of dose-risk reconstruction in a larger domain (incremental power of an epidemiological design might be one "benefit" in such an analysis).

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